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Ecological and socio-cultural potential of human modified forest landscapes in conservation of tree species diversity: the case of Vhembe biosphere reserve, South Africa

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DEDICATION

To my beloved mother, Beletset Hailemariam Tesfagaber, my late father, Ghebreslassie Araia Uqbu and my late friend, Dr Mota Lesoli

DECLARATION

I, **Mulugheta Ghebreslassie Araia**, declare that this thesis, which I hereby submit for the degree of PhD in Forest Science at the University of Pretoria, is my own work and has not previously been submitted by me for a degree at this or any other tertiary Institution.

Signature:

Date: 24/01/2020

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SUMMARY

This thesis aimed at evaluating the potential ecological and socio-cultural values of Human Modified Forest Landscapes (HMFL) in comparison to the State-protected Indigenous Forests (SIF). This was crucial in light of the increasing human domination, deforestation, and fragmentation of landscapes, and the consequent alarming rate of global biodiversity loss. At the same time, there is growing scepticism on the performance of protected areas as a standalone strategy for biodiversity conservation. However, the proponents of protected areas still pursue the expansion of its coverage to half of the earth surface, aptly known as the "Half Earth" (HE) movement. HE proponents presume that resource use-behavior is determined by ecological abundance of resources, and view local people as a threat to biodiversity.

In parallel, the opponents of the HE conservation option advocate for the New Conservation Science (NCS) and denounce the scientific basis of protected areas as a denial to the remarkable resilience of nature and the exclusion or restriction of local people to their cultural landscape as unethical. NCS proponents argue that effective management of Human Modified Forest Landscapes (HMFL), that embraces culture/Traditional Socio-Ecological Knowledge (TSEK), is more important for biodiversity than protecting relatively intact forest ecosystems that exclude local people. Studies have shown that there is a higher compliance rate by traditional society to social norms that govern cultural protected forests, than to rules imposed on state indigenous forests. There is currently no consensus about whether HMFL or SIF is a better option for biodiversity conservation; hence, the need for this study.

Furthermore, although there is a claim by previous studies on the positive link between the Vhavenda people's culture and surrounding ecosystems in the Vhembe Biosphere Reserve (VBR), South Africa, there has not been a systemic evaluation of the potential of HMFLs for conservation of tree species diversity from a socio-ecological perspective.

The study was conducted in two Forest Reserves (FRs) of the Eastern Soutpansberg forest landscape, which are part of VBR; namely Mafhela Reserve (MFR) and Thathe Vondo Forest Reserve (TVFR). MFR is relatively more simplified or fragmented by disturbance than the complex TVFR. First, tree-based traditional land use regimes of the two FRs were categorized into two major groups: (i) *Human modified forest landscape* (HMFL), under the custody of traditional authorities that comprises highly disturbed Common Resource Use Zones (CRUZ), intermediately disturbed Trees Along Streams and Rivers (TATR) and

minimally disturbed Culturally Protected (sacred) Forest Areas (CPA) and (ii) *State-protected Indigenous Forests* (SIF). Four independent studies were conducted to address the aim of the study (Chapters 2-5).

Using landscape moderation insurance and Intermediate Disturbance Hypothesis (IDH) as a framework, the study assessed the difference in the resilience of local assemblage along a land use intensity gradient. Univariate and multivariate statistical techniques, including various ordination diagrams, were used to analyse the beta (β)–diversity (Jaccard dissimilarity matric on presence/absence data) and the difference in species richness in HMFL. This study showed that land use disturbance affects mean beta (β)–diversity in an ecological contrasting manner between the simplified MFR and a complex TVFR. In MFR, while the species richness along a land use gradient response did not conform to the predication of IDH, the species composition changed along the land use gradient. In contrast, the local assemblages in TVFR showed that, while the species richness conformed to IDH, the species composition was more resilient to change. However, resilience does not mean the absence of dynamism in local assemblages. Even in the absence of human disturbance, the local neighbourhood effect together with biotic and abiotic elements, may still incur small scale changes in species composition.

In order to evaluate the efficiency of the protected area for the conservation of rare species, responses of different facets of beta (β)–diversity, using the Hellinger Distance Matrix and species abundance distribution, were used. This was crucial to disentangle the conservation value of alternative options of conservation for rare species, such as canopy and sub-canopy trees of a wide geographic range and endemic species, from the overall biodiversity value. The result showed that SIFs in the two FRs differed in their efficiency of protecting rare species, in comparison with each of the land use regimes of the HMFLs. In MFR, all land use regimes lost species richness, became more uneven, and the dominance of trees declined and was replaced by shrubs along the land use gradient in comparison with SIF. In contrast, SIF in TVFR was not exceptionally efficient compared to others, except with intensively disturbed CRUZ, where the decline of the relative abundance of rare species was observed. Almost all land uses in both FRs retained some endemic trees that were either rare or absent in SIF.

Using ecological assessment and ethnobotanical techniques, the study also tested the ecological appearance hypothesis of whether an ecological abundance of resources can sufficiently explain the use-behaviour of traditional society for various forest and tree species' utilities. Both parametric and non-parametric tests were used to analyze the data gathered from

135 households. The result from the two communities (two villages from each FR) of the Vhavenda people revealed the homogeneity of cultural values pertaining to resource use-behaviour, although they reside in ecologically different forest landscape conditions. The use-value of habitats increases with the increase of the land use intensity gradient in the multifunctional landscape as defined by cultural norms and taboos, with SIF as an exception. Despite its presumed strict protection status, SIF had the same use-value as with open access CRUZ. Almost no forest resource harvesting was reported from CPA. Moreover, abundance of species did not sufficiently explain the use-value of species. The findings show that culture plays a more predominant role in explaining use-behavior than an abundance of resources.

The study further compared the compliance behaviour of local communities towards rules that govern CPA and SIF. The findings showed that compliance behaviour of local people to CPA was a social norm while non-compliance to SIF appeared to be a new norm. On average, an individual holds about four out of the seven perceived local values that motivated the need to conserve forest and tree species diversity. These include the need to protect forests and trees of outstanding utility values, watershed protection, and cultural values (identity and symbolic value), protection of endangered species, and wildlife habitat, in descending order. This shows that, unlike the dichotomy of value orientations between HE and NCS proponents, local people held a continuum of biocentric and anthropocentric value orientation in managing their landscapes. Almost all of the participants in the study showed their willingness to take part in conservation. The study suggests that SIFs become more vulnerable to non-compliance when the necessity of resources for rural livelihoods arise due to misalignment of the rules to local values, social norms, and taboos; not because of ignorance.

Based on the findings of the study, the potential of HMFLs for the conservation of tree species diversity over the effectiveness and efficiency of protected areas depends on the prevalent socio-ecological context. It demands context-specific policies that are informed by interdisciplinary science using complimentary theories and models rather than a mere dichotomy of conservation based on ideological options of NCS and HE. The claim by the NCS on the remarkable resilience of nature is an oversimplification of a complex and non-linear response of local assemblages to disturbance that might misinform future conservation strategies. Equally, the expansion of protected areas of the HE option, with the presumption that local people are threats to biodiversity, may be deemed to fail. Instead, inclusive policy reform that integrates local peoples' cultural values and TSEK, either for restoration or

conservation of biodiversity, depending on the existing landscape context, plays a significant role. An improved landscape complexity improves the resilience of local species diversity.

LIST OF ACRONYMS

CPA	: Cultural Protected (sacred) Forest Area
CRUZ	: Common Resource Use Zone
*db*RDA	: *distance-based* Redundancy Analysis
DISTLIM	: Distance-based Linear Modelling
FR	: Forest Reserve
HMFL	: Human Modified Forest Landscape
IDH	: Intermediate Disturbance Hypothesis
LCBD	: Local Contribution to overall Beta-diversity
LEDET	: Limpopo Department of Economic Development, Environment and Tourism
MFR	: Mafhela Forest Reserve
NCS	: New Conservation Science
nMDS	: non-Matric Multidimensional Scaling
PCO	: Principal Coordinate Analysis
PERMANOVA	: Permutational Based Multivariate Analysis of Variance
PERMDISP	: Test of Homogeneity of Dispersion
SIF	: State-protected Indigenous Forests
SIMPER	: Similarity Percentage
TATR	: Trees Along Rivers and Streams
TSEK	: Traditional Socio-ecological Knowledge
TVFR	: Thathe Vondo Forest Reserve
VBR	: Vhembe Biosphere Reserve

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CHAPTER 1

1.1 Background to the study

Traveling through the Vhembe district municipality in Limpopo province, South Africa, one can observe the diversity of tree-based land use regimes that epitomize the inextricable link between nature and culture. In recognition, the whole municipality was officially designated as the Vhembe Biosphere Reserve (VBR) by the United Nations Educational, Scientific and Cultural Organization (UNESCO) in 2009 (Pool-Stanvliet 2013). Despite the availability of a coarse-scale bioregional plan (LEDET 2017), VBR does not have a clear demarcation of core, buffer, and resource use zones at the local level (Mauda 2016). However, one can casually observe along the way the spatial distribution of tree-based traditional land use regimes ranging from highly human simplified landscapes to relatively intact forest landscapes. The diversity of land use types and conditions of forest fragments in those landscapes present diverse potentials and challenges for conservation of forest and tree species diversity.

In particular, a vast area of the original forest biota of the Eastern Soutpansberg mountain has been transformed into settlement areas, commercial tree plantations, tropical fruit orchards and subsistence agriculture. In some places, the remnant indigenous forest fragments, which are mostly embedded in those commercial plantations, have been reserved under state-protected indigenous forest regimes for biodiversity conservation (Symes et al. 2000). In contrast, most of the remnant indigenous forest fragments, managed under the governance of traditional leadership, are continuously subjected to different land use disturbances or actively modified by the surrounding rural communities.

Many studies have documented the overall flora (Hahn 2006), the tree species diversity of Eastern Soutpanserg (Mostert 2006). Those studies highlighted that the Eastern Soutpansberg mountain forest is one of the richest plant biodiversity hotspots and a centre of plant endemism in southern Africa (Hahn 2006). However, from a casual observation, it is difficult to ascertain the current status of forest and tree species diversity of local assemblages except in the commercial plantations that are mostly dominated by few monoculture tree crop species. A local assemblage of tree species is considered to be highly diverse if many equally abundant (frequent in occurrence) or nearly abundant species are present in a particular area. On the other hand, it will be considered less diverse, if it composed of very few species or if only few species are abundant (Brower 1998). However, species diversity of local assemblages of tree species between state-protected indigenous forests and tree-based traditional land use regimes at different spatial scales, appears to be idiosyncratic. In some places, the local assemblage of different tree based traditional land use regimes appeared to be almost identical in species diversity among themselves including the identity of species that are co-existing (species composition). In other places, local assemblages of tree based traditional land use regimes hold the same or higher number of species present (species richness) with the state-protected indigenous forests even if there was a noticeable difference in their species composition or species abundance or both.

Moreover, some studies have been claiming that local people's behaviour on forest use for rural livelihoods is a non-random practice. It is shaped by cultural values, norms, and taboos of the local people (e.g. Araia and Chirwa 2019). Over the recent years, many studies documented the ethnobotanical values of trees and other plant species (Magwede and van Wyk 2018; Constant and Tshisikhawe 2018) and the role of good culture in conservation of the Eastern Soutpansberg mountain forests (Mutshinyalo and Siebert 2010; Khorombi 2001; Sikhitha 1999). Despite the overall claim on the prevalence of good culture and high tree species diversity of Eastern Soutpansberg (Mostert 2006), there has not been any study that linked culture and spatial distribution of species diversity. Given that the fact that disturbance affects different species differently, it is difficult to ascertain which species is prone to local extinction (local extermination) or resilient to human disturbance or cultural practices.

Moreover, despite active modification of forest landscapes in Eastern Soutpansberg has been ongoing for at least the last 4000 years (Baboolal 2014), both the species diversity (Hahn 2006) and the cultural values, norms, and taboos have been under immense threat over the last 150 years (Constant and Tshisikhawe 2018). Consequently, the persistence of the indigenous forests and tree species diversity and local people's pro-biodiversity culture into the future might be uncertain. State conservation agencies have protecting certain forest area as a precautionary measures to safe guard areas that are considered forest biodiversity hotspots. However, to the best of our knowledge, the effectiveness of state-protected indigenous forests and the potential of the human modified parts of Eastern Soutpansberg for conservation have not been systematically evaluated using a comparable ecological matrix. Thus, this motivated the need for the study.

1.2 Rationale and justification of the study

1.2.1 Global forest and tree species diversity and the performance of protected areas

Based on the existing literature, the major threats for the global and tropical forest ecosystems in many localities do not appear to be different (Cadotte et al. 2017) from the VBR. Although humanity has been actively modifying tropical forests for at least 45,000 years (Roberts et al. 2017), the recent rate of decline is alarming. The global primary forest cover has declined by about 35% from its original coverage only over the last three centuries (Watson et al. 2018), mainly due to logging, expansion of agriculture, and human settlement (Watson et al. 2018; Poker and MacDicken 2016; Kissinger et al. 2012). This must be concerning given that tropical forests only cover less than 10% of the global terrestrial ecosystems and yet still harbours two-thirds of global biodiversity (Giam 2017). More so, considering the recent claim that the planet is in the process of undergoing the sixth mass extinction event of biodiversity and its life support systems due to human disturbance (Seddon et al. 2016; Corlett 2015). In response, nation-states have been increasing the area coverage of designated forest protected areas at a global scale from 7.7% in 1990 to 16.3% in 2015 (Morales-Hidalgo et al. 2015). However, different global assessments on the performance of protected areas in the tropics have been suggesting that the majority of strictly protected areas are a failure (Oldekop et al. 2016; Watson et al. 2014).

Ecologically, there has been growing criticism of the adequacy of the protected area to represent and cover many valuable habitat heterogeneities and species (Watson et al. 2014; Joppa and Pfaff 2009). Hence, increasing areas of protection may not automatically imply protection of conservation priority areas (Watson et al. 2016). In addition, most of the few remaining tropical forests are embedded within human modified forest landscapes (Noble and Dirzo 1997). Hence, putting those fragments under strictly protected areas increases their ecological isolation and exposure to spill-over effects from the habitat loss and fragmentation of the surrounding ecosystems. This will make them more susceptible to species extinction and habitat deterioration over time (Kremen and Merenlender 2018; Laurance et al. 2012).

Socially, the establishment and management of protected areas has been criticized for the forceful exclusion of local people from ancestral settlement areas or significantly restricting the use of their landscape in a traditional way (Adams 2017; Lele et al. 2010). At the same time, the enforcement of strict protection rules in state-protected indigenous forests (e.g., restriction of access for fuelwood) by forest guards and legal systems have proven to be ineffective (Stern 2008), expensive and antagonistic (Lele et al. 2010; Wilshusen et al. 2002). Consequently, most protected areas have been suffering from non-compliance to conservation rules which is becoming a universal challenge (Solomon et al. 2015; Arias 2015). These ultimately will undermine the goal of biodiversity conservation.

1.2.2 The debate over the potential role of human modified forest landscapes for the conservation of tree species diversity

Despite more than half of the global terrestrial ecosystems having already been altered into human dominated landscapes, the global trend on the extent and severity of local level landscape modification varies from place to place (Ellis et al. 2010). There is no doubt that the expansion of commercial agriculture, forest plantations, and logging have been driving large scale tropical deforestation and fragmentation (Watson et al. 2018). Yet, tropical forests appear to be more resilient to human disturbance (Bhaskar et al. 2018; Norden at al. 2009; Wright 2005). More than a quarter of areas held by primary forests in the tropics undergo secondary succession after clearance annually. This has made secondary forests the dominant land use in human modified landscapes (Arroyo- Rodríguez et al. 2017) and a potential reservoir for biodiversity and ecosystem services (Chazdon et al. 2009). On the extreme side, some have argued that the higher species diversity in the tropical forests is the result of the concurrent effect of natural and anthropogenic disturbance (Van Germerden et al. 2003; Connell 1979).

Moreover, there has been a claim that traditional society, which occupies about 37% of all the remaining global natural lands (Garnett et al. 2018), often actively modifies their landscapes in a manner that supports biodiversity conservation (Yang et al. 2018; Sobrevila 2008, Gadgil et al. 1993). Traditional society often uses the local bio-diverse natural resources as both the motive and the mechanism for ecosystem management (Lyver et al. 2019). In the process, they culturally construct a niche filled with traditional socio-ecological knowledge systems (TSEK), a complex set of understanding, beliefs, and practices, which is suitable to adapt to the locally changing ecological conditions (Berkes et al. 2006; Gadgil et al. 1993). Many studies (e.g. Paneque-Gálvez et al. 2018; Sutherland et al. 2003) have been attesting that TSEK/culture and local biodiversity have an inextricable interdependence. The richness of TSEK has often been associated with pro-ecological behaviour and as a driver for positive and novel environmental innovation and social returns (Lyver et al. 2019). Thus, there is a growing call by Intergovernmental Science Policy platform for Biodiversity and Ecosystem Services (IPBES) (Tengö et al. 2017; Díaz et al. 2015) and others (e.g. Infield et al. 2018) to embrace culture and TSEK as it has the potential to ensure the parallel conservation of nature and culture. It also increases the likelihood of success of conservation initiatives by promoting local priorities and support (Infield et al. 2018; Redmore et al. 2018).

Jointly, the resilience of tropical forests after human disturbance (Arroyo- Rodríguez et al. 2017; Melo et al. 2013) and the potential of the rich TSEK of traditional society (Yang et al. 2018; Sobrevila 2008, Gadgil et al. 1993) have raised hope for biodiversity conservation in human modified landscapes. These have led to some conservationists arguing in favor of effective management of human modified ecosystems (e.g. traditional land use) for biodiversity conservation, in contrast to protecting undisturbed forest ecosystems (Watson et al. 2018).

Ecologically, there is growing evidence that intact primary forests have exceptional value in terms of ecosystem services and functions over human modified forest landscapes including global and regional climate regulation, local climate, biodiversity conservation, and watershed regulations (Watson et al. 2018). Also, species differ in their response to disturbance (Simons et al. 2015; Gardner et al. 2009). However, there is insufficient knowledge of whether vulnerable rare species (e.g. old-growth forest trees, endemic species) are able to maintain viable populations in human modified landscapes (Arroyo- Rodríguez et al. 2017; Melo et al. 2013; Gardener et al. 2009). Thus, it is inappropriate to assume that all forests (Watson et al. 2018; Gibson et al. 2011) and all species, regardless of their population status (Waltert et al. 2011), have equal conservation value in assigning priorities during conservation planning.

Sociologically, Low (1996) and Low and Heinen (2017) stated that our perception of traditional society as being deliberate, cooperative and respectful of their surrounding nature is a fallacy. They have a keen interest in exploiting resources when favourable opportunities and demands arise (Terborgh 2004). The low impact of traditional society on the environment is not from a conscious effort of individuals to bear short term costs with long term collective benefits in mind. It is just a combination of low population density, inefficient harvesting extraction technologies and a lack of profitable markets. Instead, it can be hypothesized that forest and tree use-behaviour to sustain rural livelihoods is ecologically driven (e.g. abundance of resources) (Low 2014; 1996). Thus, the presumption of traditional society as ecologically conscious and voluntarily willing to safeguard biodiversity may mislead future conservation strategy.

In summary, there is a growing recognition of the importance of conserving tropical forests and tree species diversity, and the dire consequences to ecosystem services and functions if lost. Yet, the above disagreement hints at the conflicting and diverse opinions on "how" and "where" to tackle the problem (Gavin et al. 2018). Instead, some conservationists (e.g. Wilson 2016) have been pursuing the notion of a substantial increase in stricter protected areas to cover half of the earth's surface, the "Half Earth" option. Others have been calling for a "new conservation science" option (e.g. Kareiva et al. 2012; Marvier 2012).

1.3 Philosophical arguments underpinning biodiversity conservation

The current debate on pursing the Half Earth (HE) or New Conservation Science (NCS) options as alternatives is not a trivial academic one. The arguments are deeply entrenched in the contrasting underlying value orientations on how the two proponents' view the relationship between human culture and nature. Thus, the proponents of each approach have been pursuing an alternative vision, policy instruments and goals for conservation applications (Gavin et al. 2018; Mace 2014). In particular, the new conservation science has prompted a tremendous change in the ethical imperatives of conservation. Some leading conservation agencies have redefined the vision, mission and approaches to conservation (Pearson 2016; Doak et al. 2015).

Wilson's (2016) proposal of pursing the HE option is based on a biocentric value orientation that has been a leading tradition for past and present biodiversity conservation initiatives. It suggests a substantial increase of stricter protected areas to cover half of the earth's surface just for the sake of the intrinsic value of nature (e.g. species and ecosystems). Based on the theory of island biogeography, researchers suggest that protecting 50% of the earth's surface would conserve about 85% of all species (Gavin et al 2018). However, protected areas do not remain ecologically static just because they are accorded a legal designation. They are subjected to continuous change due to concurrent external pressures (e.g. land use history, the condition of the surrounding forests) and internal dynamics (e.g. natural succession). Hence, biodiversity conservation demands an adaptive management approach (Messier et al. 2015; Wallington et al. 2005).

Moreover, the HE approach fails to recognize how often trying to solve biodiversity conservation challenges, solely as an ecological problem, does not yield good results (Cumming and Allen 2017). Biodiversity loss and their conservation are inextricably linked with a social process (Ban et al. 2013). With the growing scarcity of large tracks of tropical forests (Melo et al. 2013) and conservation resources (Walls 2018), how the HE option can be practically implemented, without compromising the access of traditional society to natural resource use, is questionable. Others argue that treating traditional society as equal to the highly consumptive citizens who have been playing a dominant role in the global biodiversity crisis is unethical (Shoreman-Ouimet and Kopnina 2015). Overall, the opponents of HE suggest that the traditional conservation approach has been unsuccessful, lacks a scientific basis and is not appealing enough to gain public support (Gavin et al. 2018).

On the contrary, the NCS approach (e.g. Kareiva et al. 2012; Marvier 2012) acknowledges the inextricable link between nature and people/culture. The NCS relies on the premise that intrinsic values of nature can still be maintained or enhanced while still providing anthropocentric values to people (e.g. ecosystem services). However, the NCS empathetically alleges the concept of pristine nature as a socially constructed myth and emphasizes the remarkably resilient capacity of nature over human influence. As a result, the NCS calls for re-focusing conservation efforts towards human dominated landscapes. The proponents of NCS underline that conservation of nature for people may gain greater support.

However, some conservationists have been criticizing the NCS approach for exaggerating the resilience capacity of nature after disturbance while ignoring the dire state of global biodiversity and rapid extinction of vulnerable species (Gavin et al. 2018). In fact, some biodiversity crises (e.g. species extinction) are irreversible (Doak et al. 2015). The response of local species diversity (e.g., species richness and the recovery of species composition) after disturbance over time either increases, decreases and or remains the same (Vellend et al. 2017).

Although disturbance is an inherent part of nature, the impact of human disturbance on species diversity is idiosyncratic (Fox 2013), and affects vulnerable tropical forest biodiversity adversely. For instance, some tree species can be rare, either due to their low population density as in most of the old-growth forests in the tropics (Barlow et al. 2010), or narrow geographic range (endemicity) or both. These put rare species at a higher risk of extinction (Pimm et al. 2014). This risk can also be aggravated by land use disturbances that directly remove rare species and may also create favorable conditions for their replacement by wide geographical range species (Waltert et al. 2011). The replacement may offset the loss of local species richness but reduces spatial heterogeneity (Hillebrand et al. 2018). This may adversely affect the same people which NCS is advocating for, either by shifting the spatial distribution of locally available ecosystem services, or the extinction thereof.

Others, such as Watson et al. (2019) state that an emphasis on ecosystem services for direct human benefit may shift the focus to human modified landscapes that do not support higher biodiversity values. Kopnina et al. (2016) state that the emphasis on the anthropocentric value of nature will discriminate against many species, especially those that may not have direct human use-values. Despite the acknowledgment of the marginal influence of humanity on many global ecosystems, Caro et al. (2012) showed that there are still places in many parts of the world that are not severely compromised by human intrusion. If it were not for the past decades of conservation measures, biodiversity loss of most ecosystems would have been far greater than the current record (Godet and Devictor 2018). The causes for perverse outcomes of most of the protected areas are mainly due to a lack of proper planning and their effective management thereof (Barnes et al. 2018). However, well-planned, resourced and properly managed protected areas have been proven to be effective in conserving biodiversity (Geldmann et al. 2015). Overall, the opponents of NCS argue that the priorities of NCS rest on ethical values, not in good science (Doak et al. 2015).

Different framing of the relationship between nature and culture (Mace 2014) and the debate on the agony of choice of purpose and place (Collen 2015) in conservation are not new. The fact that both HE and NCS agree on the substantive matter, the need for biodiversity conservation, their difference in opinion can still provide diverse conservation opportunities (e.g. habitat and species protection, restoration, sustainable management) (Godet and Devictor 2018; Green et al. 2015). Aichi biodiversity targets (strategic plan for the implementation of convention to biodiversity) acknowledge the need to pursue the expansion of protected areas alongside capturing the potential of other Area-based Conservation Measures (OECMs) (Dudley et al. 2018). The later strategy includes the essential role of human modified landscapes for biodiversity conservation. However, the fact that both proponents reciprocally criticize the validity of their scientific arguments only highlights the need for evidence-based conservation approaches (Cook et al. 2010; Sutherland et al. 2004) to avoid giving credit where it is not due (Watson et al. 2016). Simply extending the areas of global protected area networks (Barnes et al. 2018) or granting legal designations of areas alone do not amount to better protection of forest and tree species diversity (Hayes and Ostrom 2005).

Recently, there has been a recognition of the importance of interdisciplinary research in ecology in assessing the impact of disturbance on species diversity (Cadotte et al. 2017), the need for integration of different sources of knowledge (Fernández-Llamazares et al. 2015; Poe et al. 2014; Raymond et al. 2010; Berkes et al. 2006) and in framing biodiversity conservation as a socio-ecological issue (Ferreira et al. 2018; Ban et al. 2013). This is largely due to the recognition of the growing human dominance on nature which renders the distinction between nature and socio-cultural realms in scientific inquiry obsolete (Kueffer et al. 2015). However, this is not to undermine the importance of mainstream ecological theories and models.

The existing ecological theories and models are helpful in setting up a nil hypothesis to compare the impact of human disturbance (Cadotte et al. 2017) with the surrounding intact ecosystems. Despite the lack of a coherent framework to assess the effect of land use disturbance on species diversity (Resasco et al. 2017), existing theories still remain relevant to guide the future of conservation policies (Patterson and Williams 1998). For instance, some studies used the Intermediate Disturbance Hypothesis (IDH) (Connell 1979) and the Landscape Moderated Insurance Hypothesis (Tscharntke et al. 2012) to assess the resilience of species diversity.

In short, Connell's (1979) IDH predicts that an intermediate disturbance intensity, frequency, or extent would maximize the species richness (the number of species) of an area by delaying the competitive exclusion or promotion of co-existence between competitively inferior and dominant species together in space and time, which cannot be sustained under too rare or severe a disturbance (Yuan et al. 2016; Connell 1979). On the contrary, Tscharntke et al. (2012) state that landscape complexity, or the condition of the existing forest, provides spatial and temporal insurance; i.e. higher resilience and stability of ecological processes in a changing environment. By implication, predicting the resilience of local species diversity using IDH, without considering the moderating influence of the whole landscape, may not provide an accurate picture. The central question is then to understand under what landscape condition does IDH becomes an effective theory? There has been considerable conceptual and empirical achievements in the field of theoretical ecology. However, it does not mean that theoretical does not have limitations.

First, many studies have been using species richness matrix as a surrogate for change of species diversity in response to human disturbance. However, the loss /change of species diversity is a multifaceted construct that includes not only change in richness but also the change in the identity of co-existing species (species composition) and their relative species abundance(Hillebrand et al. 2018). The relative species abundance refers to the relative frequency of a species or how common or rare a species are in relation to overall species abundance in a given sampling unit (Gaston and Spicer 2013). In recent years, many authors have been questioning if the species composition and relative species abundance respond the same way as richness at different spatial scales to human disturbance. In response, Avolio et al. (2015) suggested a framework that links β-diversity (Change in local assemblage at different spatial scale) and species abundance distribution, as a tool to monitor the impact of global environmental change on biodiversity. The authors argue that disturbance can cause a change in overall β-diversity of a landscape either by changing (i) the mean dissimilarity of local assemblages along land use intensity gradient (Mean β-diversity); or (ii) the homogeneity of within-group variability of local assemblages among different disturbance regimes (Variance β-diversity); or (iii) simultaneously. The authors further suggested linking the results of β-diversity studies with species abundance distribution (i.e. the distribution technique that shows the relative abundance of all species constituted in the local assemblage) for better results. However, some authors (e.g. Hanspach et al. 2010) argue that the response of species abundance distribution models rely on the species trait under consideration. In this study, in addition to the different facets of β-diversity using species presence/ absence and species-abundance matrices, we linked the response of species traits to identify which species identities are vulnerable to change in their abundance (See Chapters 2 and 3 for details). From conservation perspective, the link between β-diversity, species abundance distribution and species trait may assist the species at risk of local extinction (extermination) due to human disturbance.

Secondly, the existing ecological theories and models also tend to treat humans as a causal factor similar to any other animal in an ecosystem. Humans, however, are social, cultural and psychological beings (Kueffer et al. 2015), which, therefore, demands an integration of theories and models from different fields of social science (e.g. anthropology, psychology, economics) to describe what drives human behaviour in governing their relationship among themselves and their surrounding environment (Bennett et al. 2017; Spalding et al. 2017). Indeed, theoretical ecology typically presents the issue of culture and nature as socio-ecological systems by focusing on the distribution of species diversity, energy, and material flow (Figure 1.1).

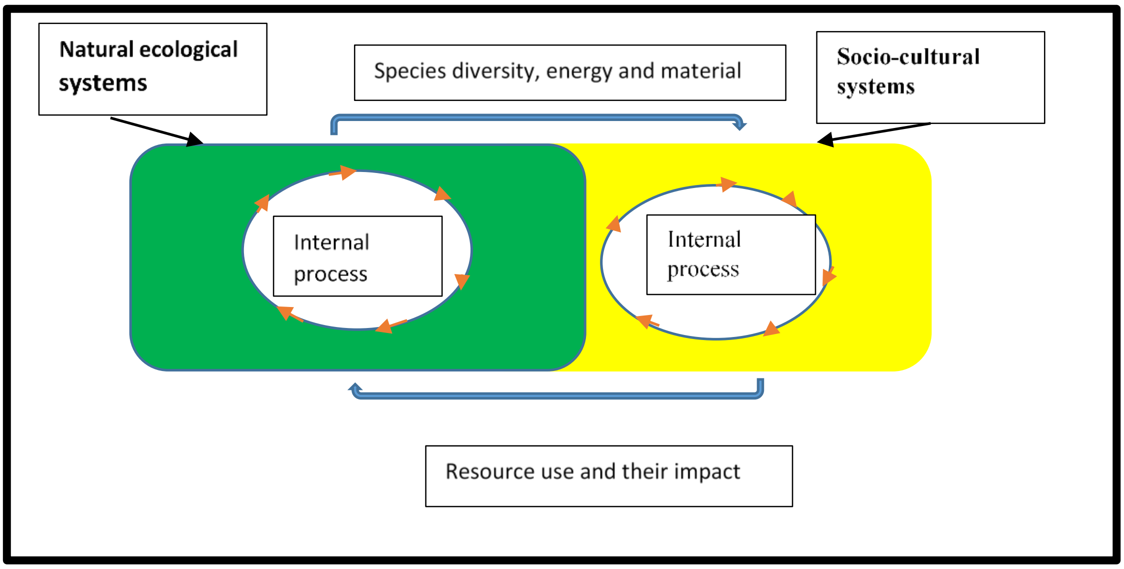

Figure 1.1. The general structure of frameworks used in mainstream ecology to analyze the interaction between natural ecological and social systems (Modified from Muhar et al. 2017, Kueffer et al. 2015).

In recent years, there has been a growing interest to the link the complex internal ecological process of a complex interaction between the biotic and abiotic components of nature that underpin the creation and maintenance of biodiversity, energy and material flows to the benefit of society (ecosystem services). To that effect, a plethora of assessment tools (e.g the Millennium Ecosystem Assessment) have been developed to classify and quantify the various ecosystem services (Daily and Matson 2008). Many ecologists also demonstrated the negative impact of resource use on the local and global biodiversity and their consequences to ecosystem services using existing ecological hypothesis and theories (e.g., IDH, Ecological appearance hypothesis). However, the fact that it still relies on ecological theories developed for pristine nature, without explicitly conceptualizing human behaviour, makes it partial (Muhar et al. 2017; Kueffer et al. 2015). For instance, the effectiveness of the ecological appearance hypothesis to predict human use-behaviour of natural ecosystems has been as controversial as the IDH which is discussed in the above (See Chapter 4 and 5 for the details).

The ecological appearance hypothesis suggests that humans depend highly on the most abundant species in their landscape for various utilities (Phillips and Gentry 1993). This implies human disturbance is ecologically driven in the same way with the original intention of the hypothesis when it was developed to test the link between are visible and abundant plants and animal herbivory (De Albuquerque et al. 2005). However, human being is a socio-culture being. There are suggestion that shows the prevalence of cultural important species that shapes in a significant way the cultural identity of a particular cultural group (Culturally keystone species) (Gaoue et al. 2017) regardless of their state of ecological abundance. Hence, recent studies have been challenging whether the ecological appearance hypothesis can sufficiently explain the resource use-behaviour of human beings in comparison with cultural norms including TSEK systems (Soares et al. 2016). This is crucial as both biodiversity loss, their conservation and restoration are largely influenced by resource use-behaviour of human beings. Hence, there is a need to explore alternative models that bring ecological and socio-cultural systems together.

In recognition of the inextricable link between nature and socio-culture systems, in particular the issue of cross-scale interaction, feedback, and reciprocity of both systems, there have been many advances in socio-ecological models and frameworks (Pulver et al. 2018; Muhar et al. 2017; Ban et al. 2013). These frameworks integrate various ecological and social theories (Holzer et al. 2018; Muhar et al. 2017). Such integration is believed to play a crucial role in systemic conservation planning by providing a thorough understanding of nature-culture interactions and areas of social consideration (e.g. compromise and trade-off) (Ban et al. 2013). Figure 1. 2 shows the general framework that integrates ecological concepts and socio-culture concepts into socio-ecological systems and their effect on human behaviour (Muhar et al. 2017). This framework embraces not only the application of ecological theories to link nature and ecosystem services but also address the situational link (e.g. status of rural livelihood) with socio-culture concepts and nature (e.g. ecological abundance) which ultimately influence human behaviour. The socio-culture concepts refer to the social organisation (e.g. individual, culture groups), Culture (e.g., values, norms, TSEK), and resource use governance (e.g. designation of different tree based tree land use regimes, traditional rules, state rules).

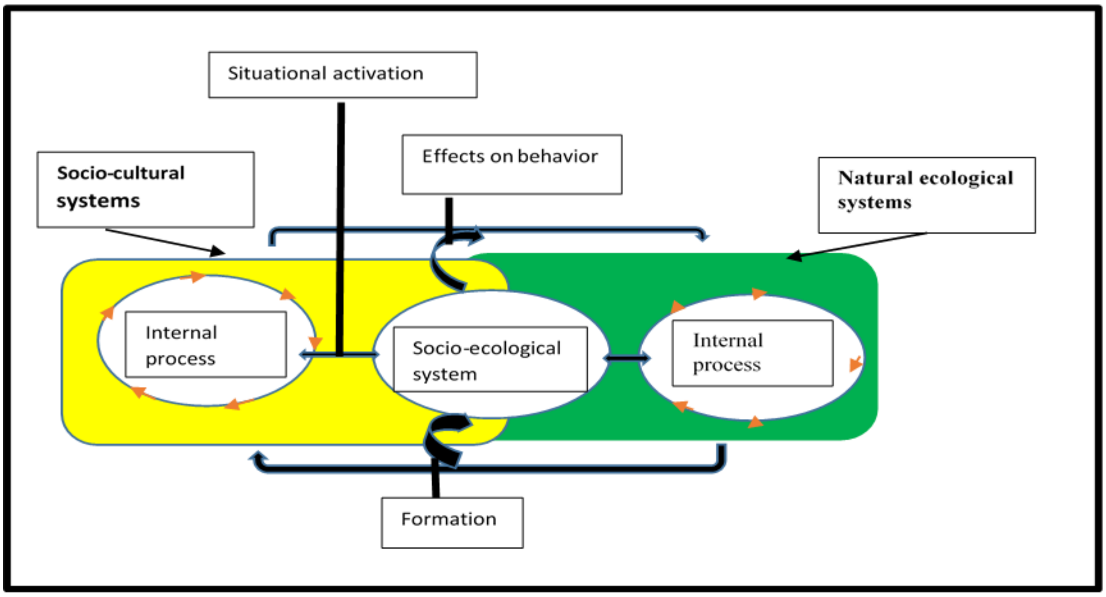

Figure 1.2. General model to integrate socio-cultural concepts into socio-ecological systems (Muhar et al. 2017).

Since no two landscapes are identical (Sharma et al. 2016), framing biodiversity issues as a socio-ecological challenge implicitly or explicitly implies that the choice and effectiveness of conservation measures are context-dependent. The response of local species diversity as measured by different matrices (e.g. richness, beta-diversity, and abundance distribution of wide geographic range and endemic species) to land use disturbance depends on the moderating effect of the forest landscape (Tscharntke et al. 2012). Moreover, land use decisions and dependence on species diversity for rural livelihoods in multifunctional landscapes are non-random activities. The disturbance intensity, ranging from open access areas to cultural protected sacred forests, are governed by culture (e.g. local perceived values, social norms and taboos, experiential knowledge), and perceived legitimacy of local institutions. Culture also influences pro-environmental behaviour (e.g. compliance to scared forest and tree species vis-a-vie state-protected indigenous forests) (Araia and Chirwa 2019). This incurs an additional dimension to the scope, complexity, and uncertainty in managing global biodiversity both within and beyond protected areas (Heywood et al. 2018) at the local level. For instance, which component of the socio-ecological system plays a predominant role in forest and tree species use and conservation? Is that the ecological abundance in nature or culture of the society residing in forest biodiversity hotspots that plays a major role in use-behaviour?

The above realization has led to some scientists calling for a need to explore ways to resuscitate and integrate traditional socio-ecological knowledge into science-based knowledge systems (Poe et al. 2014). Indeed, recent theoretical and empirical studies have shown that there are still pockets of areas, both in developing and developed countries, with rich socio-ecological knowledge. Studying traditional socio-ecological knowledge can provide insight into how to develop adaptive and mitigation strategies to global environmental challenges (Gómez-Baggethun 2014). Equally, local people have been voluntarily setting aside part of the landscape for protection of cultural and symbolic significant areas such as sacred forests, species, and ecosystem related taboos. In many places where such practices still exist, non-compliance has not been a significant challenge (Ruiz-Mallén and Corbenra 2013, Ormsby and Bhagwat 2010, Colding and Folke 1997).

The central question is that if local people have positive compliance behaviour, why is it then that non-compliance is becoming ubiquitous in most protected areas around the world (Arias 2015; Solomon et al. 2015)? Theoretically, human value has been defined as a desirable trans-situational goal varying in importance, which serves as a guiding principle in the life of a person or social group (Ives and Kendal 2014, Stern 2000). Value is believed to influence conservation attitude and behaviour of individuals and groups towards their surrounding ecosystems. To that effect, many studies have been grouping individuals into two value orientation categories: anthropocentric and biocentric (Stern 2000). The Anthropocentric value orientation has been related to (i) the egoistic value (i.e. those who focus on self-interest); and (ii) social altruistic value (i.e. those who concerns on the consequence of their action to the welfare of the others). Biocentric value orientation has been widely attached to individuals whom they are believed to be deeply concerned on the welfare of the biodiversity and surrounding ecosystem. Can those value orientations sufficiently explain the preferential compliance behaviour toward the cultural protected areas and state protected forest areas? If not, what are the most likely factors that influence local people to comply to traditional rules that governs sacred forest; but not to state conservation rules? Do individuals, as individuals and as a member of a cultural group, value and conserve forest and tree species diversity intentionally in support of the underlying value orientation of either HE or NCS in their day to day cultural life? The response to these questions may provide a clue on alternative policy options for biodiversity conservation at different spatial scales.

However, the persistence of culture, which is embedded in traditional socio-ecological knowledge (Aswani et al. 2018; Poe et al. 2014) and believed to play crucial role in individuals value and conservation behaviour has been threatened by the decoupling of the relationship between culture and nature. For instance, the enforcement of conservation rules that evict traditional society from their landscape (e.g. protected areas) (Lyver et al. 2019; Lyver and Tylianakis 2017). Hence, understanding and evaluating the socio-ecological systems in a traditional society, that still contain rich socio-ecological knowledge and rich biodiversity, will assist in finding ways to reconcile the HE and NCS options. Such evaluation should be done in comparison with strictly protected areas (e.g., state-protected indigenous forests) (Watson et al. 2016).

1.4 Aim and objectives of the study

The purpose of this research was to evaluate the potential ecological and socio-cultural values of human modified landscapes in comparison to the effectiveness of state-protected indigenous forests. Based on the results of this evaluation, this study provided a socio-ecological conceptual framework that works both for people and for the persistence of forest and tree species diversity. The study had four objectives with the series of hypothesis for each.

Objective 1. To reveal the resilience of species diversity using landscape moderation insurance and Intermediate Disturbance Hypothesis (IDH) as general theoretical frameworks.

H1: The condition of forest landscape complexity determines the pattern of mean β – diversity and the conformity of species richness response to IDH along the land use intensity gradient of Human Modified Forest Landscapes (HMFL);

H2: The condition of forest landscape complexity determines the local contribution of the HMFL to species richness and overall β-diversity in comparison with the State-protected Indigenous Forests (SIF);

H3: The influence of land use gradient, in comparison with other environmental change drivers, to overall β – diversity of a particular forest reserve depends on the conditions of forest landscape complexity.

Objective 2: To compare the effectiveness and efficiency of a protected area and human modified landscape by disentangling the conservation value from its overall biodiversity value.

H1: There is a dissimilarity in the mean and variance β-diversity, be it in an overall local assemblage or endemic species assemblage, and among different land use regimes;

H2: If H1 is accepted, it is expected that the species abundance distribution and the dominant species identity changes along the land use intensity gradient and determine the conservation value of each land use regimes.

Objective 3. To determine whether local people consciously use traditional socio-ecological knowledge (culture) in actively modifying forest landscapes or their use-behaviour is purely driven by the abundance of the ecological resource.

H1: Homogeneity of cultural value: communities from the same cultural group, but residing in different landscape conditions, demonstrate similar use-behaviour towards similar habitats (land use intensity), as specified by cultural institutions and social norms;

H2: Use-value gradient in the multifunctional landscape: the total use-value of land use regimes in the multifunctional landscape increases with the increase of the social perceived land use gradient, both at a cultural group and household level;

H3: Ecological appearance hypothesis: local people depend highly on the most abundant species in their landscape for various utilities.

Objective 4. To understand the influence of value, experiential knowledge and perceived legitimacy of rules to non-compliance behaviour.

H1: Preferential compliance behaviour: there is a difference in the proportion of local people’s non-compliance behaviour between Culturally Protected Forest Areas (CPA) and State-protected Indigenous Forests (SIF);

H2: Compliance behaviour is positively influenced by locally perceived values, experiential knowledge and perceived legitimacy of conservation of local people.

1.5 Interdisciplinary research framework

Customarily, the Vhavenda traditional people in the study area classified tree-based traditional land use regimes in their modified landscapes based on their perceived disturbance intensity. These include Common Resource Use Zones (CRUZ) (relatively high disturbed), Trees Along Rivers and Streams (TATR) (Intermediate disturbed) and Culturally Protected (Sacred) Forest Areas (CPA). Also, there are state-protected indigenous forests that are embedded within the human modified landscapes (Araia and Chirwa 2019).

Land use disturbance intensity is suggested to be a good predictor to explore the relationship between biodiversity and culture, since it is often reciprocally affected over both space and time (Bürgi et al. 2015). Thus, whether choosing the effective management of human modified landscapes over state-protected indigenous forests for conservation of tree species diversity or not, relies on the resilience of tree species diversity (species richness, *β*-diversity, and species abundance distribution) to the different traditional tree-based land use regimes (Figure 1.3). Land use disturbance drives change in species diversity from the local to landscape scale by either directly removing the species or their habitat for human use or by changing the environmental conditions (e.g. availability of sunlight, nutrients) (Mayor et al. 2015). It also affects disturbance-recovery trajectories (i.e the direction of succession stages and time of recovery after disturbance) (Arroyo- Rodríguez et al. 2017). Hence, the resilience of species diversity to land use disturbance is non-linear and complex (Allan 2004). It depends on the concurrent moderating effect of the landscape complexity (Tscharntke et al. 2012), cultural norms and taboos (Araia and Chirwa 2019), and the effectiveness of conservation rules (e.g. compliance).

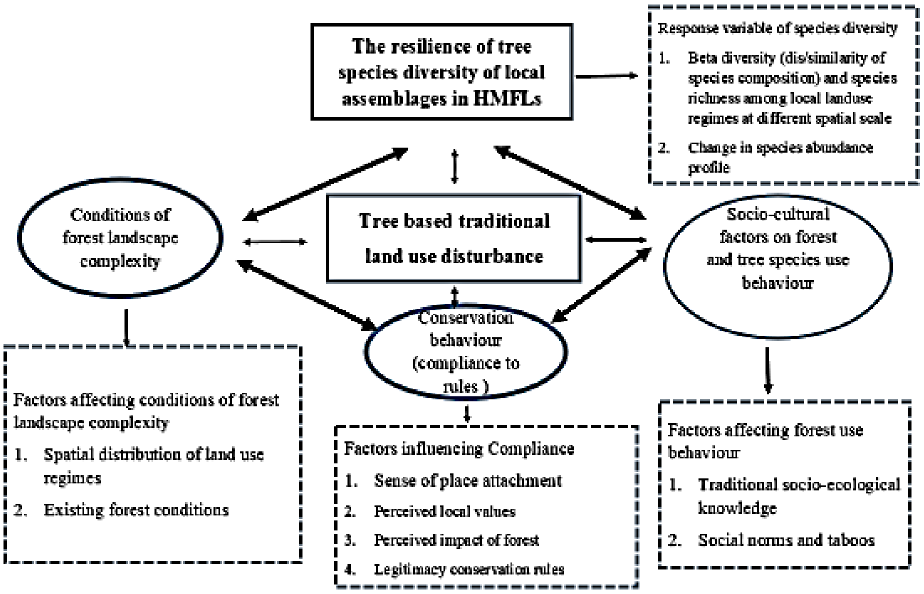

Figure 1.3. Interdisciplinary research framework.

Rectangle with the solid boundary shows the direct driver of change in species diversity /response, circle shows moderating factors to the resilience of species diversity, bi-directional arrows show reciprocal relations, uni-directional arrows direct to the dotted rectangles that contain the various variables studied for each moderator.

Local people do not superimpose different tree-based traditional land use regimes on pre-existing forest landscapes randomly. The spatial distribution of different land use disturbance intensities depends on different topographic features (e.g. altitude, slope, gradient, access) and culture (e.g. social, norms and taboos, traditional socio-ecological knowledge). Hence, no two landscapes can be identical in their forest landscape complexity. Whether the original biota or different species within a landscape remains resilient, depends on the spatial patterns of land use intensity, cross-scale interaction of disturbance regimes (Johnstone et al. 2016) and the conditions of existing forest complexity (Tscharntke et al. 2012). The condition of existing forest complexity includes the condition of forest cover, ecological connectivity (composition and configuration of the species pool) and successional stages of different patches (Arroyo- Rodríguez et al. 2017).

At the same time, in addition to the influence of culture on whether local people consciously conserve forest and tree species diversity or not, the compliance behaviour of local people to different conservation rules in use (e.g. traditional rules, state-protected indigenous forests) relies on their sense of place meaning and attachment, perceived local values of forest and tree species diversity, perceived awareness of their own impact, and the perceived legitimacy of rules. Thus, the response of species diversity and their consequence on the conservation values of different options depends on the prevailing socio-ecological context.

This complexity highlights that conservation research problems require an integrated approach, but with a wide range of socio-ecological variables and methods from the different ecological and social disciplines (Tobi and Kampen 2018; Setchell et al. 2017). To tackle such complexity, the study first identified the critical socio-ecological variables required to address each objective and hypothesis. Those variables were then pooled into an interdisciplinary research framework that was developed to guide the collection of both quantitative and qualitative data using different ecological and social science research tools. Data were analysed using a wide range of univariate and multivariate techniques.

1.6 The scope and limitation of the study

This study was conducted in two forests reserves, namely Thathe Vondo (TVFR) and Mafhela Forest reserves (MFR), which are closely located in the Eastern Soutpansberg Forest mountains in VBR. The total area of TVFR and MFR are estimated to be 1000 ha and 440 ha, respectively

(Symes et al. 2000). Almost all the residents in the two forest reserves belong to the dominant ethnic group in VBR- the Vhavenda ethnic group- but reside in different conditions. This was done to compare and infer the causal relationship among socio-ecological processes under different forest conditions. This also helps to disentangle whether culture or ecological abundance plays a predominant role in governing forest and tree species use-behaviour and compliance to state-protected indigenous forests. Hence, it does not infer how cultural diversity (a difference in culture among different communities) influences their forest and tree species use and compliance behaviour to different conservation rules.

1.7 Structure of the thesis

The thesis is divided into six chapters as follows:

Chapter 1: General introduction

Chapter 2: The contrasting effect of forest landscape condition to the resilience of species diversity in the human modified landscape: implication for the conservation of tree species

Chapter 3: Does a strictly protected area protect vulnerable local tree species better than human land use? Disentangling conservation value from biodiversity value

Chapter 4: Revealing the predominance of culture over the ecological abundance of resources in shaping local people's forest and tree species use-behaviour: The case of the Vhavenda people, South Africa

Chapter 5: Nurturing forest resources in the Vhavenda community, South Africa: factors influencing non-compliance behaviour of local people to state conservation rules

Chapter 6: Linking the ecological and socio-cultural potential of human modified forest landscapes for conservation of tree species diversity

Each chapter (Chapters 2-5) contains its own conceptual and methodological frameworks relevant to their objective and hypotheses. With regard to publications, three of the chapters (Chapters 2, 4 and 5) were submitted to different peer-reviewed journals. With those, Chapter 2, 4 and 5 are respectively, published in the following journals:

Araia, M.G. and Chirwa, P.W., 2019. Revealing the Predominance of Culture over the Ecological Abundance of Resources in Shaping Local People’s Forest and Tree Species Use Behaviour: The Case of the Vhavenda People, South Africa. *Sustainability*, *11*(11), p.3143.

Araia, M.G. and Chirwa, P.W., 2019. Nurturing forest resources in the Vhavenda community, South Africa: factors influencing non-compliance behaviour of local people to state conservation rules. *Southern Forests: a Journal of Forest Science*, 81(4), pp.357-366.

Araia, M.G.; Chirwa, P.W.; Assédé, E.S.P. Contrasting the Effect of Forest Landscape Condition to the Resilience of Species Diversity in a Human Modified Landscape: Implications for the Conservation of Tree Species. *Land* **2020**, *9*, 4.

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CHAPTER 2

Contrasting the effect of forest landscape condition to the resilience of species diversity in a human modified landscape: implications for the conservation of tree species

Abstract

Using landscape moderation insurance and Intermediate Disturbance Hypothesis (IDH) as frameworks, this study assessed the difference in local assemblage among different land use regimes (mean β–diversity), using Jaccard dissimilarity matrix in Human Modified Forest Landscapes (HMFL) at the simplified Mafhela Forest reserve and the relatively complex Thathe Vondo Forest reserve, South Africa. Then, the patterns of overall β–diversity between HMFL and State-protected Indigenous Forests (SIF) were compared. This study found that human disturbance affects mean β–diversity of local assemblages among land use regimes between the two HMFLs in an ecological contrasting manner. The HMFL in Mafhela Forest reserve had distinct local assemblages among land use regimes and did not conform to the expectation of IDH. On average, HMFL had the same average local species richness as SIF, mainly due to change in species composition (species replacement) induced by land use disturbance. Land use intensity gradient was the leading change driver to explain the overall β – diversity of the Forest reserve. The findings in the Thathe Vondo Forest reserve were in contrast with the Mafhela Forest reserve. Although HMFL had the same local species richness with SIFs, it was mainly due to a trade-off of species gain in trees along the rivers and streams and species loss in Cultural Protected Areas (sacred forests) (CPA). The contrasting findings reflecting the effectiveness of the alternative conservation strategy is context-dependent. The resilience of local assemblages and conservation value of HMFL depends on the condition of the forest complexity and cannot be captured by one theory, nor by one species diversity matrix (e.g., β -diversity or Richness). It demands the application of complementary theoretical frameworks and multilevel modeling.

Keywords: landscape moderated insurance hypothesis, land use intensity gradient, Beta (β) – diversity, intermediate disturbance hypothesis, species richness, species replacement

2.1. Introduction

It has been argued that protected areas can neither serve as a standalone strategy to protect rare or endangered species (Harvey et al. 2017) nor be effective enough to tackle the global biodiversity loss from anthropogenic disturbance (Watson et al. 2014). Also, despite an increasing hope on the conservation potential of forests and trees in modified rural landscapes, there has been a growing controversy on the richness, composition, and survival of biodiversity given persistent anthropogenic disturbances (Melo et al. 2013). Such controversy is not surprising considering that “vulnerability” (i.e sensitivity to distruabnce threat) and “representativeness” of species diversity have been the two guiding criteria for systemic conservation planning (Mittermeier et al. 2011). Hence, it is counterintuitive to presume that human modified landscapes, under different land use disturbance intensity, can be an alternative or complementary form for biodiversity conservation.

Despite the negative connotation of the literal meaning of disturbance as a disruptive force on different biological organizations (Willig et al. 2018), some have argued that the higher species diversity in the tropical forests is the result of the concurrent effect of natural and anthropogenic disturbance (Van Gemerden et al. 2003). Without disturbance, the tropical forests would have been associated with poorer species diversity (Connell 1979). Others have counterargued that the impact of anthropogenic disturbance on species diversity is idiosyncratic (Fox 2013), and affects the vulnerable tropical forest biodiversity adversely (Barlow et al. 2010).

Considering it is only a few fragments of tropical forests that have remained relatively intact (Noble and Dirzo 1997) and more than 80% of the terrestrial landmass area (Saura et al. 2018), on which approximately one-quarter of the world threatened species exist and are beyond the boundaries of global protected areas (Alroy 2017), assessing the potential biodiversity value of alternative options is crucial. In parallel, there has been growing criticism of the protected area for its inadequacy to represent and cover many valuable habitat heterogeneities and species (Joppa and Pfaff 2009, Watson et al. 2014), besides suffering from social disobedience (Lele et al. 2010). Consequently, there has been increasing research interest on the resilience of tropical forest biodiversity from human disturbance (Arroyo-Rodríguez et al. 2017). However, there is still no consensus on the resilience of species diversity and the potential of human modified landscapes for biodiversity conservation (Melo et al. 2013). This could be due to a lack of a coherent framework to assess the effect of land use disturbance on species diversity (Resasco et al. 2017) that ultimately determines the biodiversity value of a modified human landscape.

Firstly, most studies have relied on the Intermediate Disturbance Hypothesis (IDH) (Yeboah and Chen 2016) and species richness to assess the effect of land use on species diversity (Hillebrand et al. 2018). The hypothesis predicts that an intermediate disturbance intensity, frequency, or extent would maximize the species richness of an area. This is mainly by delaying the competitive exclusion or promotion of co-existence between competitively inferior and dominant species together in space and time, which cannot be sustained under too rare or severe a disturbance (Yuan et al. 2016, Connell 1979). Despite numerous research over decades to prove the application of IDH, there is no consensus on the validity of the predication (Yeboah and Chen 2016, Assede et al. 2012).

Also, species richness index, as a surrogate to compare the biodiversity values of local assemblages may not sufficiently capture compositional dissimilarity at the different spatial hierarchies that may arise either due to the separate effects of land use disturbance regimes or other co-variables in a landscape (Hillebrand et al. 2018). For instance, species richness across the land use regimes (with different disturbance intensities) can remain the same due to an equal proportion of local gain (re-colonization/immigration) and loss (local extermination) of species. The same scenario can also happen when different site conditions favour and replace an equal amount of species and disfavour others at one point in time due to differences in life history traits. In both cases, dissimilarity among sites is expected. The dissimilarity trend may either progress through the route of landscape divergence (Laurance et al. 2007) or convergence to the reference state over the recovery time (Derroire et al. 2016), depending on the resilience of forest biodiversity (Ghazoul et al. 2013). Hence, species richness alone may not fully explain the resilience of species diversity in response to disturbance.

Recently, beta (β)-diversity, the component of gamma (γ)-diversity that accumulates from dissimilarity of local assemblages (change in alpha (α) -diversity) has been suggested to be a more reliable matrix (Socolar et al. 2016). Local assemblages among land use regimes within a landscape or between landscapes can vary, either due to the difference or dissimilarity in species identity (replacement) or in species richness. Beta (β)-diversity can also be partitioned into the local contribution of a site (LCBD) (e.g., within Human Modified Forest Landscape (HMFL) *vs.* within protected areas) to compare the relative importance of different alternatives to (γ) –diversity (Legendre and Gauthier 2014).

Secondly, every landscape is a unique socio-ecological system (Sharma et al. 2016). The spatial patterns of extent, frequency, and intensity of resource extraction in a particular landscape are driven by elevation (Zhang et al. 2013), accessibility, distance from the villages, and availability of preferred species for specific uses (Shova and Hubacek 2011). This suggests that a particular forest landscape complexity emerges out of a myriad of interactions of anthropogenic and environmental change drivers (Sharma et al. 2016) that are superimposed on the original forest biota. Whether the original biota of a landscape remains resilient depends on the spatial patterns of land use intensity, cross-scale interaction of disturbance regimes (Johnstone et al. 2016) and the conditions of existing forest complexity (Tscharntke et al. 2012). The condition of existing forest complexity includes the condition of forest cover, ecological connectivity (composition and configuration of a species pool) and successional stages of different patches (Arroyo- Rodríguez et al. 2017).

According to the landscape moderated insurance hypothesis (Tscharntke et al. 2012), local assemblages in a complex landscape are expected to have better resilience and stability of ecological processes, even under a continuously changing environment. Better ecological connectivity of a complex landscape enhances the supply of propagules from the species pool and stronger neighbourhood effect (e.g. the proximity between forest patches, local competition and facilitation) among land use regimes of different intensity during disturbance-recovery dynamics. Hence, natural factors (e.g., elevation, slope gradient, position) become of ecological importance when the human influence on pre-existing forest complexity is minor or when human influence is widespread and fairly uniformly distributed across the whole landscape. The converse is true when land use disturbance overrides and simplifies the condition of the pre-existing environmental conditions (Allan 2004). In that context, the existing landscape complexity moderates the effect of land use on β –diversity (Pardini et al. 2010) at a different spatial scale (Tscharntke et al. 2012). Consequently, many studies have suggested that the potential of the human modified landscape for conservation is context-dependent (Melo et al. 2013).

The primary objective of this study was to investigate the effect of land use regimes and environmental variables in the human modified landscape on the resilience of species diversity, under different conditions of forest landscape complexity. The study was further intended to reveal the consequence of land use regimes of different intensity gradients on the cumulative contribution of the human modified landscape to the conservation of tree species, in comparison with relatively well-protected forest areas. This is crucial as the few remaining tropical forest fragments are found embedded within the same human dominated biosphere (Noble and Dirzo 1997).

Drawing upon the above arguments and scientific theories, the study hypothesized (H) the following:

H1: The condition of forest landscape complexity determines the pattern of mean β – diversity and the conformity of species richness response to IDH along the land use intensity gradient of human modified forest landscapes (HMFL);

H2: The condition of forest landscape complexity determines the local contribution of the HMFL to species richness and overall β diversity (LCBD) in comparison with the strictly protected area;

H3: The influence of land use gradient and other environmental change drivers to overall β – diversity of a particular forest reserve depends on the conditions of forest landscape complexity.

2.2. Materials and Methods

2.2.1. Study area

This study was conducted at Mafhela Forest and Thathe Vondo Reserves in the Vhembe Human Biosphere Reserve (VBR) located in Limpopo Province of South Africa. The two Forest Reserve areas (FR) belong to the eastern part of Soutpansberg Mountain Forest complex that stretches from Louis Trichardt to Thohoyandou. Thathe Vondo (TVFR) and Mafhela Forest Reserves (MFR) are located at 22^{0}52$^{'}$ S, 30 ^{0}20$^{'}$ E and 23^{0}01$^{'}$ S, 30 ^{0}30.35$^{'}$ E, respectively (Figure 2.1). Both Reserves have an altitudinal range of 700-1700 m. above sea level. The areas receive rainfall from October to March (on average-724 mm), and the average temperature ranges from 35^{o}C in summer to 18 ^{o}C in winter. Both Forest Reserves retain a few of the remaining moist forest cover in South Africa (Symes et al. 2002). Notwithstanding, local people residing within the boundaries of the Reserves still practice various tree-based traditional land use, whose potential impact for conservation is unknown.

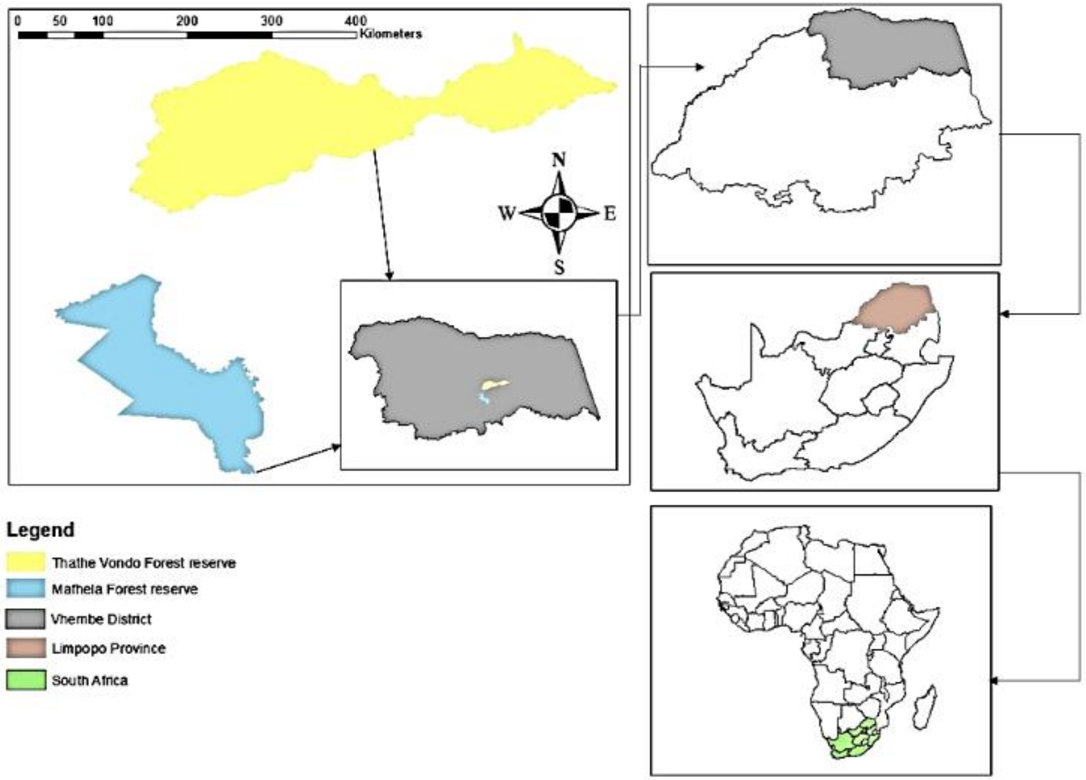

Figure 2.1. Location of the study area in the Vhembe Biosphere Reserve in Limpopo Province, South Africa

For this study, the existing land use regimes were classified, in consultation with traditional leaders, based on the configuration of trees, their typical land use, cultural practices, protection gradient, and their management system. Given the perceived disturbance gradient indicated by the traditional leaders, the existing tree-based traditional land use regimes of the two Forest Reserves were grouped into two major groups as follows:

a) *Human modified landscape (HMFL):* consisting of three tree-based traditional land use regimes under the custody of traditional authorities and the local community. These are:

- Trees Along Streams and Rivers (TATR): local community members are not allowed to harvest live trees, but occasionally access the place for livestock grazing, watering, and shading (relative intermediate disturbed);
- Common Resource Use Zones (CRUZ): this is an open access area for the harvesting of wild food, construction materials, livestock browsing and grazing, traditional medicines and others (highly disturbed);
- Culturally Protected Forest Areas (CPA): these include sacred/holy forests that are protected by royal families for cultural values and only accessible to them (minimally disturbed).

b) *State indigenous forests (SIF):* these are fragmented forest patches, with minimal to no human disturbance and legally protected by government conservation agencies.

2.2.2. Sampling design

Three different asymmetrical nested sampling designs with hierarchical factors relevant to the hypotheses and objectives of the research were adopted after Anderson et al. (2008). To analyze the first hypothesis on β – diversity along the land use gradient in HMFL of the FR and the conformity to IDH, two factors -Transect (Tr) and land use regime (La) were considered. The transect was nested under land use regimes. To determine the effect of land use on β – diversity, land use regimes were kept as a fixed variable (Figure 2.2).

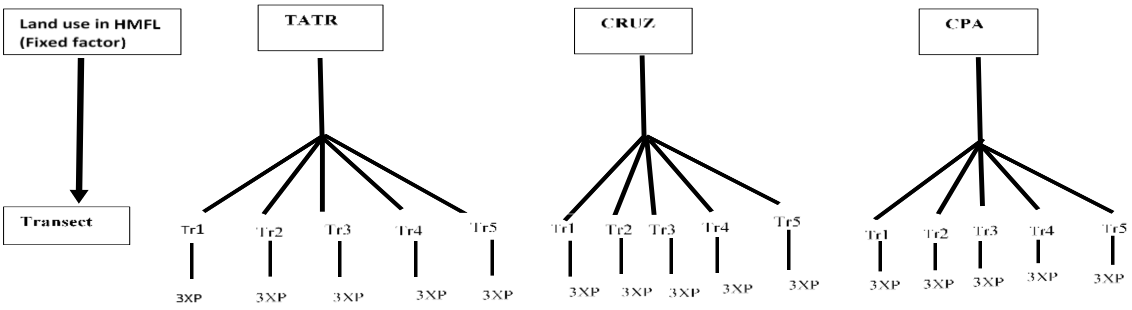

Figure 2.2. Schematic diagram of the sampling design of β – diversity in Human Modified Forest Landscape (HMFL) of the Forest Reserves (FR) and the conformity to Intermediate Disturbance Hypothesis (IDH).TATR = Trees Along Rivers and Streams, CRUZ = Common Resource Use Zone, CPA = Culturally Protected Forests, Tr =Transect, P=plot

To compare the overall species richness and the local contribution to β – diversity (LCBD) of HMFLs against SIF (HMFL *Vs.* SIF) for the second hypothesis, three factors were considered; Transect, land use, and HMFL *Vs.* SIF. Transects were nested in land use regimes. Land use regimes were nested in HMFL *Vs.* SIF, and HMFL *Vs.* SIF was kept as a fixed factor (Figure 2.3). Lastly, for the third hypothesis to explore the influence of land use regimes and environmental change drivers to overall β – diversity of a particular forest reserve, the plot was nested in Forest Reserves (FR).

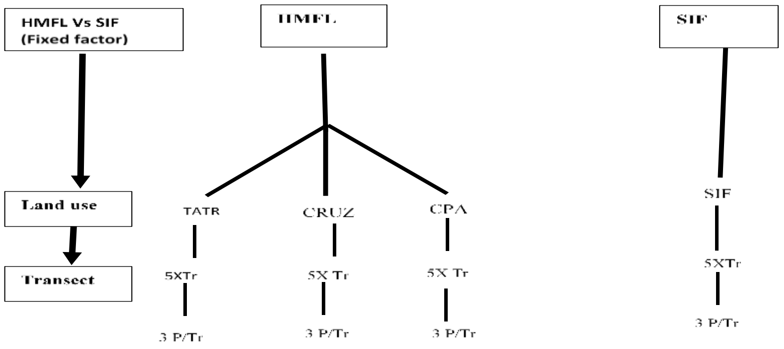

Figure 2.3. Schematic diagram of the sampling design to compare the overall species richness and the local contribution to β – diversity of Human Modified Forest Landscape (HMFLs) against SIF (HMFL Vs. SIF) for the second hypothesis.

Each FR consisted of three land use regimes of the HMFLs (CRUZ, TATR, and CPA) and SIF. Identification and location of land use regimes within the forest reserves were done with the guidance of local informants (Sheil et al. 2006; 2002). In each land use regime, five transects (Tr) were established (Annexture 2.1). However, only three transects were used for data collection in TATR in Mafhela Forest Reserve as a large part of the remaining blocks of forest patches had been cleared for horticultural production.

Previous studies in many tropical countries showed that a 40 m x 5 m transect as appropriate for tree species diversity survey in the tropics (Gillison 2006). In this study, the transect length was 50 m long. All transects were separated from each other by at least by 200 m. Unlike other similar studies (e.g. Gillison et al. 1996) that placed four (4) 5 m x 5 m square shaped sample plots in each transect, this study established three (3) 20 m x 10 m rectangular plots (P) that were spaced 10 m apart along a linear transect.

The modification of Gillison (2006) and Gillison et al. (1996) sampling approaches were important to strike a balance between the observer's efficiency of data collection and effectiveness of the sampling effort. Experience in different countries showed that observers fatigue increases if the transect size larger than a 40 m x 5 m is used in a complex vegetation (Gillison 2006). Gillison (2006) attributed the rough train as a likely factor for observer's fatigue, which also characterise the ragged mountain forests of this study area. Hence, the slight increase in transect length combined with the reduction in the number of sample plots per transect was intended to alleviate observers fatigue. At the same time, rectangular sample plots have been proven to capture more species than square shaped sample plots of the same size (Hairiah et al. 2001), let alone when relatively larger sample plot size was used in this study.

Thus, the overall sampling area coverage per transect combined with the slight increase in transect length was an attempt to capture a board range of local habitat heterogeneity.

2.2.3. Data collection

2.2.3.1. Measurement of tree species assemblage

In this study, all perennial woody plants with a diameter height of ≥ 2 m were considered as trees (van Wyk and van Wyk 1998). Wyk and van Wyk (1998) suggest that the artificial distinction between shrubs and trees often breaks down in practice. For instance, despite a typical tree is considered to have a single trunk, it may also have multiple stems. In this study, all trees at breast height (dbh) ≥ 2 cm and enumerated (Pinard et al. 2013). The scientific and vernacular names (from local informants) of observed tree species in each plot were recorded. In instances where tree species identification was not possible in the field, tree voucher specimens were collected and later identified at the Thohoyandou Botanical Garden and Herbarium. The study used a binomial nomenclature (Genus and species) to present the scientific name of all the trees observed, and as accepted by National Herbarium in Pretoria (van Wyk and van Wyk 1998). The authors of the tree names are cited as the species appear for the first time in the text based on the information available on South African National Biodiversity Institute (SANBI 2019) and Plants of the World online (PoWO 2019).

2.2.3.2. Change drivers of *β* – diversity

Land use regime, accessibility, distance from the villages, elevation, slope, and positions of transects within land use and landscape and their geographical locations were recorded. Land use regimes were coded SIF (4), CPA (3), TATR (2) and CRUZ (1) based on a perceived disturbance intensity in ascending order. The elevation of each transect within land use was recorded using GPS (Annexure 2.1). A slope/landscape gradient was recorded using Suunto PM-5/369 PC clinometer. These were grouped and coded as follows: 2^0-5^0 degree (gentle to undulating - coded 1), 5^0-6^0 degree (moderate - coded 2), 11^0-18^0 degree (moderately steep - coded 3), and 19^0-30^0 degree (steep - coded 4) (Young 1980). The position of each transect was recorded as the bottom, foot-slope, mid-slope, shoulder and top as the landform of the terrain was irregular in both forest areas and coded as 1, 2, 3, 4 and 5, respectively (Sheil et al. 2002). Walking distance from the village (hereafter known as distance) to sample plots, which may influence the extent of forest and tree species harvesting by the local community (Shova and Hubacek 2011), was also estimated in km.

2.2.4. Statistical analysis

The effectiveness of the sampling effort on the number of species observed for the whole study area, FRs and land use regimes was evaluated using a species accumulation curve based on Bootstrap estimators in Primer-E (Clarke et al. 2015). An effective sampling effort captures ≥80% of the estimated species richness (Foggo et al. 2013). This was then followed by comparing the condition of forest landscape complexity of the two forest reserves. First, SIMPER (Similarity Percentage) analysis of land use regimes was done on the original abundance matrix to identify dominant species of each FR (Anderson et al. 2008). SIMPER also provides an output on the contribution of a species to intra-group (within forest reserves) similarity by taking the average contribution of i^{th} species (Av. Sim), of overall pairs of sample plots within a group (j,k), of a species in the Bray-Curtis similarity formula (Equation 2.1).

$$S_{jk}(i) = 200.\min(y_{ik}) \,/\, \sum_{i=1}^{p}(y_{ij} + y_{ik}) \qquad (2.1)$$

Where $S_{jk}(i)$ represents the similarity between the J_{th} and k_{th} sample, y_{ij} represents the entry in the i^{th} row and J column of the abundance data matrix, that is the abundance for the ith species in the j^{th} sample (i=1,2…p, j=1,2…n).

To describe and compare the spatial patterns of land use disturbance, change drivers for each HMFLs were subjected to a spearman rank correlation test using draftsman plot routine; a routine that provides a Spearman correlation coefficient of pairwise variables of all combinations of variables(Clarke et al. 2015).

2.2.4.1 The effect of land use regimes on the difference of mean β – diversity in HMFL

To assess the difference in mean β – diversity between land use regimes within each forest reserve, first, the information was put into an abundance-based species-sample matrix. The original abundance-based species-sample matrix was transformed into presence/absence format followed by preparation of Jaccard similarity coefficient matrix (Clarke et al. 2015). Jaccard similarity coefficient ($S_{jk}(i)$) calculates the likelihood of a single species picked at random from two sites without considering the joint absence (Anderson et al., 2008) as follows(Equation 2.2):

$$S_{jk}(i) = 100[\frac{\mathrm{a}}{a+b+c}] \tag{2.2}$$

Where $S_{jk}(i)$ represents the similarity between the J^{th} and k^{th} samples and S is the probability (X 100). a, b, and c represent the number of species which are present in both samples, the number of species present in sample *J* but absent from sample in *K,* and the number of species present in sample *K* but absent from sample in *J*, respectively.

First, the $S_{jk}(i)$ matrix was then subjected to non-Metric Multidimensional Scaling Ordination (nNMDS) to visually assess the patterns of mean *β*-diversity of local assemblage among different land use regimes for HMFLs (Avolio et al. 2015; Clarke et al. 2015). Points in ordination plot with the same colour represent, plots within the same land use regime, while the different colours represent different land use regimes and their closeness to each other in terms of their degree of similarity of local assemblages or lower value for Jaccard dis similarity coefficient ($S_{jk}(i)$).

The $S_{jk}(i)$ matrix was subjected to Multivariate analysis of variance in PERMANOVA (permutation-based MANOVA) with 999 permutations to test if there is a statistically significant difference ($p \leq 0.05$) in mean *β*-diversity of local assemblages (group centroids) among the land use regimes. PERMANOVA is a geometric partitioning of multivariate variation in the space of a chosen dissimilarity measures according to a given ANOVA design, with P- value obtained using permutations. This was followed by a *post-hoc* pairwise comparison between land use regimes. To measure the effect size of the difference (dissimilarity in mean between pairwise land use regimes, distance from pairwise centroids (Av. D_J) was calculated using *Distance among centroids* routine. An Av. D_J takes a percentage value between (0,100), with the ends of the range representing the extreme possibility; (0) represents identical species assemblages between sample points and (100) represents distinct local assemblages between sample points(Anderson et al. 2008). Change in mean *β*–diversity can happen either due to species richness difference (local extermination/immigration), change in species composition (replacement of species identity) or both between samples points (Legendre 2014).

2.2.4.2. Species richness difference *($\Delta\overline{d}$)* along the land use gradient

To determine if there was a significant difference in species richness between land use regimes, Margalef index (d) for overall species richness was calculated using the DIVERSE function in Primer-E 7(Anderson et al. 2008). Margalef index (*d*) is an indicator for species richness/count (S) and takes into consideration the effect of size (N) for the fact that within a larger number of individuals more species are expected (Clarke et al. 2015) ((Equation 2.3)).

$$d = \frac{\mathrm{S}-1}{\log \mathrm{e}\ \mathrm{N}} \tag{2.3}$$

A resemblances matrix of *d* - sample plots were then developed using the Euclidean distance. This was analyzed using PERMANOVA ($p \leq 0.05$). When PERMANOVA is used to do a univariate ANOVA, the P-values are obtained by permutation, and therefore it avoids the assumption of normality (Anderson et al. 2008). A pairwise comparison was then used to compare the species richness difference and the conformity to IDH using PERMANOVA. This was done separately for both HMFLs.

Where significant differences were detected, this was then followed by Hedge (g) metric calculation to detect the effective size of the richness difference between pairs of land use regimes with the same HMFL. The Hedge (g) metric is a weighted average mean standard difference based on a pooled variance measure (Coetzee et al. 2014). It was calculated as follows (Equation 2.4):

$$g = \frac{X_a - X_b}{SDpooled} \tag{2.4}$$

Where X_a and X_b refer to the mean of paired samples, and SD_{pooled} refers to the pooled standard deviation. SD_{pooled} was calculated as follows (Equation 2.5):

$$\mathrm{SDpooled} = \sqrt{\frac{(n_a-1)SDa^2+(n_b-1)SDb^2}{n_a+n_b-2}} \tag{2.5}$$

Where n_a and n_b refers to the sampling size of the paired samples, SDa^2 and SDb^2 refer the square of the standard deviation of the paired samples.

Since Hedge (g) is a biased estimator of population effective size, we used the commonly used *J* correction factor to calculate the biased corrected Hedges' g value or g* =g*J* (Equation 2.6)

$$J = 1 - \frac{3}{4\,(n_a + n_b - 2) - 1} \qquad (2.6)$$

2.2.4.3. Change in species composition (identity replacement) along land use gradient

To determine if there were any significant differences of species composition along land use gradient, PERMDISP (a test of homogeneity of dispersion) procedure was employed on Jaccard similarity coefficient matrix ($p \leq 0.05$). When PERMDISP is used on the Jaccard similarity coefficient, it provides a test of significance between the sampling points on the identity of species they contain. This was then followed by pairwise comparison on species composition between land use regimes. Also, PERMDISP generates a mean square distance (hereafter referred as Mean) of a site to a group centroid (within group dissimilarity) that can directly be interpreted as the percentage of unshared species within a group when Jaccard dis/similarity matrix is used (Anderson et al. 2008).

2.2.4.4. A local contribution of the human modified forest landscape to overall β diversity of Forest reserve

To analyze the impact of human modification of forest landscape on overall structure (variability) of local assemblage, a Jaccard coefficient matrix that composed of HMFLs and SIF, was organized. The three levels, TATR, CRUZ, and CPA were nested under HMFL (Fig 6). After visually inspecting the patterns of overall mean β-diversity of local assemblage among different HMFLs and SIF using Principal Coordinate Principal Coordinated Analysis (PCA), the Jaccard coefficient matrix was subjected to a PERMANOVA test between HMFLs and SIF for each FR. To analyze the impact of human modification of forest landscape on overall mean species richness ($đ_{overall}$), a species richness (*d*) resemblance matrix of each FR that included SIF was prepared using the same procedures as above. This was then subjected to PERMANOVA test between HMFLs and SIF for each FR. In the context of this study, *overall mean species richness* ($đ_{overall}$) refer to the average species richness of plots found in a large sample. The species in each plot was first calculated by DIVERSE routine and then *overall mean species richness* was calculated by AVERAGE routine using Primer-E software (Clarke et al. 2015).

To analyze the contribution of HMFLs and SIF to overall β-diversity of the FR, the Jaccard coefficient matrix that was composed of HMFL and SIF was used. . Since PERMANOVA test showed insignificant difference in mean β-diversity, the Jaccard coefficient matrix was subjected to pairwise PERMDISP test (test of homogeneity of dispersion) between HMFLs and SIF. When PERMANOVA test result is insignificant for mean β-diversity, the PERMDISP test result shows whether the sample plots remained identical or diverge into distinct species assemblage (Avolio et al. 2015). This was then further inspected using PCO diagram. In the Principal Coordinate Analysis (PCO) diagram, the site with high LCBD is that found far from the multivariate centroid of the graphs. If a site has large LCBD, it either indicates high conservation value due to unique assemblage or conversely may indicate a degraded site with poor species assemblage that may need a restoration action (Legendre and Gauthier 2014). In the context of this study, LCBD was investigated to look at how much, on average, HMFL contributed to unique species assemblage in the FR.

2.2.5. The influence of land use and other environmental drivers in overall β – diversity of human modified Forest reserve

Based on the output of the draftsman plot routine of all change drivers, the multicollinearity test of correlation between all pairs of the change drivers was found to be below the acceptable cut-off threshold (R=0. 95). This was then followed by Distance-based linear modeling (DISTLM) and distance-based redundancy analysis *(dbRDA)* to explore the link between β -diversity with change drivers. DISTLM relies on multiple regression models that can accommodate a mixture of categorical and continuous predictors using *dbRDA*; a constrained ordination of sample sites using the same resemblance matrix on the Jaccard similarity coefficient (Anderson et al. 2008).

The relationship between each environmental variable and overall β –diversity of the FRs was initially analyzed separately (excluding other variables) in the marginal test. Variables were then subjected to a forward selection procedure (sequential test, R^2 selection criterion), in which the amount of variability explained by each variable added to the model was conditional of the variables already in the model. P-values for the marginal tests were obtained by using

999 permutations and using the Jaccard similarity coefficient matrix. Distance-Based redundancy analysis was used to visualize the results of the DISTLM (Anderson et al. 2008).

2.3 Result

2.3.1 Description of forest complexity condition

The study recorded 2125 number of individual's trees in total; out of which 957 and 1168 of trees were from MFR and TVFR, respectively. The total number of species observed in the whole study area, in MFR and in TVFR were 110, 72, and 88, respectively. The species accumulation curve based on Bootstrap technique estimated the whole study area, MFR, and TVFR to host about 125, 82, and 99 species, respectively (Figure 2.4). Hence, the sampling technique used in this study captured 88.70 % of the total species estimated for the whole study, 88.00% for MFR, and 88.18 % of TVFR.

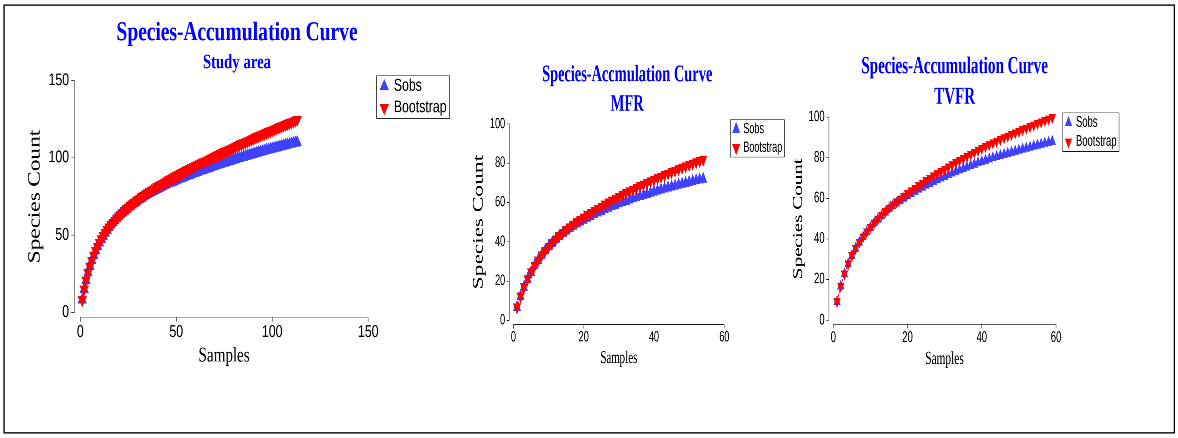

Figure 2.4. Species accumulation curve of the study area, MFR (Mafhela Forest Reserve) and TVFR (Thathe Vondo Forest Reserve) based on Bootstrap estimators.

In MFR, both the observed and estimated species number showed SIF had relatively the highest number of species, followed by CRUZ, CPA and TATR in descending (Table 2.1). In TVFR, both the observed and estimated species number showed that CRUZ had relatively the highest number of species followed by TATR, SIF, and CPA in descending order. Similar to the whole study and Forest reserves, the sampling technique used in this study also captured the majority of the species in all land use regimes in both Forest reserves. All the species in the forested landscape were indigenous species to South Africa except *Psidium guajava L.* in MFR and *Citrus limon (L.) Osbeck* in TVFR in their Common resource use zone (Annexure 2.2).

Table 2.1. Observed and estimated species number for land use regimes for MFR (Mafhela Forest reserve) and TVFR (Thathe Vondo Forest reserve).

Land use regimes	Number of species in MFR		Sampling effectiveness (%)	Number of Species in TVFR		Sampling effectiveness (%)
	S_{ob}	S_{boot}*		S_{ob}	S_{boot}*	
TATR	18	21	85.71	55	68	80.88
CRUZ	33	39	84.61	57	69	82.61
CPA	26	34	78.80	26	30	86.67
SIF	39	45	86.67	54	65	83.07

The SIMPER analysis revealed that the MFR was dominated by 7 tree species that contributed about 70% of the total abundance of tree species for the whole landscape, out of which *Englerophytum magalismontanum (Sond.) T.D.Penn., Bridelia micrantha (Hochst.) Baill.,* and *Psidium guajava* accounted for about 50 % of the total abundance of the trees. In TVFR, 12 tree species dominated and contributed about 70 % of the total abundance, out of which *Syzygium cordatum Hochst.ex C.Krauss., Xymalos monospora (Harv.) Baill., Englerophytum maglismontanum, Aphloia theiformis (Vahl) Benn, Podocarpus falcatus (Thunb.) R.Br. ex Mirb.* and *Cassine eucleiformis (Eckl. & Zeyh.) Kuntze* accounted for 50% of the total abundance of the tree of the landscape.

Table 2.2 shows that the spatial distribution of different land use regimes, based on disturbance intensity gradient, by the local people are negatively and strong correlated with the distance of the forest landscape from the village in both Forest reserves. Access, the position of the terrain and slope gradient had weak correlation with the pattern of the spatial distribution of land use gradient by the local people across both Forest reserves. However, the communities residing in both Forest reserves substantive differ in the how they distribute land use disturbance gradient against the elevation gradient despite both reside in a similar mountain range. While land use gradient had a negative and very strong correlation with elevation in MFR, this correlation was weak in TVFR.

Table 2. 2 Spearman rank correlation matrix of land use gradient and environmental change drivers

Change drivers	Thathe Vondo Forest Reserve (TVFR)					Mafhela Forest Reserve (MFR)				
	Land use	Distance	Access	Position	Gradient	Land use	Distance	Access	Position	Gradient
Distance	-0.58					-0.55				
Access	0.24	-0.17				0.24	0.64			
Position	0.30	0.48	-0.06			0.17	0.13	-0.08		
Gradient	0.19	-0.57	0.24	-0.75		0.04	-0.43	-0.12	0.66	
Elevation	-0.31	0.48	-0.47	0.09	-0.14	-0.91	0.74	0.44	-0.06	-0.33

2.3.2 The effect of land use gradient on mean β – diversity of HMFL

The visual inspection of nMDS in MFR (Figure 2.5) showed that the local assemblage of all land use regimes in HMFLs was distinct from each other. The PERMANOVA test results in Table 2.3 show that there was a significant dissimilarity in mean β – diversity among different land use regimes in MFR (F_2=7.39; P=0.001). Both pairwise comparison and distance between pairwise centroid (Av.Dj) confirmed that all land use regimes contain highly distinct local assemblage from each other (Table 2.4).

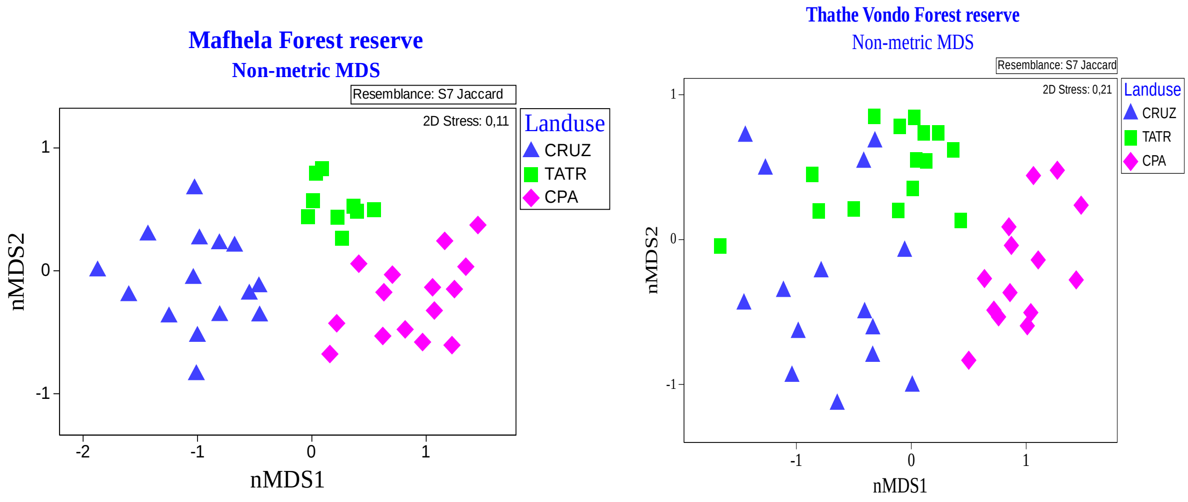

Figure 2.5. Multi-dimensional scaling of local assemblage of different land use regimes in MFR and TVFRs.

The visual inspection of NMDS in TVFR showed that the local assemblage of all land use regimes is not as distinct as in MFR. Although there was a significant dissimilarity in mean

β – diversity among land use regimes in TVFR; (F_1=3.81; P=0.001, the pairwise comparison in TVFR revealed that there was a significant difference among all pairs of land use regimes. However, the distance between pairwise centroid (Av.Dj) showed that (TATR & CRUZ) was fairly similar. Also, the local assemblages between (TATR & CPA) was at the mid-point of the similarity-dissimilarity continuum (Av.Dj=50%).

Table 2. 3 PERMANOVA results for patterns of β–diversity of HMFL along a land use gradient in human modified forest landscape (HMFL) at MFR (Mafhela Forest reserve) and TVFR (Thathe Vondo Forest reserve).

HMFLs	Source	Df	SS	MS	Pseudo-F	P(Perm)	unique terms
MFR	La	2	44,509	22,254	7.398	0.001	987
	Tr	10	30,082	3,008.20	1.537	0.001	996
	Res	26	50,865	1,956.30			
	Total	38	125,460				
TVFR	La	2	31,482	15,741	3.8127	0.001	998
	Tr	12	49,717	4,143.10	1.8422	0.001	994
	Res	29	65,222	2,249			
	Total	43	146,690				

Land use gradient (La, fixed factor, three levels) and Transect (Tr, random factor) were nested in La. Degree of Freedom (df), Sum of square (SS), F ratio (Pseudo-P), Permuted probability values (P) are shown.

Table 2. 4 Pairwise comparison of β-diversity gradient in human modified landscape (HMLF) in Mafhela and ThatheVondo forest reserves

Pairwise land use regimes	HMFLs in MFR			HMFLs in TVFR		
	T	p(Perm)	Av. D_J (%)	T	p(Perm)	Av. D_J (%)
(TATR & CRUZ)	2.53	0.016	61	1.59	0.012	38
(TATR & CPA)	2.77	0.022	55	2.22	0.008	51
(CRUZ & CPA)	2.84	0.008	61	2.05	0.006	50

TATR = Trees along rivers and streams, CRUZ = Common resource use zone, CPA = Culturally protected forests, t= pairwise statics value, p= Permuted probability values and Av. D_J = average Jaccard dissimilarity (Percentage)

2.3.3 Species richness along land use intensity gradient

The overall mean species richness ($\text{đ}_{overall}$) in Human modified forest landscape in MFR was 2.82 (SD= 0.67). The mean đ (SD) for TATR, CRUZ, and CPA were 2.78 (0.74), 3.19(0.63) and 2.49 (0.46), respectively. PERMANOVA test for species richness (*d)* showed that there was a significant difference among land use regimes (F_3=5.466, P=0.028) (Table 2.5). However, the pairwise comparison of species richness between (TATR, CRUZ) (t=1.45, p=0.26) and between (TATR, CPA) (t=1.52, p=0.09) did not detect significant differences.

The overall mean species richness of HMFL in TVFR was 3.78 (SD=1.02). The đ (SD) for TATR, CRUZ and CPA were 4.62 (0.99), 3.71(0.69) and 2.96 (0.59), respectively. PERMANOVA test for species richness difference ($\Delta\bar{d}$) found that there was a highly significant difference among land use regimes (F_3=15.454, P=0.001). The pairwise comparison of species richness detected a highly significant difference between (TATR, CRUZ) and between (TATR, CPA) with (t=3.18, p=0.03, g=-0.87) and (t=5.45, p=0.008, g= -2.57), respectively.

Table 2. 5 PERMANOVA results for richness difference between land use intensity in human modified forest landscape (HMFL) at Mafhela (MFR) and Thathe Vondo forest reserve (TVFR).

HMFLs	Source	Df	SS	MS	Psuedo-F	P (Perm)	Unique Perms
MFR	La	2	3.740	1.870	5.466	0.029	965
	Tr	10	3.421	0.342	0.970	0.529	999
	Res	26	9,715	0.374			
	Total	38	16.876				
TVFR	La	2	19.333	9.667	15.454	0.001	998
	Tr	12	7.508	0.626	1.046	0.455	999
	Res	29	17.350	0.598			
	Total	43	44.822				

Land use gradient (La, fixed factor, three levels) and Transect (Tr, random factor) were nested in La. Degree of Freedom (df), Sum of square (SS), F ratio (Pseudo-P), Permuted probability values (P) are shown.

2.3.4. Change in species composition (identity replacement) along land use gradients

In MFR, the PERMDISP test showed that there was a significant difference (F2, 36 =9.11, P=0.001) in species composition along the land use gradient. The pairwise comparison of PERMDISP result showed that there was a significant difference in the species composition between (TATR, CRUZ) and between (TATR, CPA) with (t=0.002; p=0.002) and (t=2.44; p=0.003), respectively. The local assemblage in TATR, CRUZ, and CPA had about 37.2%, 49.4%, and 44.9 % within group dissimilarity (percentage of unshared species), respectively.

In TVFR, PERMDISP test showed a significant difference in species composition among land use regimes (F2, 41 =, P=0.001). However, the pairwise PERMDISP result showed that there was a highly significant difference in species composition between (TATR, CRUZ) (t=3.03; p=0.013); but not between (TATR, CPA) (t=1.42; p=0.20). The local assemblage in TATR, CRUZ, and CPA had 49.7%, 55.6%, and 46.6% within group dissimilarity, respectively.

2.3.5. The local contribution of the human modified forest landscape to overall *β*-diversity of Forest reserve

The PERMANOVA test result showed that there was no significant difference between the mean *β*-diversity between (HMFLs, SIF) in both forest reserves; (F_1=0.68, P=0.845) in MFR and (F_1=0.26, P=0.817) in TVFR.

The overall mean species richness ($\bar{d}_{overall}$) of the whole MFR was 3.00, and the standard deviation (SD) was 0.46. The mean $\bar{d}$ (SD) for HMFLs and SIF were 2.72(0.64) and 2.82(0.67) respectively. PERMANOVA test for species richness difference ($\Delta\bar{d}$) between HMFL and SIF confirmed that there was no significant difference (F_1=1.98, P=2.86). The PERMDISP test revealed that there was a significant difference ($F_{1,\ 52}$ = 16.58, P=0.001) in within group dissimilarity in species composition between HMLF (Mean =61.80) and SIF (Mean= 54.18). The PCO confirms that most part of the HMFLs contains many sample plots that were far from the ideal local assemblage of the FR than SIF in FR. In particular, PCO1 showed that CRUZ had a remarkable distinct species assemblage, not only with SIF, but also the other land use regimes within HMFL (Figure 2.6).

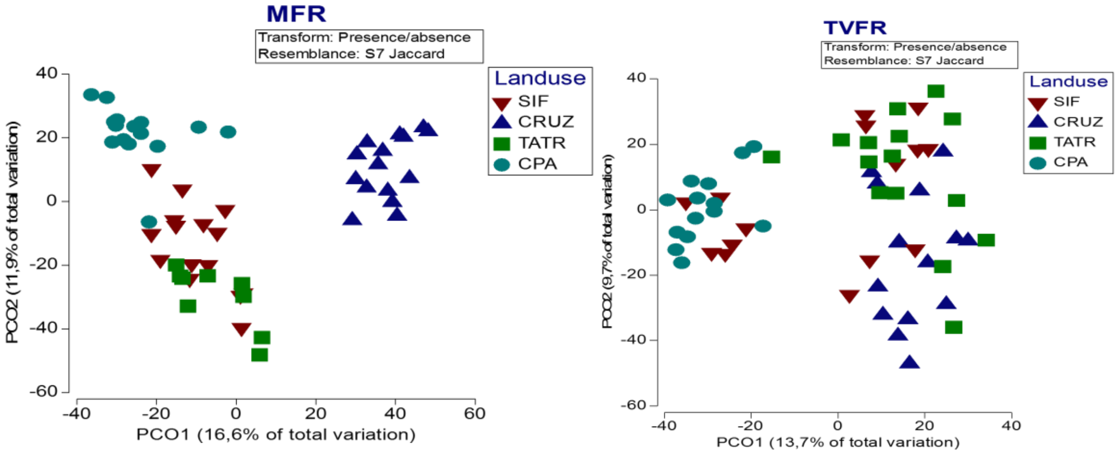

Figure 2.6. Principal Coordinate analysis (PCO) of local assemblages among different land use regimes within HMFLs and SIF

The overall mean species richness (đ$_{overall}$) of the whole TVFR was 3.75, and the standard deviation (SD) was 0.93. The mean đ (SD) for HMFLs and SIF part were 3.78(1.02) and 3.66 (0.61), respectively. PERMANOVA test for species richness difference ($\Delta\bar{d}$) between HMFL and SIF found that there was no significant difference (F_1=0.014, P=0.741) (Table 6). The PERMDISP test revealed that there was no significant difference in within group dissimilarity in species composition between HMLF (Mean =56.6%) and SIF (Mean = 50.5%) ($F_{1,\ 57}$ = 16.58, P=0.331). The PCO shows sample plots in HMFLs and SIF have similar distribution of sample plots from the ideal local assemblage of the FR (Figure 2. 6).

2.3.6 Change drivers influencing overall β-diversity of Forest reserves

The marginal test using DISTLM showed that each element of change drivers was found to be statistically significant in explaining the overall β-diversity of both forest reserves. The total sum of the individual contribution of each change driver explains about 53.56% and 38.79% of overall β-diversity in MFR and TVFR, respectively. However, the contribution and the significance of those elements in total explain 39.55% in MFR and 28.63 % in TVFR of the overall β-diversity when tested with the sequential test of DISTLM (Figure 2.7). The drop from the total marginal contribution of each change driver in the sequential test may indicate the prevalence of covariance or interaction of land use and other change drivers in shaping the overall β-diversity patterns of a landscape. In MFR, land use regimes explained the highest portion of (13.85%) of the overall β-diversity pattern. This was then followed by elevation (8.42%), position (9.17 %), and distance (8.40 %). In TVFR, elevation explained the highest variability (9.8 %) followed by a position (5.3%) and slope gradient (3.8%)

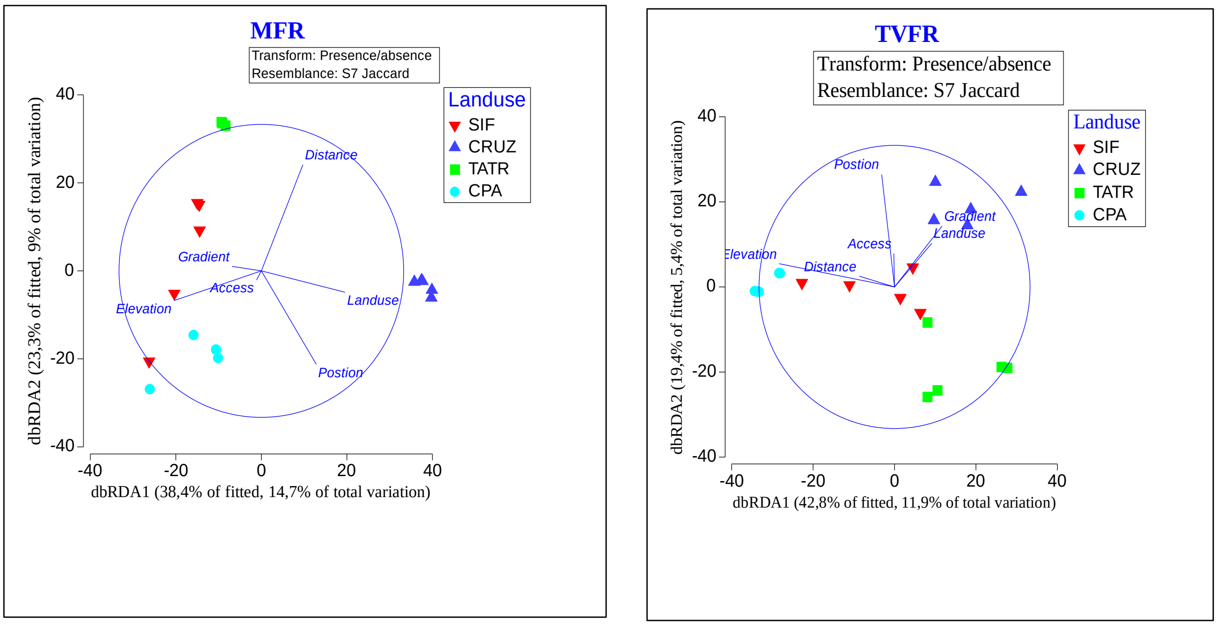

Figure 2.7. *db*RDA (Distance-based redundancy analysis) graphs on the relationship of land use regimes (based on land use intensity gradient) and other change drivers to overall β-diversity.MFR = Mafhela Forest reserve, TVFR = Thathe Vondo Forest reserve; TATR = Trees along rivers and streams, CRUZ = Common resource use zone, CPA = Culturally protected forests, SIF=State-protected indigenous forest.

2.4 Discussion

It is evident that anthropogenic disturbance alters the species diversity through a combination of human land use factors by (i) directly removing preferred tree species for livelihood (e.g., fuelwood and timber); (ii) arresting local successional recovery through the recurrent use, including grazing and herbal medicines; (iii) directly or indirectly changing the local conditions (e.g. soil, moisture, sunlight, competition) (Mayor et al. 2015) and; iv) applying different social norms to govern parts of a forest landscape (Araia and Chirwa 2019. Notwithstanding, the impacts of land use disturbance on biodiversity are neither temporary nor fully avoidable (Ellis et al. 2012). Furthermore, each landscape, on which different land use intensity is superimposed, is a unique socio-ecological system (Sharma et al. 2016) and differs in their moderating effect on the resilience of local species diversity (Tscharntke et al. 2012).

The findings from SIMPER and correlational test of land use intensity gradient with other environmental change drivers imply that Mafhela Forest Reserve (MFR) was a relatively simple forest landscape. It had lesser species diversity and dominated by few disturbance tolerant species; on which similar land use regime was spatially clustered across the same elevation range. On the contrary, TVFR was complex, species-rich and dominated by a mix of intermediate and late successional tree species. Unlike MFR, the spatial pattern of land use regimes was more heterogeneous. Hence, a clear understanding of the effect of land use under different conditions of forest landscape complexity in human modified landscapes is crucial for better conservation and management of biodiversity.

2.4.1. The effect of land use gradient on mean β – diversity of HMFL

Confirming to the hypothesis1, the two HMFLs demonstrated the contrasting effects of land use disturbance to mean β-diversity and conformity to the expectation of IDH. Similar findings of the possibility of divergence and convergence of local assemblages (e.g. Laurance et al. 2007) and conditional conformity on the response of species richness to intermediate disturbance hypothesis have been reported by many studies (e.g. Yuan et al. 2016). This can be attributed to the difference in the resilience capacity of the local assemblage of land use regimes due to the variability of landscape moderation effect on land use disturbance to species diversity at different spatial scale (Tscharntke et al. 2012).

As expected in a simple forest landscape, land use disturbance in MFR enhanced higher dissimilarity in mean β-diversity of local assemblage among all land use regimes. Regardless of the land use intensity gradient, all land use regimes had a distinct local assemblage from each other. This could be because each land use intensity has been creating an environment that suits a set of co-existing species with a similar life-history trait (habitat specialist) (Johnstone et al. 2016). This is consistent with the findings on the lack of conformity of species richness to IDH and the strong evidence of species replacement in HMFLs of MFR. The weak evidence in species richness gradient implies that species replacement played a dominant role in the patterns of *β-diversity*. Hillebrand et al. (2018) underline that when immigration and local extermination (change in species composition) become frequent in a landscape, the species richness can still recover over time to the same level despite the change of species composition.

Furthermore, the higher dissimilarity along a land use gradient in MFR also hints at the limited influence of landscape species pool and the breakdown of ecological connectivity of the landscape (ecological fragmentation) due to the homogenous spatial patterns of land use intensity across the landscape(Li et al. 2016). Ecological fragmentation hinders the recovery of species composition of a vulnerable landscape through natural succession. Conversely, it may enhance further divergence into a set of alternative stable states in the landscape (Laurance et al. 2007) or homogenised the whole landscape by few invasive species over time. In particular, the fact that an exotic species *Psidium guajava* became one of the top three dominant species in MFR hints at the fact that the fate of existing indigenous forests tree species diversity is far from certain. A study conducted in Kakamega rain forests showed that *Psidium guajava* has the capacity to change habitats, alter ecosystems functions and services, and replace indigenous forest successfully. This successful invasion by *Psidium guajava* has been attributed to its ability to produce many seeds, store seeds even in deep layers of soil profile with long viability period, prolific regeneration and establishment capacity even under harsh conditions, such as fire (Kawawa et al. 2016). Moreover, it has many long distance dispersal agents (such as mammals and birds) and the leaves' allelopathic nature that hinders the regeneration and establishment of indigenous species (Chapla and Campos 2010).

Contrastingly, in a relatively complex forest landscape of TVFR, the local assemblage of land use regimes shared the majority of species among each other. However, the similarity in local assemblage declined along the land use gradient. Such pattern hints at the higher influence of landscape species pool, ecological connectivity (de Juan et al. 2013) and the positive influence of forest landscape complexity during disturbance-recovery (Tscharntke et al. 2012). The conformity of species richness response to IDH in the presence of clear gradient in mean β-diversity hints at the fact that species replacement was practically insignificant to override an orderly local extermination/gain gradient of the TVFR. Hence, the observed β-diversity patterns in HMFLs can be explained by how disturbance affects the mechanisms of species coexistence in a relatively complex landscape.

As expected in the IDH, the intermediately disturbed TATR had a maximum species richness by delaying the competitive exclusion or promotion of co-existence between different life-history traits. The similarity of species shared by CPA and CRUZ with TATR (pairwise

D_J), indicates that TATR retained the majority of competitively inferior species of CRUZ and the competitively dominant species of CPA that could have been locally extinct under too rare or severe disturbance regimes (Yuan et al. 2016; Connell 1979). The severe decline in species richness in local assemblages between (CPA, TATR) may imply that the better traditional protection of CPA resulted in a very severe local extermination of competitively inferior species due to a competitive dominance of a few late successional species.

The findings in TVFR are not surprising considering the recent claim by Munyati and Sinthumule (2014) on the decline of deforestation rate and recovery of forest conditions in TVFR vegetation. The conformity of species richness response to IDH may hint that the local assemblage of land use regimes (along a land use intensity gradient) in HMFL in TVFR are more resilient to land use disturbance. However, resilience does not mean the absence of dynamism. Even in the absence of human disturbance, the local neighbourhood effect, together with biotic and abiotic elements, may still incur small scale changes in species composition (Ghazoul et al. 2015).

2.4.2. The local contribution of the human modified forest landscape to overall *β*-diversity of Forest reserve

Overall, the higher local contribution of HMFLs to overall *β*-diversity in MFR through different land use activities might be contributed substantively to *Ɣ*-diversity (overall biodiversity) of the landscape. The contribution might also appear as a confirmation to the recent criticism on the insufficiency of some protected areas to cover the scale of compositional dissimilarity (e.g. Joppa and Pfaff 2009, Watson et al. 2014). However, higher overall *β*-diversity does not automatically imply that human modification of forest landscape enhances the quality and amount of biodiversity (Socolar et al. 2016). For instance, the highest contribution of some of the sample plots in CRUZ to LCBD implies that the substantial proportion of the HMFL was a degraded ecosystem wherein the substantive parts of the original biota were replaced by competitively inferior and early successional species at the local level.

The fact that land use regimes followed by distance and elevation explain the most substantial proportion of overall *β*-diversity (section 3.5) highlights that the current condition of forest complexity at a landscape level is highly simplified by anthropogenic disturbance. As such, the landscape species pool may not rescue the local extermination of old-growth tree species (Pardini et al. 2010) unless restored. This may also show that, even if some seed of canopy and sub-canopy tree may arrive, the fact that recurrent land use disturbance (e.g. in CRUZ) may still delay the competitive exclusion of shrub species as expected by IDH. Instead, in line with the intermediate landscape complexity hypothesis (Tscharntke et al. 2012), the state-protected indigenous forests (SIF) appeared to be more effective in safeguarding species-rich hotspots of a vulnerable landscape. However, considering the findings of Laurance et al. (2012) on the impact of environmental deterioration outside of the majority of tropical protected areas for the ecological health of the interior part of protected areas, even the sustainability of the remaining relatively species-rich SIF is uncertain. This is mainly due to the reduction of the minimum dynamic area required for proper ecosystem functioning (Hansen and DeFries 2007) and a desperate demand for forests for rural livelihood. The shift in local species composition may have a detrimental effect on ecosystem provision to the local community.

In a contrasting ecological manner, human modification in TVFR did not adversely impact the overall landscape assemblage and had equal mean local species richness and LCBD. Following the landscape moderated insurance hypothesis (Tscharntke et al. 2012), the presence of better conditions of forest landscape complexity might have been assisting in a rapid recovery of ecological process under a continuous land use pressure (Pardini et al. 2010). The fact that natural factors, such as elevation, slope gradient and position of the terrain are leading drivers to explain the overall β-diversity shows that human modification did not override the natural gradient of γ-diversity; instead it kept the species diversity in dynamic equilibrium. Natural factors become ecologically important when the anthropogenic impact disturbance is minor or when disturbances are widespread and fairly uniformly distributed across a landscape (Allan 2004). As such, the effectiveness of SIF in TVFR is lower than its counterpart in MFR. The cumulative impact of land use on HMFL did result in comparable average local species richness and contribution to overall β-diversity with their counterpart SIF.

2.5 Conclusion

With the recent prediction of mass extinction of species and the decline of ecosystem services, the debate on whether to maximize on the potential of human modified forest landscapes as an alternative or complementary strategy to protected areas is a non-trivial issue. However, the response for the effectiveness of both conservation alternatives lies in our understanding and response to the question: under what condition does anthropogenic disturbance enhance, erode or remain harmless to the pre-existing natural forest conditions in the human biosphere? The contrasting findings between the simple and the complex human modified forest landscape reflect the contrasting insurance value of existing conditions of forest landscape complexity between the HMFLs of the two Forest reserves. They also reflect that the resilience capacity of local assemblages due to land use pressure can neither be fully explained by one theory nor captured by one species diversity matrix (e.g., Richness or *beta*-diversity). Using the species richness index alone may obscure the effect of area by averaging the local species richness without discerning the colonization and extinction dynamics. Hence, it demands the application of complementary theoretical frameworks and multilevel modeling.

The higher dissimilarity of local assemblages, in the absence of conformity of species richness response to IDH, of a simplified MFR may imply that the replacement of original forest biota in HMFLs is a function of the local extermination of intermediate and late successional species in the whole forest reserve. Conversely, the higher species richness of state-protected indigenous forests imply that strictly protected areas can be an effective conservation tool to protect biodiversity hotspots in a simplified forest landscape. It can serve as refugia and source of propagules for the recover of the local lost species if complemented with the restoration efforts of the overall forest landscape complexity. Moreover, reconfiguring the spatial patterns of land use regimes across the HMFLs, to increase landscape connectivity may also play a crucial role in restoration. The restoration efforts must consider councious effort to control further encroachment of the landscape by *Psidium guajava*. On the contrary, ensuring collaborative and holistic landscape management in TVFR, using IDH as a guiding tool, may ensure the sustainability of the current forest landscape complexity and the retention of the rich species diversity. However, a conclusive remarks can not made based on *beta*-diversity, that are only exclsuively dependent on the presence/ absence data. It does need further research on abundance based data to explictly expose which species traits and their relative abundance of species mostly affected by distrubance (Chapter 3). This will assist to effeciently allocat the increasingly limited conservation resources for conservation priority species and habitates.

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CHAPTER 3

Do strictly protected areas protect vulnerable local tree species better than human land use? Disentangling conservation value from biodiversity value

Abstract

The debate on the implication of the growing scarcity of global resources for biodiversity conservation is escalating. Some conservationists argue that effective management of human modified ecosystems is a preferable strategy than protecting undisturbed forest ecosystems. However, there has not been sufficient empirical evidence on the efficiency of alternative strategies using a comparable ecological performance matrix. Hence, this study evaluate the efficiency of the protected area for the conservation of rare species. By linking the response of different facets of beta (*β*)–diversity and change in species abundance distribution, this study disentangled the conservation value of alternative options for rare species, such as canopy and sub-canopy trees of wide geographic range and endemic species, from the overall biodiversity value. The study was conducted in Mafhela Forest reserve (MFR) and Thathe Vondo Forest reserve (TVFR), South Africa. The result showed that state-protected indigenous forests (SIF) in both Forest reserves supported the highest overall tree species diversity. However, the two SIF of the forest reserves demonstrated different pattern in their efficiency in protecting rare species. In MFR, all land use regimes had a significant difference in mean *β*–diversity, variance *β*–diversity and land use contribution to overall *β*–diversity. Overall, all land use regimes in MFR lost species, became more uneven, and the dominance of trees declined and replaced by shrubs along the land use gradient in comparison with SIF. In contrast, TVFR showed different pattern of response in species diversity to disturbance. Unlike MFR, SIF in TVFR was not exceptionally efficient in comparison with the other traditional land use except with intensively disturbed common resource use zones. The contrast reflects the difference in the condition of the forest landscape complexity of the two reserves. In conclusion, allocating the limited resources to reinforce SIF in MFR, accompanied with restoration of the whole forest landscape, would be a preferred biodiversity conservation strategy. On the contrary, investing the limited resources in empowering local people in TVFR to sustain their cultural practices in managing the whole landscape would be preferable.

Keywords: Conservation value; biodiversity value; rare species; beta (*β*)-diversity; species abundance distribution

3.1. Introduction

The original intention of having protected areas was to preserve places of particular ecological importance (Watson et al. 2016). Nonetheless, most recent studies have focussed on assessing the effectiveness of a global investment of protected areas in delivering ecological, social, and economic benefits (Castro et al. 2015). Like many integrated conservation and development programmes with a multiplicity of purpose (Robinson et al. 2004), this shift in focus rendered protected areas to be "a jack of all trade and a master of none." As such, there has been a growing criticism on the effectiveness of protected areas, as a conservation strategy, in sustaining biodiversity and ecosystem services. At the same time, there has been an emergence of a new conservation paradigm that de-emphasizes the need for protection of biological hotspots based on the presumption of the remarkable resilience of nature to human influence (Doak et al. 2015). Consequently, some conservationists argue that effective management of human modified ecosystems (e.g. traditional land use and resource management) is more important for biodiversity conservation than protecting undisturbed forest ecosystems (Watson et al. 2018).

The above polarised opinions emanate from the conundrum of deciding on which area and species of a landscape to protect; commonly known as "the agony of choice" in conservation (Collen 2015). Amid this conundrum, many studies have been signaling that species richness after human disturbance increases in some places (Zulu et al. 2018; Syampungani 2008) and, decreasing or remaining the same in other places (Hillebrand et al. 2018; Vellend et al. 2017). Through time, the distribution and trend of local species richness after anthropogenic disturbance may recover fully (Vellend et al. 2017). However, neither the occurrence of high local richness nor the recovery of richness after disturbance of a particular site is self-sufficient to guide conservation decision (Hillebrand et al. 2018) and priority setting under the growing limitation of global conservation funding (Walls 2018).

Protecting or restoring species-rich areas may suggest efficient use of a limited conservation resource, thereby implying that a large number of species can be retained in small areas (Lamoreux et al. 2006). However, species richness does not necessarily correlate with areas of the exceptional abundance of rare species (Marchese et al. 2015; Pimm et al. 2014). Tree species can be rare either due to its low population density as in most of the old growth forests in the tropics (Barlow et al. 2010) or narrow geographic range (endemicity) or both.

These put rare species at higher risk of extinction (Pimm et al. 2014). This risk can also be aggravated by land use disturbance that directly removes rare species and creates favourable condition for their replacement by wide geographical range species (Waltert et al. 2011). The replacement may offset the loss of local species richness but reduces the spatial heterogeneity (Hillebrand et al. 2018).

The equality of local species richness may obscure the extinction debt (i.e. the risk of local extermination of species that can occur gradually following habitat loss and fragmentation) (Kuussaari et al. 2009). An extinction debt manifests when the likelihood of an equal number of species richness in different land use regimes arises in comparison with their adjacent protected areas, even though the abundance of rare species are affected negatively (Hillebrand et al. 2018). Some species of the original forest biota can remain as singletons or doubletons that cannot persist as a viable population in isolation (Barlow et al. 2010). Moreover, the positive relationship between the extent of areas and species richness is considered as one of universal law in ecology (Lawton 1999). Yet, due to a competing land use demand and limited conservation resources, protected areas are expected to represent the biological organization (species, population, community, and ecosystems) of a landscape as efficiently as possible (McIntosh et al. 2017) in a relatively small area. Hence, the risk of providing conservation credit where it is not due (Watson et al. 2016) may become high when comparison is made between the effectiveness of relatively small protected area with larger area of unprotected landscape.

Watson et al. (2016) suggest that the potential of alternative conservation strategies should be assessed using comparable ecological performance matrix. It must reflect both the difference in species richness and the relative species abundance between sampling points (Hillebrand et al. 2018). The conceptual and methodological advance in beta (β) -diversity (compositional dissimilarity between sampling points within a landscape) has made a significant contribution in unifying the concept of species richness and replacement of species identity among sampling points (Legendre 2014). However, similar to species richness studies, the reliance of most beta (β)-diversity studies on species presence/absence data cannot provide a comprehensive picture of the global biodiversity crisis. For instance, one may observe all the same species at two sites, but those two species may have different abundance rank in ascending order in the magnitude species abundance - the common species in one site may be rare in the other and the vice versa. Even when the species may have the same rank, they may still differ in their evenness (Barwell et al. 2015). Evenness refers to the equitability in relative species abundance in a sampling unit.

Recently, Avolio et al. (2015) suggested a framework that links β-diversity and species abundance distribution, as a tool to monitor the impact of global environmental change on biodiversity. The authors argue that disturbance can cause a change in overall β-diversity of a landscape either by changing (i) the mean dissimilarity of local assemblages along land use intensity gradient (Mean β-diversity) or (ii) the homogeneity of within-group variability of local assemblages among different disturbance regimes (Variance β-diversity) or (iii) simultaneously. Recently, many authors have been suggesting the use of β-diversity for different conservation applications. For instance, pairwise β-diversity matrix has been used to discern the impact of disturbance between land use regimes while the overall β-diversity matrix has also been used to scale up the local impact of land use disturbance to higher spatial hierarchical scale (e.g., human modified landscape) (McGill et al. 2015). Also, Anderson et al. (2008) demonstrated the use of mean and variance β-diversity in human dominated ecosystems as a sign of stress or instability of biological communities due to disturbance. However, unlike β-diversity, the response of local assemblage in abundance distribution has proven to be elusive.

Against the intuitive correlation of high disturbance intensity with the decline of species evenness and species richness, many studies have been showing that the response of species abundance to disturbance intensity depends on the ecological traits of species (Simons et al. 2015). However, many of trait-based studies have been conducted without reference to any theory or predictable mechanisms on the response of local assemblage to disturbance (Pulsford et al. 2016).

Recently, there has been a suggestion that intermediate disturbance hypothesis (IDH) can provide a conceptual framework in understanding the response of tree species of original biota to disturbance gradient. IDH predicates a mixture of different growth forms (shrubs, sub-canopy, and canopy trees) would co-exist in an intermediately disturbed local assemblage. Whereas disturbance tolerant shrubs dominate areas of high disturbance intensity, the dominance of canopy trees prevails in low disturbance intensity areas (Pulsford et al. 2016). This implies that understanding the ecological traits of the species that contributed most to pairwise β-diversity may provide a better picture of the quality of biodiversity that can persist under different disturbance regimes. Such understanding helps to set conservation priorities

(McGill et al. 2015) for the use of limited conservation resources efficiently. Thus, there is a need to concurrently determine the effect of tree-based traditional land use regimes on β-diversity, species abundance distribution, and the dominant species traits in response to disturbance intensity gradient. The principal aim of this study was, therefore, to compare the efficiency of the protected area and different land use regimes in human modified forest landscape for the conservation of vulnerable tree species diversity. The comparison was made by disentangling the conservation value of land use regimes for retaining priority species that are sensitive to human disturbance (such as rare old-growth forest and endemic tree species) from their overall biodiversity value.

Drawing upon the above arguments and scientific theories, the study hypothesized (H) the following:

H1: There is a dissimilarity in the mean and variance β-diversity, be it in overall local assemblage or endemic species assemblage, and among different land use regimes.

H2: If H1 is accepted, it is expected that the species abundance distribution and the dominant species identity changes along the land use intensity gradient and determine the conservation value of each land use regimes.

3.2. Materials and Methods

3.2.1. Study area

Similar to the resilience of species diversity, this study was conducted in Mafhela Forest and Thathe Vondo Reserves, Vhembe Biosphere Reserve (VBR) located in Limpopo Province, South Africa. As indicated in section 2.1 (Chapter 2), land use regimes were classified Common Resource Use Zones (CRUZ) (relatively high disturbed), Trees along rivers and streams (TATR) (Intermediate disturbed) and Cultural protected (Sacred) forests (CPA). Also, there are state-protected indigenous forests that are embedded within the human modified landscapes (Araia and Chirwa 2019).

3.2.2. Sampling design

Nested sampling design with hierarchical factors was adopted after Anderson et al. (2008). Three factors Plot (P), Transect (Tr), and land use gradient (La) were considered in data collection. The plot was nested under Transect and Transect was nested under land use regimes. To determine the effect of land use on β – diversity and species abundance distribution, land use gradient was kept as a fixed variable. Each Forest Reserves (FR) consisted of three land use regimes of the HMFLs (CRUZ, TATR, and CPA) and a state-protected indigenous (SIF). Identification, location of land use gradient and sampling layout was done in the same way with Chapter two.

3.2.3. Data collection

Data was collected with the same procedure discussed in Chapter 2.2.3. In the context of this study, tree species that are exclusively found within the Southern Africa biogeographical region were considered as endemic species. Those that are also found beyond this region were as wide geographical region species (van Wyk and van Wyk 1998).

3.2.4. Statistical Analysis

The effectiveness of the sampling effort on species observed (S_{ob}), for the whole study area and each reserve, was evaluated using species accumulation curve based on Bootstrap estimators in Primer-E (Clarke et al. 2015). The sampling effort was 88.70 % of the whole study and about 88.00% for each FR. Sampling effort that captures ≥80% of the estimated species richness can be considered effective (e.g., Foggo et al. 2003) (See Table 2.1).

3.2.4.1 Analysis of β-diversity based on relative species abundance of overall local assemblages

To analyze β-diversity of overall local assemblage or all species that co-exist in a specific place, the original species abundance matrix was subjected to Hellinger distance matrix analysis

(Legendre 2014). Hellinger distance refers to a Euclidian distance on the square root of relative species abundance-sample data where the relative species abundance calculated by dividing species abundance by the sum of the total individuals of a sample (Clarke and Gorley 2015). The Hellinger distance matrix was then subjected to non-Metric Multidimensional Scaling Ordination (nNMDS) to visually assess the mean and variance β-diversity of local assemblage among different land use regimes for each Forest reserve (Avolio et al. 2015; Clarke et al. 2015).

Figure 3.1 demonstrates the possible multidimensional scaling of on the response of local assemblage to a hypothetical experiment. The white triangles represents local assemblage of state-protected indigenous forests (SIF) as Control, and black circles represents tree based traditional land use regimes in HMFL as a treatment. The possible alternative outcome are (A) No change in mean or variance of replicates both among and within treatments. (B) No change in mean but increased variance with no pattern (B1) or with several distinct new local assemblages (B2). (C) No change in mean but reduced variance. (D) Change in mean and no change in variance. (E) Change in mean and increased variance, with no distinct pattern (E1) or several new community states (E2). (F) Change in mean and reduced variance (adopted and modified from Avolio et al. 2015).

Avolio et al. (2015) contend that if disturbance does not change the mean and variance β-diversity, it does not have an effect on species abundance distribution. Whereas, if no change in mean β-diversity is accompanied by change in variance β-diversity, then it implies that the species abundance distribution of a treatment sample is affected by either gain/loss rare species or the change of evenness of species abundance. In contrast, if change in mean β-diversity by distrubance is accompanied by no change in variance β-diversity, it implies that the consequence of disturbance on species abundance distribution of a treatment sample occurred mainly due to the uniformity of disturbance impact within treatment (e.g. among transects within a specific land use regime) . However, if disturbance changes the mean and variance β-diversity of a local assmebldge, then it implies that the species abundance distribution is altered due to disturbance on effect on species loss/gain.

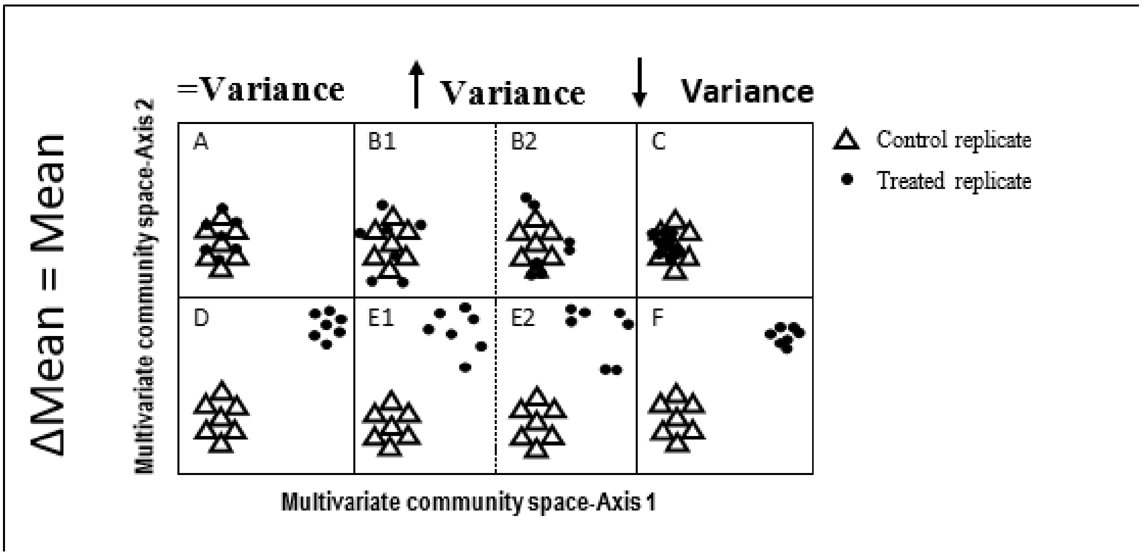

Figure 3.1. Possible ways of dissimilarity of local assemblages (replicates) among and within land use disturbance regimes (mean in rows, variance in columns).

3.2.4.2 Mean β-diversity among land use regimes

In order to test the significant difference ($p \leq 0.05$) in mean β-diversity among land use regimes, the Hellinger distance matrix was subjected to Multivariate analysis of Variance in PERMANOVA (permutation-based MANOVA). PERMANOVA is a direct multivariate analog to the usual ANOVA with testing via 999 permutations to accommodate the frequent violation of the assumption in ecological community data. Pairwise comparison between SIF and the other land use regimes were then performed to detect the significance of the change in mean dissimilarity (Pair-wise mean β-diversity) along a land use gradient (Anderson et al. 2008). To measure the effect size of the difference (dissimilarity in mean between pairwise land use regimes, Hellinger distance from pairwise centroids was calculated using Distance among centroids routine. Unlike distance generated by any similarity matrix that takes a value between (0,1), Hellinger distance takes value (0, $\sqrt{2}$); (0) represents identical local assemblages in terms of the relative species abundance between sample points and ($\sqrt{2}$) represents distinct local assemblages between sample points(Conde and Dominguez 2018, Legendre and Gallagher 2001). Hence, the Hellinger distance from pairwise centroids was then scaled to be the value of (0, 1) by dividing the results with ($\sqrt{2}$) (Legendre and Gallagher 2001).

3.2.4.3 Variance *β*-diversity among land use regimes

To test for a significant difference in variance *β*-diversity among land use regimes, PERMDISP (a test of homogeneity of dispersion) was employed via 999 permutations. A pairwise comparison was then used to show if there was a change in variance *β*-diversity along the land use gradient. PERMDISP also generates a mean square distance of sampling unit to group centroid (Anderson et al. 2008) that can be interpreted as within group variance in relative species abundance.

3.2.4.4. The effect of *β*-diversity on species abundance distribution of overall local assemblage.

To determine the significance of the change in species abundance distribution, K-dominance curve was used to visually inspect the averaged species abundance distribution of land use regimes. The K-dominance curve are cumulative ranked abundance curves plotted against species rank in the order of species abundance. The steepness (% dominance) represented in the Y-axis shows the dominance of species abundance, and the length of graphs in the X-axis shows species richness and rarity. Dominance refers to inverse of evenness. In the absence of baseline data on species gain/loss, the species gain/loss of each land use regimes was bench marked against the SIF (the control treatment).

Analysis of similarity percentage (SIMPER) of FR (La) was used to identify the dominant species of each land use regimes (Cut-off 70%) and, to determine the species that contributed most to mean *β*-diversity (Cut-off 50%) (Clarke et al. 2015). The SIMPER analysis helped to discern the effect of land use disturbance on the growth form of species (shrubs, sub-canopy, and canopy trees) and the geographic range of the species.

3.2.4.5. Analysis of *β*-diversity for the local assemblage of endemic species

First, endemic species abundance- sample matrix was developed from the original abundance-sample matrix. Hellinger distance matrix was then produced. This was followed by developing an aggregated abundance matrix of all wide geographic range species to use them as a co-variable. This was important, considering many studies have been indicating that disturbance aggravates the replacement of endemic species by a wide range and common species (Waltert et al. 2011). The Hellinger distance matrix was then subjected to PERMANOVA ($p \leq 0.05$) with an abundance of wide range species as a co-variable. The co-variable was then eliminated due to lack of statistical significance (Annexure 3.1). Since the Hellinger distance matrix did not perform well due to the total absence of rare species in many sample plots, the original endemic species abundance- sample matrix was then subjected to Bray-Curtis similarity coefficient analysis with dummy variable. Hence, it is only the main effect of land use on the Bray-Curtis similarity coefficient and the pairwise comparison that was reported in this study. PERMDISP was then used to analyse the variance β-diversity. For change in species abundance distribution, the study followed the same procedures used for overall local assemblage (See section 2.4.3.above). The Bray-Curtis similarity coefficient between two samples are calculated as follows (Equation 3.1):

$$S_{jk} = 100 \frac{\sum_{i=1}^{p} 2min(y_{ij,}y_{ik})}{\sum_{ij}^{p}(y_{ij}+y_{ik})} \qquad (3.1)$$

Where y_{ij} represents the entry in the i_{th} row and j_{th} column of the data matrix, that is the abundance of i_{th} species in the j_{th} sample (i=1,2….., p, j=1,2,….,n). Similarly, y_{ik} represents the count for the i_{th} species in the Kth sample and min $(y_{ij,}y_{ik})$ represents the minimum of the two counts. The dummy variable was crucial to overcome the challenge of the absences of species in some plots. S_{jk} takes the value of 100 when two samples are identical, and it takes 0 when the two samples are completely distinct. Shade plot was used to visually display the average abundance (4th root transformed) of endemic species of each land use regime and the pattern β-diversity in their respective reserve.

3.3. Results

The study recorded 2125 individual trees. Wide geographic range species were the dominant individuals in both Forest reserves. In MFR, the study encountered 957 trees out of which 869 individuals were wide range trees. Overall, the study encountered 110 tree species; out of which 31 of them were endemic to Southern Africa. MFR had actual species richness (Overall=72; endemic=20). In MFR, SIF had the highest overall species richness (39) and endemic species richness (11).

In contrast, in TVFR, the study encountered 809 trees out of the total 1168 trees encountered in the study. TVFR had relatively higher species richness (Overall= 88; endemic species=25) than MFR. The SIF in TVFR had remarkable high species richness than its counterpart in MFR. Despite the fact that both CRUZ and TATR in TVFR had almost as equal species richness as the adjacent SIF, SIF still had the highest endemic tree species richness of all land use regimes (Table 3.1).

Table 3. 1 Tree species richness of land use regime in Mafhela (MFR) and Thathe Vondo (TVFR) Forest reserves

Land use regimes	Species richness in MFR		Species richness in TVFR	
	Overall	Endemic	Overall	Endemic
SIF	39	11	54	19
CRUZ	33	7	57	13
TATR	18	5	55	13
CPA	26	8	26	8

3.3.1. Analysis of β-diversity of overall local assemblages

The visual inspection of NMDS in MFR (Figure 3.1) showed that the local assemblage of all land use regimes was distinct from each other. The PERMANOVA test result (Table 3.2) confirmed that there was a significant difference in the mean β-diversity among land use regimes (F_3=6.37; p=0.001). The pairwise comparison also showed that the pairwise mean β-diversity difference was consistent between SIF and each of the other land use regimes in HMFL (Table 3.3). The Hellinger distance between SIF and CRUZ, TATR, and CPA were 0.62, 0. 58, and 0.58, respectively. Also, the PERMDISP test result showed a significant difference in variance β-diversity among land use regimes ($F_{2, 36}$ =9.11, p=0.001). Except between (CRUZ and SIF), the pairwise analysis further confirmed the significant difference in mean variance β-diversity between (SIF and TATR) and between (SIF and CPA). The Hellinger distance between SIF and CRUZ, TATR, and CPA were 0.53, 0. 53, 0.44 and 0.39, respectively.

In contrast to MFR, the visual inspection of NMDS in TVFR (Figure 3.2.) showed that all land use regimes share almost similar local assemblages with SIF, although CRUZ appeared to be slightly distinct from SIF. The PERMANOVA test (Table 3. 2) confirmed a significant difference in mean β-diversity among land use regimes in TVFR (F_3=2.44; P=0.001). However, the pairwise comparison showed weak evidence of a difference in mean β-diversity in TVFR than in MFR. The Hellinger distance between SIF and CRUZ, TATR, and CPA were 0.38, 0. 40, and 0.40, respectively. The significant difference in pairwise mean β-diversity was only observed between (SIF and CRUZ). The PERMDISP test revealed that the prevalence of a significant difference in variance β-diversity among land use regimes to β-diversity ($F_{3, 55}$ =6.95, p=0.003). Table 3.3 shows that there was a significant difference in pairwise test between (SIF and TATR) and (SIF and CPA); but not between (SIF and CRUZ). The Hellinger distance to group centroid of SIF, CRUZ, TATR and CPA were 0.59, 0.59, 0.52 and 0.52 Hellinger distance to group centroid (within group variance), respectively.

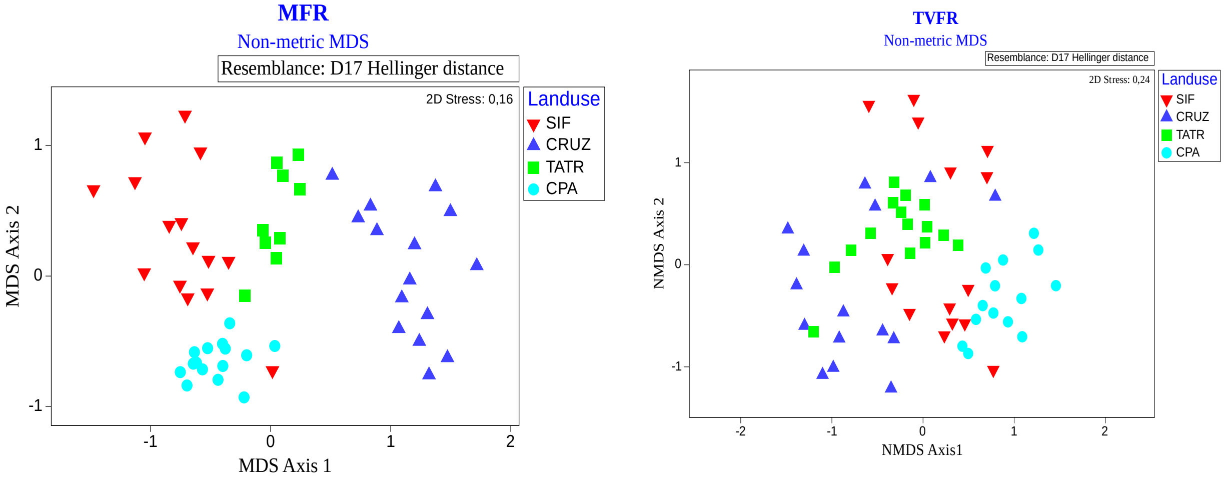

Figure 3.2. Non matric multi-dimensional scaling (NMDS) of Mafhela Forest reserve (MFR) and Thathe Vondo Forest reserve (TVFR).SIF= State-protected indigenous forests, CRUZ=Common resource use zone, TATR=Trees along rivers and streams, and CPA= culturally protected forests.

Table 3. 2 PERMANOVA results for the similarity of local assemblages among land use gradients regimes in Mafhela Forest reserve and Thathe Vondo Forest reserve; denoted as MFR and TVFR respectively

Forest reserve	Source	Df	SS	MS	Pseudo-F	p(Perm)	unique terms
MFR	La	3	16.12	5.37	6.37	0.001	998
	Tr	14	11.81	0.84	2.14	0.001	997
	Res	36	14.15	0.39			
	Total	53	42.09				
TVFR	La	3	8.06	2.68	2.44	0.001	998
	Tr	16	17.62	1.10	2.15	0.001	997
	Res	39	19.96	0.51			
	Total	58	45.67				

df=Degree of Freedom, SS=Sum of square, Pseudo-P =F ratio (Permuted), Permuted (p) = probability values (P)

Table 3. 3 Pairwise comparison of mean dissimilarity and variance of local assemblage along land use gradient of Mafhela Forest reserve (MFR) and Thathe Vondo Forest reserve (TVFR).

Pairwise land use regimes	MFR				TVFR			
	Mean β		Variance β		Mean β		Variance β	
	T	P	T	P	T	P	T	P
(SIF & CRUZ)	2.20	0.009	0.34	0.75	1.30	0.04	0.59	0.59
(SIF&TATR)	2.06	0.016	3.98	0.003	1.25	0.091	3.24	0.006
(SIF& CPA)	2.52	0.013	5.32	0.001	1.11	0.24	3.56	0.002

SIF= State-protected indigenous forests, CRUZ=Common resource use zone, TATR=Trees along rivers and streams, and CPA= culturally protected forests, t= pairwise statics value, and p= Permuted probability values

3.3.2. Change in species abundance distribution of overall assemblage.

The dominance plot (Figure 3.3) showed that the SIF had higher species diversity, evenness, and richness in both Forest reserves. However, the response of species abundance distribution to different land use regimes showed markedly contrasting patterns between MFR and TVFR. In MFR, species richness declined substantially in all land use regimes in comparison with their adjacent SIF. However, the decline in evenness of species diversity was slight in CRUZ and TATR but substantial in CPA. In contrast, in TVFR, all land use regimes had almost equal species richness with SIF except for a very substantive decline of species richness in CPA. However, almost all land use regimes retained similar evenness of species abundance with SIF.

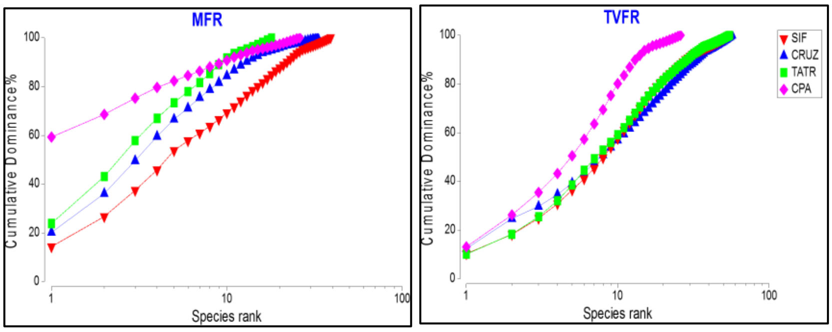

Figure 3.3. K-dominance plot of MFR (Mafhela Forest reserve) and TVFR (Thathe Vondo Forest reserve). SIF=State-protected indigenous forests, CRUZ=Common resource use zone, TATR=Trees along rivers and streams, and CPA=culturally protected area

3.3.2.1. Species contribution to pairwise β-diversity

All dominant species in all land use regimes in MFR were wide range indigenous species except *Psidium guajava* (exotic species) in CRUZ. The SIMPER analysis did not only find the relatively highest number of canopy and sub-canopy tree species in SIF but also they were dominant. These include *Aphloia theiformis*, *Schefflera umbellifera (Sond.) Baill.*, *Syzygium gerrardii*, *Nuxia floribunda Benth.*, and *Englerophytum maglismontanum*. However, as expected by IDH, the dominance of canopy and sub-canopy trees of SIF declined in abundance along the land use intensity gradient. All of the dominant species of SIF were locally extinct and replaced by shrub species in CRUZ; mainly *Psidium guava, Parinari curatellifolia Planch. ex Benth.*, and *Annona senegalensis Pers.*. However, there were still some relatively rare canopy trees of SIF in TATR (*Bridelia micrantha* and *Ficus capensis Thunb*) and CPA (*Englerophytum maglismontanum* and *Podocarpus falcatus (Thunb.) R.Br. ex Mirb.*). Those species were more abundant in TATR and CPA than in SIF. All species that contributed the most to pairwise mean β-diversity between SIF and the other land use regimes were wide range indigenous species except for *Psidium guajava* (exotic species) (Table 3.4).

Table 3. 4 SIMPER analysis of MFR (Mafhela Forest Reserves): Av. Abund (Average abundance) of discriminating species between the SIF (State-protected indigenous forests) and other land use regimes in Human modified forest landscape

Average dissimilarity between (SIF & CRUZ)= 97.98 %				Average dissimilarity between (SIF&TATR) = 87.54%				Average dissimilarity between (SIF & CPA) = 87.78%			
	Av. Abund		Cum. %		Av. Abund		Cum. %		Av. Abund		Cum. %
Species	**SIF**	**CRUZ**		Species	**SIF**	**TATR**		Species	**SIF**	**CPA**	
Psidium guajava^{S}	0	3.93	10.58	*Ficus capensis*^{CA}	0.27	2.9	10.29	*Englerophytum maglismontanum*^{SC}	1.4	9	26.54
Parinari curatellifolia^{S}	0.07	5	21.10	*Aphloia theiformis*^{SC}	2.4	0	19.42	*Aphloia theiformis*^{SC}	2.4	0	34.82
Annona senegalensis^{S}	0	3.33	29.89	*Bridelia micrantha*^{CA}	0.4	2.3	27.57	*Schefflera umbelifera*^{CA}	2.07	0	41.62
Aphloia Theiformis^{SC}	2.4	0	35.74	*Syzygium gerrardii*^{CA}	1.8	0	35.09	*Syzygium gerrardii*^{CA}	1.8	0	48.38
Dichrostachys cinerea (L.) Wight & Arn.^{S}	0.07	2.47	41.23	*Schefflera umbelifera*^{CA}	2.07	0	42.56	*Podocarpus falcatus*^{CA}	0.2	1	53.57

Schefflera umbelifera^{CA}	2.07	0	46.07	*Syzygium cordatum*^{CA}	0.53	1.8	49.94
Syzygium gerrardii^{CA}	1.8	0	50.76				

SIF= State-protected indigenous forests, CRUZ=Common resource use zone, TATR=Trees along rivers and streams, and CPA= culturally protected forests, Av. Abun= Average abundance per plot, and Com (%) = cumulative contribution (Percentage).

In TVFR, the dominant tree species of each land use regime were a mixture of wide range and endemic species. The dominant species in SIF of TVFR were *Syzygium gerrardii*, *Xymalos monospora*, *Eugenia natalitia Sond.*, *Englerophytum maglismontanum*, *Cassine euceiformis*, *Podocarpus falcatus, Psychotria capensis (Eckl.) Vatke,* and *Aphloia theiformis*. Against the expectation of IDH, there was not a clear gradient of change in the dominance of growth forms along with land use intensity. Except for the decline in abundance of dominant species of SIF in CRUZ, most of the dominant species were more abundant in TATR and CPA than in SIF. The species that contributed the most to pairwise β-diversity between SIF and the other land use regimes were a mixture of canopy and sub-canopy trees of different geographic range except in CRUZ. In CRUZ, *Parinari curatellifolia* shrub was the leading dominant species. In contrast, SIF did not endow *Parinari curatellifolia* (Table 3.5)

Table 3. 5 SIMPER analysis of TVFR (Thathe Vondo Forest Reserves): Av. Abund (Average abundance) of discriminating species between the SIF (State-protected indigenous forests) and other land use regimes in Human modified forest landscape

Mean β between (SIF & CRUZ) = 97.98 %				Mean β between (SIF & TATR)= 87.54%				Mean β between (SIF &CPA)= 87.78%			
Species	Av. Abund		Cum.	Species	Av. Abund		Cum.		Av. Abund		Cum.
	SIF	**CRUZ**	%		**SIF**	**TATR**	%	Species	**SIF**	**CPA**	%
Parinari curatellifolia$^{(S)}$	0	2.33	6.58	*Aphloia theiformis*$^{(SC)}$	0.87	2.53	6.77	*Syzygium gerrardii*$^{(CA)}$	2.07	1.93	8.73
Syzygium gerrardii^{CA}	2.07	0.47	12.74	*Cassine euceiformis*$^{E(SC)}$	1.6	1.53	12.97	*Cassine euceiformis*$^{E(SC)}$	1.6	0.86	15.65
Mimusops obovata$^{E(CA)}$	0.2	2.33	18.88	*Syzygium gerrardii*$^{(CA)}$	2.07	0.2	18.68	*Xymalos monospora*$^{(CA)}$	1.13	1.93	21.96
Cassine euceiformis$^{E(SC)}$	1.6	0.07	23.46	*Schefflera umbelifera*$^{(CA)}$	0.4	2.07	23.79	*Podocarpus falcatus*$^{(CA)}$	1.33	1.14	28.25
Eugenia natalitia$^{E(SC)}$	1.2	0.8	27.84	*Syzygium cordatum*$^{(CA)}$	0.2	1.8	28.32	*Ochna holstii*$^{(SC)}$	0.6	1.36	34.28
Podocarpus falcatus $^{(CA)}$	1.33	0.33	31.72	*Englerophytum maglismontanum*$^{(SC)}$	0.73	1.67	36.76	*Eugenia natalitia*$^{E(SC)}$	1.2	0.07	38.44

Olea capensis$^{(CA)}$	0.67	0.87	35.35	*Eugenia natalitia*$^{E(SC)}$	1.2	1.2	40.70	*Aphloia theiformis*$^{(SC)}$	0.87	0.5	42.44
Combretum erythrophyllum$^{E(SC)}$	0.53	0.93	38.85	*Combretum erythrophyllum*$^{E(CA)}$	0.53	1.6	44.21	*Olea capensis*$^{(CA)}$	0.67	1.07	46.37
Xymalos monospora$^{(CA)}$	1.13	0.07	42.14	*Podocarpus falcatus*$^{(CA)}$	1.33	0.2	47.71	*Rothmannia capensis*$^{E(SC)}$	0.27	1	50.18
Englerophytum maglismontanum$^{(SC)}$	0.73	0.93	45.25	*Xymalos monospora*^{CA}	1.13	0.8	50.70				
Acacia ataxacantha$^{(S)}$	0.93	0.55	48.17								
Brachylaena discolour$^{E(S)}$	0.87	2.61	51,07								

N.B. the superscript "*e*"= endemic species, S=Shrub, SC= sub-canopy tree, and CA=canopy tree. SIF= State-protected indigenous forests, CRUZ=Common resource use zone, TATR=Trees along rivers and streams, and CPA= culturally protected forests, Av. Abun= Average abundance per plot, and Com (%)= cumulative contribution (Percentage).

3.3.3. Analysis of *β*-diversity for the local assemblage of endemic species.

PERMANOVA test result for MFR (Table 3.6) found a significant difference in the mean *β*-diversity among land use regimes in their local assemblage of endemic tree species (F_3=1.76; p=0.037). However, the pairwise comparison found that the significant difference in pairwise mean *β*-diversity lies only between (SIF and CRUZ) (Table 3.7). The PERMDISP test result did find a significant difference in variance *β*-diversity among all land use regimes ($F_{3,\ 50}$ =6.022, p=0.012). Except between (CPA and SIF), the pairwise analysis further confirmed the significant difference in mean variance *β*-diversity between (SIF and CRUZ) and between (SIF and TATR). The Bray-Curtis distance to group centroid for SIF, CRUZ, TATR and CPA were 40.77%, 29.92%, 22.30% and 37.30%, respectively.

The PERMANOVA test revealed a significant difference in mean *β*-diversity among different regimes in TVFR (F_3=2.34; P=0.005) (Table 6). Except between (SIF and CPA), the pairwise comparison did not found that the significant difference in mean *β*-diversity between (SIF and CRUZ) and (SIF and TATR). PERMDISP test did not found a significant difference in variance *β*-diversity among all land use regimes ($F_{3,\ 46}$ =2.36, P=0.096). The Bray-Curtis distance to group centroid for SIF, CRUZ, TATR and CPA were 49.29%, 43.33%, 40.29% and 43.54 % respectively.

Table 3. 6 PERMANOVA results for patterns of β – diversity of endemic species assemblage among land use gradients Mafhela and Thathe Vondo forest reserves; donated as MFR and TVFR, respectively

Forest reserve	Source	Df	SS	MS	Pseudo-F	P(Perm)	unique terms
MFR	La	3	9820.4	3273.5	1.76	0.037	999
	Tr	14	25,925	1851.9	1.50	0.004	999
	Res	20	44,232	1228.7			
	Total	37	79,997				
TVFR	La	3	25,652	8550.7	2.34	0.005	997
	Tr	15	58,553	3659.6	2.30	0.001	999
	Res	31	61,992	1589.5			
	Total	49	146160				

df=Degree of Freedom, SS=Sum of square, Pseudo-P =F ratio (Permuted), Permuted (p) = probability values (P)

Table 3. 7 PERMANOVA results of β – diversity of endemic species among land use gradients of Mafhela and Thathe Vondo forest reserves; donated as MFR and TVFR respectively.

	Mean β		Variance β		Mean β		Variance β	
	T	P	T	P	T	p	T	P
(CRUZ & SIF)	1.6	0.011	2.56	0.045	1.26	0.19	1.80	0.12
(TATR & SIF)	1.39	0.09	3.86	0.008	0.86	0.54	2.88	0.009
(CPA & SIF)	1.08	0.13	1.02	0.374	1.15	0.21	2.05	0.78

SIF= State-protected indigenous forests, CRUZ=Common resource use zone, TATR=Trees along rivers and streams, and CPA= culturally protected forests, t= pairwise statics value, and p= Permuted probability values

3.3.3.1. Change in species abundance distribution of endemic species

The dominance plot (Figure 3.4) showed that the SIF had highest species evenness, and richness in both Forest reserves. However, the response of species abundance distribution to different land use regimes showed contrasting patterns. In MFR, the evenness of species abundance substantially declined in CPA. The decline in evenness of species abundance distribution in CRUZ and TATR was not as severe in CPA. On the contrary, while CRUZ and CPA lost a few rare species, the loss of species in TATR was substantial. In contrast, there was noticeable decline in evenness of species abundance distribution in all land use regimes. In terms of actual species richness, there was substantive loss of species both in CRUZ and TATR, except in CPA.

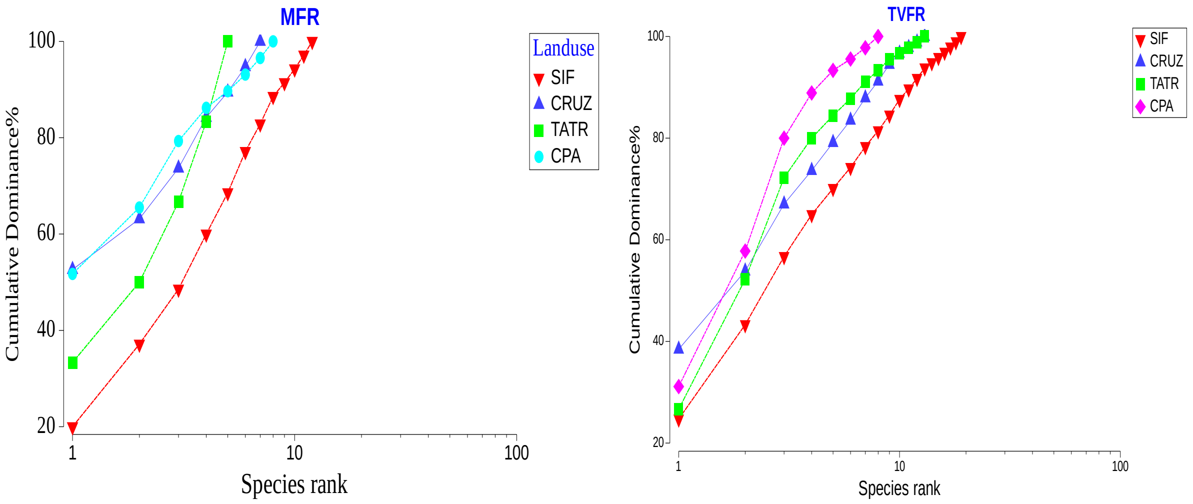

Figure 3.4. K-dominance plot of MFR (Mafhela Forest reserve) and TVFR (Thathe Vondo Forest reserve).SIF= State-protected indigenous forests, CRUZ=Common resource use zone, TATR=Trees along rivers and streams, and CPA= culturally protected forests.

The shade plot for MFR (Figure 3.5.) showed that one canopy tree (*Combretum erythrophyllum (Burch.) Sond.*) and two Sub-canopy trees (*Ochna arborea Burch.ex DC.* and *Bersama tysoniana Oliv.*) were the dominant species in SIF. As expected by IDH, the dominant SIF species were replaced by shrubs in CRUZ. *Brachylaena discolor DC.* shrub was a dominant species in CRUZ and contributed most to the mean β – diversity between (CRUZ and SIF). Despite the disappearance of most of the canopy and sub-canopy trees of SIF in TATR, TATR had retained some of the canopy trees in SIF. The canopy tree, *Rhus chirindensis Baker f.*, was not only the most dominant species in TATR but contributed most to the mean β – diversity between (TATR and SIF). In CPA, although it retained most of the canopy trees of the SIF, the increase of *Combretum erythrophyllum (Burch.) Sond.* and *Mimusops obovata Sond.* Alongside with the disappearance of *Ochna arborea* in CPA contributed most to the mean β – diversity between (CPA and SIF).

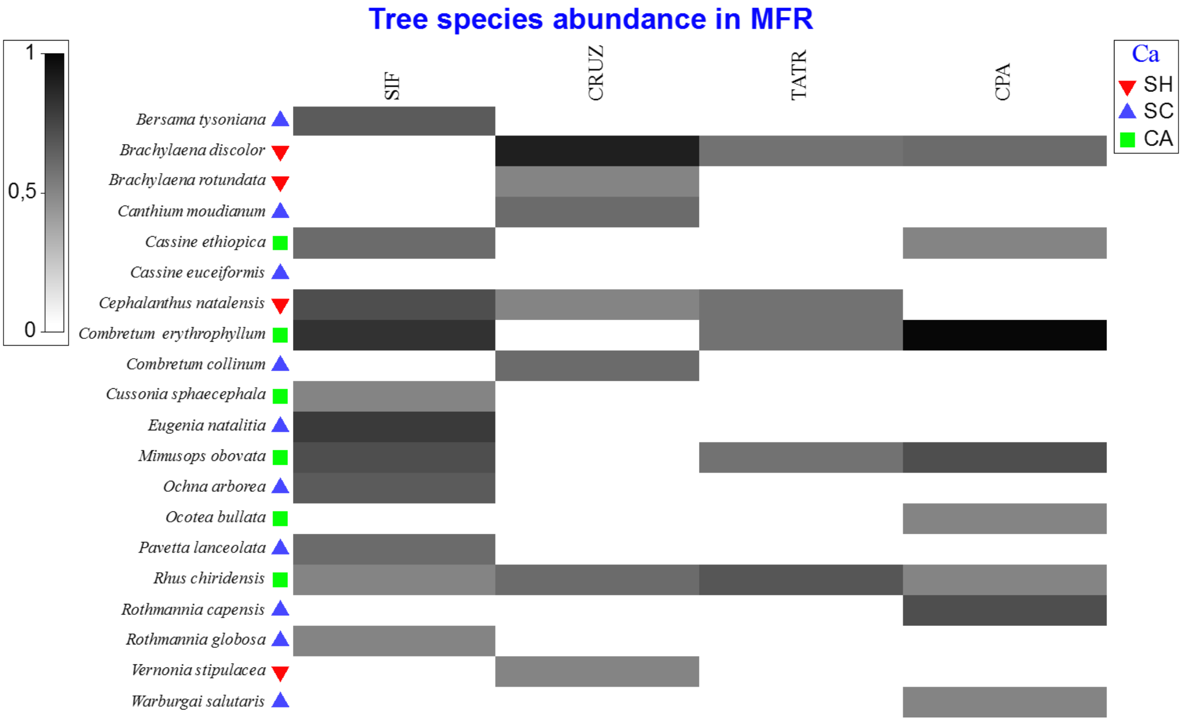

Figure 3.5. Shade plot of average endemic tree species abundance (4th transformed) in MFR(Mafhela Forest Reserves).N.B. SIF= State-protected indigenous forests, CRUZ=Common resource use zone, TATR=Trees along rivers and streams, and CPA= culturally protected forests. N.B. White in the shade plot represents absences and the depth of gray is proptional with abundance

In the shade plot for TVFR (Figure 3.6), *Combretum erythrophyllum*, *Cassine euceiformis* and *Eugenia natalitia* were observed to be the dominant species in SIF. Against the prediction of IDH, despite the decline of *Eugenia natalitia* and *Cassine euceiformis* in CRUZ, some canopy trees (*Mimusops obovata* and *Combretum erythrophyllum*) were still retained in relatively higher abundance. Hence, the species contributed most to the pairwise mean β-diversity was mainly change in abundance of canopy species by another canopy species. A similar pattern of replacing dominant canopy species of SIF by another canopy species was observed in TATR and CPA.

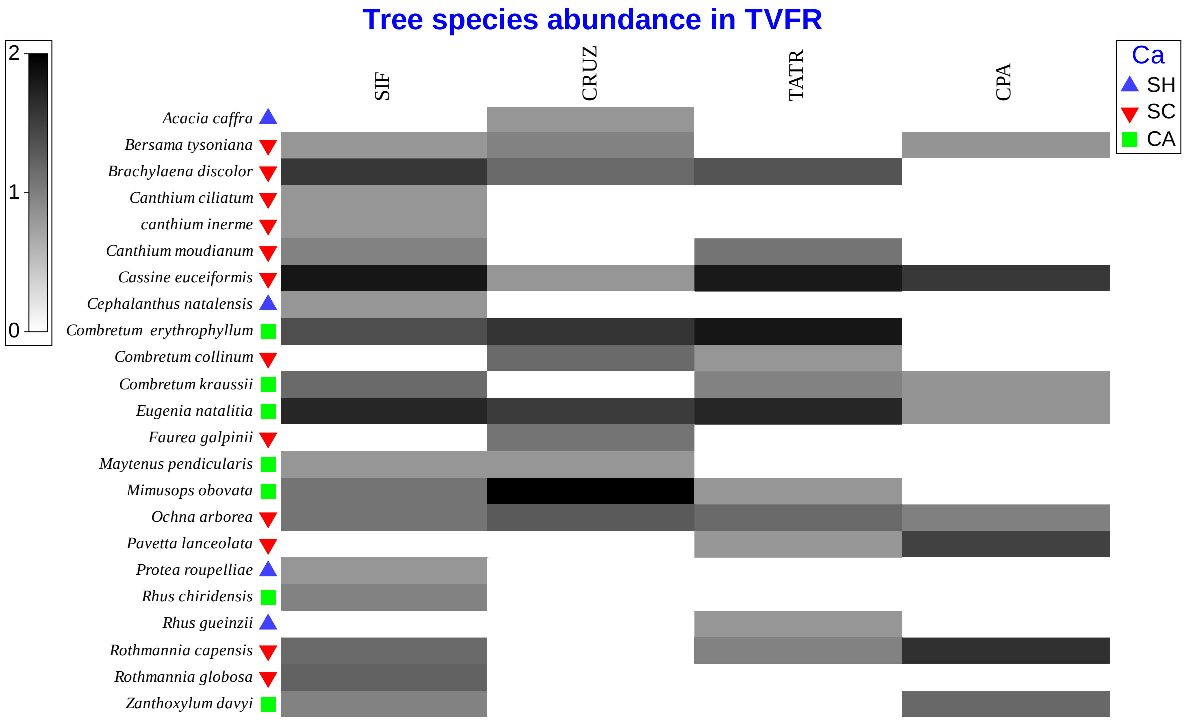

Figure 3.6. Shade plot of average endemic tree species abundance (4th transformed) in MFR(Mafhela Forest Reserves).N.B. SIF= State-protected indigenous forests, CRUZ=Common resource use zone, TATR=Trees along rivers and streams, CPA= culturally protected forests, SH =Shrub, SC=sub-canopy tree, and CA=canopy tree. N.B. White in the shade plot represents absences and the depth of gray is proptional with abundance

3.4. Discussion

The results of this study showed that the patterns of different facets of tree species diversity among land use regimes differed (β-diversity and species abundance distribution) between Mafhela (MFR) and Thathe Vondo (TVFR) Forest reserves (Figure 11). State-protected indigenous forests (SIF) were relatively more effective in supporting the overall tree species diversity in their respective landscapes (Table 3.1; Figure 12). However, the efficiency of state-protected indigenous forest in protecting the conservation value of priority species (rare old-growth forest and endemic tree species) from human land use disturbance in comparison with the potential of tree-based traditional land use regimes is context-dependent.

A recent study by Araia and Chirwa (2019) showed that tree-based traditional land use decision making by local communities in both Forest reserves is a non-random activity. Communities in both Forest reserves (FRs) share similar cultural practices in managing their multifunctional landscape for various forest products in pursuit of their rural livelihood using cultural norms and taboos. However, due to the fact that the two communities reside in different conditions of forest landscape complexity, the contrasting response of local assemblages to disturbance is not a surprise. The condition of forest landscape complexity moderates the resilience of local species diversity after disturbance through the supply of propagules from the species pool and due to neighbourhood effect (Tscharntke et al. 2012). The higher forest landscape complexity provides higher insurance to the resilience of local assemblages.

Nonetheless, higher tree species diversity in both State-protected indigenous forests does not necessarily mean that state protection of indigenous forest is more superior in protecting vulnerable tree species than the other traditional land use regimes. Instead, under the growing limited global conservation resources, the central question should be to understand the conditions under which different conservation options become efficient to conserve species that are more vulnerable to local extermination. Based on the findings, this discussion attempts to determine the conditions under which the different options become efficient.

3.4.1. Linking β-diversity of overall local assemblages to change in species abundance distribution

Overall, the local assemblage of all land use regimes in MFR was distinct from each other (difference in Mean β-diversity). The species that contributed most to the dissimilarity between SIF and the other land use regimes were all wide range species that suit a particular environment created by a specific land use intensity. The difference in Mean β-diversity among land use regimes in MFR implies the breakdown of total habitat area of the landscape (ecological fragmentation) into spatially segregated environments. Thus, a particular land use regime endows a set of co-existing species of life-history traits that are suited to the particular environment created by the disturbance intensity (Johnstone et al. 2016), as expected by IDH (Pulsford et al. 2016). It also shows that there was a limited influence of species pool and weak neighbourhood effect on the recovery of local assemblage after local disturbance (Tscharntke et al. 2012). However, the fact that CRUZ showed different pattern of variance β-diversity that reduced variance β-diversity of TATR and CPA implies that they differ on their impact on the species abundance distribution (Avolio et al. 2015).

The findings showed that both state-protected indigenous (SIF) and CRUZ (Common resource use zone) had the highest variance β-diversity of MFR, followed by intermediately disturbed TATR (Trees along rivers and streams) and cultural protected areas (CPA) in decreasing order. Given Araia and Chirwa's (2019) findings that local people in the study area still extract forest products from SIF, the highest variance β-diversity of strictly protected SIF to overall β-diversity appears to be odd. However, this most likely shows that state-protected areas may still reduce the disproportionate impact of human disturbance in comparison with CRUZ due to fear of legal retribution (Cetas et al. 2015) or their distant location from settlement areas (Spracklen et al. 2015). As such, the amount of forest harvesting from SIF might not be so severe as to override the ecological integrity and natural conditions. The maintenance of ecological integrity and natural conditions are crucial for the persistence of species and habitats (Jones et al. 2018) and for rapid recovery after disturbance. However, the similarity of pairwise variance of CRUZ with SIF shows that the highest intensity of land use disturbance exerted the highest stress level to local assemblage (Anderson et al. 2008). As such, the dominant canopy and sub-canopy trees of SIF were replaced by shrubs (e.g., *Psidium guava, Parinari curatellifolia*, and *Annona senegalensis*) in CRUZ. Shrubs tend to adapt to a disturbed and unproductive habitat better than trees due to morphological and physiological superiority (Götmark et al. 2016). The slight decline in evenness and species richness, in the presence of replacement of tree species of the original biota by shrubs, only highlights the severity of local extermination of the vulnerable canopy and sub-canopy trees (Hillebrand et al. 2018). Avolio et al. (2015) suggested that the difference in mean β-diversity that was accompanied by equal variance β-diversity between local assemblages hints that the change due to disturbance may be uniform within the land use regimes under consideration.

The local assemblage of trees along the river and streams (TATR) differed in mean and variance β-diversity with SIF. Also, variance β-diversity in TATR declined in comparison with SIF and CRUZ. The difference in mean β-diversity confirms the dissimilarity in the local assemblages. The decline in variance β-diversity highlights that the few remaining TATR fragments are relatively stable. Avolio et al. (2015) suggested that the difference in mean β-diversity that accompanied by decline in variance β-diversity between local assemblages hint that the dominance of certain species throughout the land use regimes under consideration. While some of the canopy and sub-canopy trees of SIF declined in abundance and others disappeared, it still retained few canopy trees that were rarely found in SIF (e.g., *Bridelia micrantha* and *Ficus capensis*). *Bridelia micrantha* and *Ficus capensis* are mostly found in moist riverine and stream habitats in southern Africa regions (van Wyk and van Wyk, 1998). Although the local assemblage of TATR appeared to have similar evenness in species abundance with CRUZ, the species richness declined substantially. Based on the field observation, the decline of species richness is mainly due to the current expansion of horticultural fields. Many studies have been reporting the decline of species richness with the decline of forest habitat area (Lawton, 1999).

Culturally protected areas (CPA) showed a remarkable difference in mean and variance β-diversity with SIF. CPA had the lowest variance β-diversity that implies it is more stable than SIF. Hence, there was a sharp decline in species evenness in CPA (increased dominance). Despite the disappearance of the canopy and sub-canopy trees, CPA had canopy trees (*Englerophytum maglismontanum* and *Podocarpus falcatus*) that were extremely rarely found in SIF. This could be, as in other parts of the world (Ormsby and Bhagwat, 2010), where local people respect social norms and rules that govern culturally protected areas (Sacred forest) due to their religious and symbolic significance. Hence, the lowest disturbance intensity in CPA might have enhanced the competitive exclusion of shrub species or reduced their abundance (Pulsford et al. 2016).

In contrast, the overall local assemblage of all land use regimes in TVFR appeared to be similar despite the similarity slightly declined in highly disturbed CRUZ (Figure 11; Table 3.3). Overall, the response of the dominant growth form did not conform to the expectation of IDH. The majority of species that contributed most to the pair-wise mean β-diversity were wide-range species, and few endemic species also played a role. Both SIF and CRUZ had the equally the highest variance β-diversity. The variance β-diversity declined equally in TATR and CPA. Unlike in MFR, all land use regimes in TVFR had similar evenness in species abundance and species richness except a substantive decline of overall species richness in culturally protected areas (Figure 12). All land use regimes retained almost all the dominant canopy and sub-canopy tree species of the original biota. Also, the relative abundance of dominant growth form did not consistently change along the protection-use gradient. The intensively disturbed CRUZ was dominated by *Parinari curatellifolia* shrub while the abundance of the canopy and sub-canopy species of SIF declined. On the contrary, TATR and CPA had a higher overall abundance of the canopy and sub-canopy trees than the SIF itself (Table 3.5).

Overall, the better condition of local assemblage in TVFR in all land use regimes may be attributed to better ecological connectivity of a complex landscape. Complex landscapes have higher ecological connectivity that enhances the supply of sufficient propagules from the species pool and through neighbourhood effect among land use regimes during local disturbance-recovery dynamics (Tscharntke et al. 2012). However, species may still be lost or decline in abundance as part of a natural process as is the case in CPA. Many studies have been showing that local people's loyalty to traditional rules that restricts human disturbance in cultural protected areas (sacred areas) due to spiritual and symbolic reasons (Ormsby and Bhagwat, 2010).

3.4.2. Linking β-diversity of local assemblages of endemic species to change in species abundance distribution

There was a significant difference among land use regimes in mean β-diversity at Mafhela (MFR) and Thathe Vondo (TVFR) Forest reserves regardless of their difference in the total abundance of wide range species. Unlike the claim of some studies (e.g., Waltert et al. 2011), the findings imply that land use disturbance does not promote the replacement of endemic species by wide range species. Instead, it is most likely that endemic species and wide range species are inherently segregated across environmental space due to their difference in habitat requirement. Endemic species tend to prefer minor habitats, generally warmer with high potassium, calcium, magnesium, and phosphorus than their counterpart wide geographic range species (Xu et al. 2015). In light of the absence of endemic species in MFR and very few in TVFR contributing most to β-diversity of the overall local assemblage, we suggest that the adverse effect of disturbance on endemic species was most likely overshadowed by the inherent rarity of endemic species (Pimm et al. 2014). In both reserves, the strictly protected SIF retained the highest species diversity in endemic species in comparison with their adjacent land use regimes within the same landscape.

In MFR, all land use regimes had a similar local assemblage of endemic species as SIF. CRUZ was the only exception in that it differed in mean β-diversity with SIF. The local extermination of canopy and sub-canopy trees of SIF in CRUZ was not only severed but also shrub species dominated the remaining CRUZ fragments. This is consistent with Avolio et al. 2015) predication that the change in mean with no change in variance accompanies a major shift in local assemblage across the whole land use regimes. However, this do not imply CRUZ do not have some canopy and sub-canopy species that were not found in SIF and the other land use regimes (Fig 6). On the contrary, the loss of endemic species in TATR was substantive, although it retained the same evenness of species abundance. All the remaining species in TATR were the sub-set of SIF. The severe loss in species is most likely related to the reduction of habitat areas due to the current expansion of horticultural farms along with the riverine ecosystem. CPA had shown a remarkable difference with CRUZ and TATR. Although it retained most of the trees species of SIF, the species abundance distribution of local assemblage became uneven. This implies that the low disturbance intensity due to culturally protection promoted a competitive reduction in the abundance of some species (extinction debt) and promoted the dominance of few competitively superior species.

In TVFR, all land use regimes had similar local assemblage with SIF except CPA. However, all land use regimes differ in variance β-diversity except in CPA. Despite the loss of species and local extermination debt in CPA, all land use regimes had some of the canopy and sub-canopy trees that are rare in SIF. Considering the negative effect of fragmentation on the persistence of isolated and small population size, those trees outside of the state-protected indigenous forests in both FRs can play a significant role for the persistence of endemic species both at local and landscape-scale. Smaller populations in fragmented habitat are more vulnerable to local extermination due to geographic and environmental stochastic, lower genetic diversity and interrupted biotic interactions with their seed and pollen dispersal agents (Hafftman et al. 2003).

3.5. Conclusion

The results from this study showed that the two state-protected indigenous forests supported the highest overall species diversity in their respected landscape attesting to the effectiveness of the strategy in conserving overall tree species diversity- both wide geographic range and endemic species to Southern Africa. However, the two state-protected indigenous forests also showed a substantive difference in protecting vulnerable tree species assemblage from land use disturbance in comparison with the surrounding tree-based traditional land use regimes. Overall, with the growing concerns of the global biodiversity loss and limitation of conservation resources, the study underlines the need for evidence-based priority setting to pro-actively protect vulnerable rare tree species from local extermination. The approach used in this study for evaluating the quality of biodiversity of alternative potential conservation options, using a predefined ecological matrix (rare old-growth forest and endemic tree species), is an attempt in that direction.

The study also shows that the recent claim of the remarkable resilience of species diversity is an oversimplification of a complex and non-linear response of local assemblage to disturbance. The response of local assemblage to land use disturbance depends on a complex set of socio-ecological factors. These include intensity of tree-based traditional land use disturbance, the trait of species under consideration (growth form and geographic range) and the conditions of the surrounding forest landscapes. The findings in MFR highlight the fact that state-protected indigenous forest is a superior conservation strategy- be it to overall species diversity or canopy and sub-canopy tree species. On the contrary, wide range and endemic trees in relatively simplified or highly fragmented human forest landscape are more vulnerable to local extermination due to land use disturbance. In particular, the change in overall tree species diversity into a completely distinct assemblage of shrubs in common resource use zone demonstrates a regimes shift to a different stable state due to land use pressure. Hence, investing the limited conservation resources to reinforce the state-protected indigenous forests along with restoration of the degraded human modified ecosystem should be a top priority. However, the presence of some conservation priority species in TATR and CPA hints to the fact that those land use regimes in human modified landscape can play a significant role as a supplementary conservation strategy.

In contrast with MFR, the findings in TVFR highlight to the fact that the canopy and sub-canopy wide range species in the human modified landscape with better conditions forest landscape complexity are more resilient to disturbance than in a simplified landscape. Except the decline in abundance in CRUZ, conservation priority species were more abundant in relatively less disturbed traditional land use regimes than in state-protected forests of TVFR. However, rare endemic canopy and sub-canopy trees as vulnerable to local extermination as in MFR. In this regard, effective management of the whole human modified landscape may play a complementary role in biodiversity conservation. Hence, empowering local people to sustain their cultural practices through collaborative forest and tree species conservation projects may ensure the persistence of the full of set of vulnerable species with possible least cost.

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CHAPTER 4

Revealing the predominance of culture over the ecological abundance of resources in shaping forest and tree species use-behavior of local people: The case of Vhavenda people, South Africa

Abstract

The resurrection of traditional socio-ecological knowledge system as a complimentary biodiversity conservation tool for poorly performing protected areas has been fuelling a new debate on what drives the resource use-behaviour in human modified forest landscapes. In this study, using ecological assessment and ethnobotanical techniques, we tested if ecological abundance of resource can sufficiently explain the use-behaviour of traditional society for various use; fuelwood, construction materials, livestock grazing and browsing, wild food, and traditional medicines. Both parametric and non-parametric tests were used to analyse the data. The result from two different communities of the Vhavenda people revealed that there was homogeneity of cultural values pertaining to resource use-behaviour although they reside in ecologically different forest conditions. The use-value of habitats increases with the increase of land use intensity gradient in multifunctional landscape as defined by cultural norms and taboos. State-protected indigenous forest was an exception. Despite it presumed strict protection status, it had the same use-value as with open access common resources use zone. While Local people complement their household demands from the trees along the rivers to some extent, there was almost no forest resource harvesting from culturally protected (scared) forests. The study also found that the effect of household characteristics on the difference of user-behaviour was within the bounds of cultural norms and taboo that governs land use decision. Moreover, abundance of species at the forest landscape did not sufficiently explain the use-value of species. Generally, the main findings show that culture plays a predominant role in explaining use-behavior. Neither is resource use decision of forest landscape are random nor is the concept of protected areas a new concept to traditional society. Hence, local action with a genuine partnership with local people, based on respects of cultural values, can play a significant role in averting the global biodiversity crisis.

Key words: cultural value, traditional socio-ecological knowledge, Land use gradients, use-value index, ecological appearance hypothesis

4.1. Introduction

The protected area approach to biodiversity conservation has been claimed to be an effective and efficient strategy under the limited conservation resource scenario. However, the sustainability of its outcome in biodiversity conservation has generated a polarised scientific opinion (Doak et al. 2015). Different global assessments on the performance of protected areas have been suggesting that the majority of strictly protected areas are a failure- both in terms of ecological and social output (Heywood et al. 2018; Oldekop et al. 2016). The challenge for sustainability and effectiveness of protected area approach, in part, lies in its management strategy that often tries to solve biodiversity conservation challenges as purely an ecological problem (Cumming and Allen 2017). It fails to address competing for social demands (Oldekop et al. 2016) and often considers local people as part of the problem of biodiversity loss; not part of the solution (De Pourcq et al. 2016). This contravenes the long-held belief that people use traditional socio-ecological knowledge in managing landscapes since time immemorial.

Traditional socio-ecological knowledge (TSEK) is an adaptive and complex system of experiential knowledge, practices, and beliefs of local people in governing relationships among themselves and with their surrounding ecosystems (Berkes et al. 2004). Using this complex set of TSEK, local people have been making various land use management decisions to obtain multiple benefits in their landscape. These include delimitation of areas of the common resource use zone, culturally protected areas/sacred forests, protection of rare species and rotational resource/successional management of vegetation (Berkes et al. 2004). By doing so, local people have been reconciling the livelihood demands and biodiversity protection sustainably. The knowledge, innovation, and practices of such communities have been co-evolving to adapt to the changing environmental, political and socio-economic changes (Gómez-Baggethun 2014) through trial and error over time (Berkes et al. 2004). The legacy of pro-environmental cultural values and traditional conservation practices still exist in many parts of the world (e.g., Ruiz-Mallen and Corbenra, 2013, Gadgil et al. 1993). Nonetheless, there has been a disagreement surrounding the validity of such claims.

For instance, Low (1996) highlighted that local people's resource user-behaviour is ecologically driven, based on an abundance of resources. It does not correlate with attitude including compliance with sacred protection. According to Low (2014; 1996), the low impact on their environment is not often from a collective conscious effort to conserve their natural resources. Local people do not willingly sacrifice short term benefits with the expectation of greater common good in the long run. The low impact is purely due to a combination of low population density, inefficient extraction technology and lack of profitable markets for extracting products. While the recent meta-analysis by Gonçalves et al. (2016) appear to partly support Low (1996) in that there exists a correlation between abundance of species, and overall use-values of a species, the correlation with a use-value of species for a specific category of utility was inconsistent. The utilities include fuelwood, construction materials, livestock grazing, and browsing, wild food, and traditional medicines that are important for the maintenance of rural livelihood.

Contrary to Low (1996) and Gonçalves et al. (2016), the recent study by Soares et al. (2016) suggests that cultural factors play predominant role over ecological factors in driving plant use and knowledge. The abundance of a species or relative ecological importance of a species does not correlate to use-value. One of the possible explanations for lack of consistent conformity of abundance as a driver for human use-behaviour in Gonçalves et al. (2016) could be due to the mismatch of the theoretical basis of most of the studies to actual human behavior. Most of the ethnobotanical studies were conducted using the "ecological appearance hypothesis" that was formulated by Fenny (1976) and Rhoades and Cates (1976) for a different ecological question. The original intention of the hypothesis was to test if plant species that are visible and abundant are more susceptible to herbivory (De Albuquerque et al. 2005). Unlike free grazing in herbivory, local communities design cultural institutions and social norms (e.g., traditional bylaws, rituals, and ceremonies) to regulate access and to sanction appropriate corrective measures when contravention to governing rules of common resources are detected (Ruiz-Mallen and Corbenra 2013, Colding and Folke 1997; 2001). There are suggestions that the presence of culturally important species that shapes the cultural identity of a particular cultural group in a significant way (Culturally keystone species) (Gaoue et al. 2017) regardless of their state of ecological abundance.

The above disagreement hints on the complexity of human behavior in managing relations among themselves and with the surrounding ecosystems. The inherent assumption of ecological appearance hypothesis that individuals in society are rational and efficient in their choices of a species for various utilities (Gaoue et al. 2017) is too simplistic to predict human use-behavior of communal resources. Human behavior is also influenced by cultural institutions and social norms (Ruiz-Mallen and Corbenra 2013, Colding and Folke 1997; 2001).

Those cultural institutions and norms do not only govern the behavior of individuals towards a specific species (e.g. species related taboos and totems) . Also, they govern collective behavior towards the whole biodiversity of a landscape by using a complex set of TSEK and habitat-related taboos. However, the question of how those cultural institution and norms determine the spatial distribution of land use intensity across different spatial hierarchy of a landscape is not adequately researched (Pásková 2017). Land use intensity is suggested to be a good predictor to explore the relationship between culture and biodiversity since it is often reciprocally affected by both over space and time (Bürgi et al. 2015).

Moreover, all people have a culture that governs their relationship with their surrounding ecosystems. It is often easier to grasp the influence the culture of different people has in their environmental behaviour when the lifestyle of groups is markedly distinct from each other (Head et al. 2005). At the same time, every landscape has a peculiar spatial heterogeneity of ecological resources that have been shaped either by natural forces, cultural disturbance or both (Pickett and Cadenasso 1995). Jointly, the aforementioned arguments imply that there is need to study individuals and communities from a homogenous cultural group who reside in different forest landscapes in order to determine which factors play a predominant role in use- behaviour. Thus, the central question of this study was if culture and ecological abundance have equal power to explain forest and tree use-behaviour of local communities? To the best of my knowledge, there has not been any research to that effect. The findings from the study may provide knowledge to promote sustainable and collaborative conservation of forest landscape that works for biodiversity and local people (Kremen and Merenlender 2018). Based on the above arguments, the study formulated and tested the following hypotheses:

H1: Homogeneity of cultural value: communities from the same cultural group, but residing in different forest landscape conditions, demonstrate similar use-behavior towards similar habitats (land use intensity) as specified by cultural institutions and social norms.

H2 The use-value gradient in the multifunctional landscape: The total use-value of land use regimes in multifunctional landscape increases with the increase of socially perceived land use intensity gradient both at a cultural group and household level.

H3: Ecological appearance hypothesis: local people depend highly on most abundant species in their landscape for various utilities (Phillips and Gentry 1993).

4.2. Research methods

4.2.1. Study area

As the previous studies (Chapter 2 and 3), this study was undertaken at two research sites in the Vhembe Biosphere Reserve (VBR) of South Africa: Thathe Vondo and Mafhela Forest Reserve Areas. This section provides additional information that are relevant to the objective of this particular study.

Both Forest Reserves is dominated by the Vhavenda people. The Vhavenda people are known to have a rich traditional ecological knowledge and social norms to govern their landscape for multifunctional purposes (Mutshinaylo and Siebert 2010, Khorombi 2001). The traditional ecological knowledge has been co-evolving along with many environmental and institutional reforms. For instance, parts of the landscape (Holly forests) that used to be governed by tribal authorities for centuries was replaced by commercial plantations due to forced displacement in terms of the 1936 Native Trust and Land Act No. 18. (von Maltitz and Shackleton 2004). While the displacement in MFR happened in 1979 (Kabanda and Munyati 2010), the displacement in TVFR happened in 1947 (von Maltitz and Shackleton 2004). In both cases, the government also established a few fragmented indigenous forest patches within the parameter of the commercial plantation-which is still under stricter protection purely for conservation reasons.

In the place where local people relocated, there are still remnants of forest patches, including culturally protected areas (Holly forests) under the custodian of traditional leaders. A recent inventory of useful plants by Magwede and van Wyk (2018) revealed that the Vhavenda still relies on about 189 species of trees and shrubs for various utilities including fuelwood, construction materials, livestock grazing, and browsing, wild food, and traditional medicines. However, whether the legacy of their culture or ecology (abundance) of remnant forest and tree species plays a predominant role in user behavior is unknown. Land use regimes were classified a similar way as in chapter 2 and 3.

4.2.2. Data collection and analysis

4.2.2.1. Ecological assessment and analysis

Each Forest Reserve (FR) consisted of three land use regimes of the HMFLs (CRUZ, TATR, and CPA) and SIF. Identification and location of land use regimes within the forest reserves were done with the guidance of local informants. A similar approach of identifying atypical land use regimes by local informants was used by Sheil et al. (2002) and believed to be appropriate to capture the local people's perception and tree management practices in rural landscapes (Sheil et al. 2006). In each landuse regime, five transects (Tr) were established (except in 3 in TATR and Mafhela Forest Reserve). Each transect was 50 m long and separated from each other at least by 200 m. In each transect, three (3) 20 m x 10 m rectangular plots (P) was established and spaced 10 m apart along a linear transect.

All perennial woody plants with a diameter height of ≥ 2 m were considered as trees (van Wyk and van Wyk 1998). Wyk and van Wyk (1998) suggesting that the artificial distinction between shrubs and trees often breaks down in practice. For instance, despite a typical tree as being considered to have a single trunk, it may also have multiple stems. In this study, all trees at breast height (dbh) ≥ 2 cm were enumerated (Pinard et al. 2013). The scientific and vernacular names (from local informants) of observed tree species in each plot were recorded. In instances where tree species identification was not possible in the field, tree voucher specimens were collected and later identified at the Thohoyandou Botanical Garden and Herbarium.

The effectiveness of the sampling effort on species observed (S_{ob}), for the whole study area and each reserve, was evaluated using species accumulation curve based on Bootstrap estimators in Primer-E (Clarke et al., 2015). Sampling effort that captures ≥80% of the estimated species richness can be considered effective (e.g. Foggo et al., 2003). Because, our sampling effort was 88.70 % of the whole study and about 88.00% for each FR, it was then only the actual observed species that were reported in this study. The similarity in species composition between the two FRs was calculated using Jaccard similarity coefficient. To compare the similarity of species abundance between the FRs, two-way similarity percentage (SIMPER) analysis was used (Cut-off=70%) (Equation 2.1.).

4.2.2.2. Ethnobotanical assessment and analysis

In this study, a sequentially mixed sampling technique was used to select villages and sample population (Onwuegbuzie and Collins 2007). First, four villages from Thatho Vondo Forest Reserve (TVFR) and Mafhela Forest Reserve (MFR) were purposefully chosen as these villages were located within the perimeter of the Forest Reserves. Tshidzive and Tshilungwi in Thathe Vondo Forest Reserve have 312 and 253 households respectively; while Belemu and Tshiema in Mafhela Forest Reserve have 99 and 113 households, respectively (Figure 4.1).

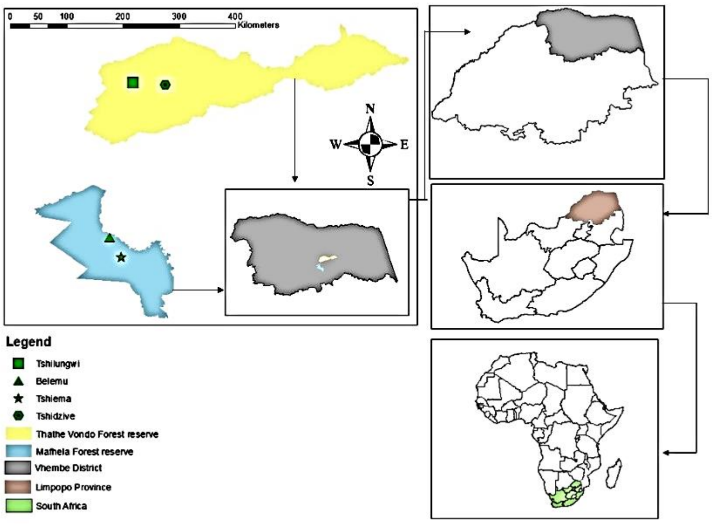

Figure 4.1. Location of the study area in the Vhembe Biosphere Reserve in Limpopo Province

This was then followed by determining the sample size of respondents (n) required out of the total population (N=770) using Watson (2001) equation as stated below:

$$n = \frac{\frac{P(1-P)}{\frac{A^2}{z^2}+\frac{P(1-P)}{N}}}{R} \qquad (1)$$

Where n=sample size required, N =number of people in the population, P=estimated variance of a population, as a decimal, A=Precision desired, as expressed in decimal, Z=Based on confidence level, R=Estimated response rate, as a decimal. Consequently, the required sample size (n) was calculated with an estimated variance in population (P) of 30 %, estimated precision of 5 %, and confidence level of 95% (Z=1.96) and estimated response rate of 95%. Accordingly, the required sample size was approximately 78 households representing a sampling intensity (n/N) of 10.13 %. However, this was increased to 20% when 135 households were interviewed thereby decreasing the sampling error and increasing the reliability of the sample statistic to estimate the population parameter (Bhattacherjee 2012). The actual households for the interviews were selected using systematic random sampling, constituting every 5th household, from the list of residents provided by local chiefs. Where interviews with household heads were not possible, a person above 21 years, available in the household during the interview, was considered with their consent.

Household Surveys

Both structured questionnaire and free-listing (da Silva et al. 2014) were developed based on a preliminary analysis of the relevant literature and interviews of the forestry sector officials and field observations of the livelihood activities of community members in the study area (Annexure 4.1). The information collected included household characteristics (gender, household size, age, marital status, and educational status), and the kind of forest utilities for their livelihood along all land use intensity gradient. The category of utilities was predetermined based on the preliminary analysis of the field visit. These include fuelwood, construction materials, livestock grazing, and browsing, wild food, and traditional medicine. Table 4.1 shows the household characteristics of the respondents. The questionnaire was then translated into the local language and pretested on six households in Belemu (Annexure 4.2).

Table 4. 1 Household characteristics of participants in the study areas

Household	**Category**	**No**	**%**	**Household**	**Category**	**No**	**%**
Gender	Female	82	60.70	Marital status	Married	57	42.23
	Male	53	39.30		Single and other*	78	57.77
HH size	Small (1-3)	26	19.30	Age	Young adults (21-40)	25	18.52
	Large (≥ 4)	109	80.70		Middle age (40-60)	27	20.00
					Olds (≥60)	83	61.48
Educational	No education	28	20.70				
	Primary school	47	34.80				
	≥Secondary school	60	44.4				

Single and other * includes household respondent who are not married, divorced or widows

Data analysis

First, four different ethnobotanical importance indices were calculated. For household use-value for a specific land use regime, the data was coded one (1) for those who affirmed use and zero (0) for those who did not use a particular land use regimes for a specific utility. This was then followed by calculating the total use-value of a specific land use regime by summing the number of utility a household uses from a species divided by the total number of utility (da Silva et al. 2014). Fidelity level of a given land use regime was then calculated by dividing the number of informants who affirmed the use of the land use for a specific use of the total numbers of informants involved in the interviews in percentiles. Percentage of fidelity reflects the informant consensus on the extent of the importance of a particular item for a specific utility (da Silva et al. 2014, Friedman 1986). Finally, the use-value of a species for a specific utility was calculated by dividing the number of informants who cited a species for specific use by the total number of informants (Rossato et al. 1999, da Silva et al. 2014). The total use-value of a particular species was then calculated by summing the use-value of a particular species of all utilities (da Silva et al. 2014).

The homogeneity of cultural values

The homogeneity of cultural value between the two communities was analyzed using the Mann-Whitney U test for the significant difference using the Statistical Package for Social Sciences (SPSS) version 20 (SPSS 2011). We used the household use-value as the proxy for cultural values with the assumption that local community with the same culture will have the same extent of use of forest and tree species products for the same land use gradient, as defined by cultural institutions and social norms. This will happen despite the difference in forest landscape conditions of their residence. When a significant difference was detected in Mann-Whitney U test, the effect size (r) was calculated by dividing the standard mean rank (Z) by the square root of population size ($r= Z/\sqrt{N}$) (Morgan et al. 2013). The effect size (r) is considered smaller ($r= |0.10|$), medium ($r= |0.30|$), larger $|r=0.50|$) and very large effect ($r\geq |0.70|$). The result was considered significant at $p = 0.05$.

The use-value gradient in multifunctional landscape

To investigate the consistency of use-value along a land use intensity gradient, the analysis was done both at a cultural group and household level. Because the Mann-Whitney U test showed homogeneity of cultural value between the two Forest reserves, first, a contingency table of fidelity value of each land use was developed for the whole study area as one cultural group. This was crucial to determine which forest land use was most popular and for what utility. This was then followed by Friedman rank test to determine if there was a significant difference in mean total use-value (sum of all utilities) of households among land use regimes. A Friedman rank test is a non-parametric analogue of ANOVA for related samples that is used to analyse data when the homogeneity of variance assumption is violated. When a significant difference from Friedman rank test was spotted, the Wilcoxon rank test was used to test the significant difference of mean total use-value between different pairs of the land use gradient (Morgan et al. 2013). Secondly, to determine if household characteristics (gender, household size, and marital status) affect mean total use-value in each land use regimes, we used a Mann-Whitney U test (for gender and marital status).

The use-value of species and the ecological appearance hypothesis

To describe the number of useful species for each utility category, all species (at least for one use) per FR that were cited by participants were enumerated. That was then turned into a percentage out of the total species encountered during the field inventory. The number of species cited for the specific utility was enumerated and turned into a percentage of all useful species cited in each FR. This study only reports the top useful species based on their order of total use-value. The mean use-value a species for each utility was calculated using summary statistics routine in Premier-E (V7) software.

To test for significant difference of ecological appearance hypothesis, the average abundance of species from the forest inventory results, species use-value for a specific utility and all utilities were subjected to a Spearman correlation test. The Spearman correlation test was chosen as the data was not conforming to the normal distribution required for the parametric test. In this analysis, we only correlate only those cited species that during the forest inventory (Ribeiro et al. 2014). This was done separately for the two Forest reserves because the relationship between usefulness and appearance might be specific to the area of influence (e.g., Macía 2008; Guèze et al. 2014).

4.3. Results

4.3.1. Ecological assessment

The study recorded 2125 individual trees and 110 tree species in total in the area. The total number of species encountered in MFR and TVFR was 72 and 88, respectively. The two forest reserves are dissimilar in their species composition (55.5%) and contain distinct local assemblages in terms of species abundance (Av. Dis=91%). The mean number of trees per plot for MFR and TVFR were 8.62 and 10.52, respectively. The SIMPER analysis output (Table 2) revealed that the MFR was dominated by seven tree species that contributed about 70% of the total abundance of tree species for the whole landscape, out of which *Englerophytum maglismontanum*, *Bridelia micrantha*, and *Psidium guajava* accounted for 50 % of the total abundance of the trees. In TVFR, the local assemblage was relatively diverse and 12 tree species dominated and contributed about 70 % of the total abundance, out of which *Syzygium gerrardii*, *Xymalos monospora*, *Englerophytum maglismontanum*, *Aphloia theiformis*, *Podocarpus falcatus*, and *Cassine eucleiformis* accounted for 50% of the total abundance of the tree of the landscape. Table 2 shows the average abundance of the dominant species and their contribution to the similarity of species distribution within each forest reserves.

Table 4. 2 Average abundance (Av.abu) of tree species and their contribution (% Con) to the similarity of species distribution to forest reserves

Scientific name	Mafhela Forest reserve (MFR)			Scientific name	Thathe Vondo Forest reserve (TVFR)		
	Av.abu	Sim/SD	% Con		Av.abu	Sim/SD	% Cont
Englerophytum maglismontanum	3.06	0.67	39.13	*Syzygium gerrardii*	1.15	0.38	10.31
Parinari curatellifolia	1.44	0.25	3.83	*Cassine euceiformis*	1.02	0.42	5.37
Psidium guajava	1.09	0.27	5.99	*Aphloia theiformis*	1.00	0.41	7.72
Bridelia micrantha	1.00	0.36	6.4	*Xymalos monospora*	0.97	0.52	9.99
Annona senegalensis	0.93	0.28	4.13	*Englerophytum maglismontanum*	0.85	0.48	6.91
Aphloia theiformis	0.67	0.36	5.24	*Eugenia natalitia*	0.83	0.34	3.92
Ficus capensis	0.65	0.28	3.87	*Parinari curatellifolia*	0.81	0.24	4.83
				Podocarpus falcatus	0.75	0.3	6.11
				Schefflera umbelifera	0.75	0.35	4.26
				Olea capensis	0.68	0.33	4.4
				Mimusops obovata	0.66	0.36	6.45

Av. Abu= average abundance, Sim/SD= Similarity/SD, and % Con= Percent contribution of a species

4.3.2. The homogeneity of cultural values

Forty-five (45) respondents in MFR and ninety (90) in TVFR were interviewed in the study. All of the respondents extracted at least one forest utility from their landscape to sustain their livelihood. Mann-Whitney U test between the mean rank of the total use-value of forest utilities for a rural livelihood from MFR SIF (66.81), and TVFR SIF (68.81) was not significantly different (U=2,098; p=0.727). Although there was a statistically significant difference between the mean rank of total use-value of MFR CRUZ (55.89) and TVFR CRUZ (74.06) (U=2,570; p=0.01), the difference in mean ranks for the two land use regimes in total use-value was very weak (r=0.21). There was no significant difference in the mean rank of the total use-value of MFR TATR (70.19), and TVFR TATR (66.91) (U=1,926; p=0.60). Similarly, there was no significant difference in the mean ranks between MFR CPA (66.78) and TVFR CPA (66.81) (U=2,080; p=0.737), in total use-value.

4.3.3. Use-value along a land use intensity gradient

The contingency table for the fidelity value (Table 4.3) shows that local people relied on forest landscapes for various forest utilities to sustain their rural livelihood. The most popular utility was wild food, followed by fuelwood, livestock grazing, and browsing, wood for construction materials and traditional medicine in descending order. As predicted in the hypothesis, the total use-value of a land use regime increases with the increase in land use intensity gradient in the human modified part of the landscape with the exception of SIF. The total use-value in human modified landscape increased from CPA, TATR, and CRUZ, respectively. However, despite the presumed strict protection in SIF, it had almost the same total use-value as CRUZ for all utilities followed by TATR and CPA.

Table 4. 3 The fidelity value of land use regimes and forest reserve

Utility category	Fidelity value of land use regimes				
	SIF	CRUZ	TATR	CPA	LS FV $_{tot}$ for specific utility
Fuelwood	60.00	51.85	14.81	4.44	131.1
Construction material	34.81	36.30	5.93	3.70	80.74
Grazing and browsing	50.37	43.70	28.15	0	122.22
Wild food	53.33	50.37	23.70	20.74	148.14
Traditional medicine	33.33	30.37	5.19	2.22	71.11
LU FV $_{tot}$ for all utilities	231.85	212.59	77.78	31.11	553.31

SIF=indigenous forests; CRUZ= Common resource use zone, TATR= Trees along rivers and streams, and CPA= Culturally protected forests, LS FV $_{tot}$ = Landscape total fidelity value, and LU FV $_{tot}$=Total fidelity value of a land use

A Friedman rank test for total use-value among land use regimes confirmed that there was a significant difference among the mean ranks of land use in their total use-value (χ^2 = 136.84; df=3; p=0.001). The Wilcoxon test with Bonferroni correction (Comparison wise alpha=0.017) showed that all land use regimes differ in their total use-value from each other except between (CRUZ, SIF). In both cases, the highest ranks imply that local people attach equally high total use-value to both CRUZ (3.09) and SIF (2.97) followed by TATR (2.17), and CPA (1.77). The effect size analysis (Table 4) showed that, except for no difference between (SIF, CPA), the difference between (TATR, CPA) was a medium while the other land use regimes pairs were large or larger than expected in behavioral studies.

Table 4. 4 Wilcoxon rank test results between pairs of land use regimes in their total use-value

Pairs of land use regimes	Mean ranks	P (Probability value)	Effect size(r)
(CRUZ, SIF)	(3.09, 2.97)	0.278	0.09
(CRUZ,TATR)	(3.09, 2.17)	0.00	0.62
(CRUZ, CPA)	(3.09, 1.77)	0.00	0.71
(SIF,TATR)	(2.97,2.17)	0.00	0.57
(SIF,CPA)	(2.97,1.77)	0.00	0.68
(TATR, CPA)	(2.17;1.77)	0.00	0.42

SIF=indigenous forests; CRUZ= Common resource use zone, TATR= Trees along rivers and streams, and CPA= Culturally protected forests

At household level (Table 4.5), Mann-Whitney U test on the effect of gender on forest use harvest showed that there were no significant differences in mean ranks of total use-value among all land use regimes (P≥0. 05). Marital status did not also affect the total use-value of land use regimes (P≥0. 05), except for the mean ranks of total use-value in SIF. The mean ranks of married women (76), and other groups (62.15) were significantly different (U=1767; p=0.05; r=0.17). The effect of household size on total use-value showed a significant difference only in CRUZ, and SIF. In CRUZ, the mean rank of small size household (82) was significantly different from the mean rank of large size household (64) (U=1053, p=0.04, r =0.17). In SIF, the mean rank of small size household (49.5) was significantly lower than the mean rank (72.40) (U=937, p=0.00, r=-0.23).

Table 4. 5 The effect of household characteristics on the total use-value among land use regimes

Land use	Gender				Marital status				Household size			
	Mean ranks		U	P	Mean ranks		U	P	Mean ranks		U	p
	Male	Female			Married	Not			Small	Large		
CRUZ	69.36	67.12	2101	0.74	64.77	70.36	2039	0.40	82	64	1053	0.04*
SIF	71.41	65.8	1993	0.40	76.00	62.15	1767	0.04*	49.56	72.40	937	0.00**
TATR	73.93	64.16	1858	0.10	68.42	67.69	2199	0.90	62.13	69.40	1264	0.34
CPA	73.90	64.19	1860	0.06	69.47	66.92	2139	0.62	60.92	69.69	1233	0.18

CRUZ=Common resource use zone, SIF=indigenous forests, TATR=Trees along rivers and streams, CPA=Culturally protected areas, U= Manny U test static, and p=Probability

4.3.4.The use-value of species and the ecological appearance hypothesis

4.3.4.1. The use-value of tree species

Overall, local people cited sixty-eight (68) useful species out of the one hundred ten (110) species found during the field inventory. Twenty-eight (28) species and forty-two (42) species were cited in MFR and TVFR, respectively (Annexure 4.1). Table 4.6 shows that the overall percentage of useful species for MFR was roughly 39% of the total species enumerated in the forest reserve. People in MFR cited the highest number of useful species for fuelwood and grazing, followed by construction materials, traditional medicine, and wild food in descending order. The overall mean use-value of a species was 0.35. In terms of the top ten useful species and their use-value, *Bridelia micrantha* was the most important multipurpose species while *Aphloia theiformis* was the least (Table 4.7).

In TVFR, the overall percentage of useful species for TVFR was 47.72%. People in TVFR cited more numbers of useful species than people in MFR. People cited the highest number of useful species fuelwood followed by traditional medicine, construction materials, livestock grazing, and wild food was cited in descending order (Table 4.6). The overall mean use-value of a species was 0.39. In terms of the top ten useful species and their use-value, *Parinari curatellifolia* was the most important multipurpose species while *Syzygium cordatum* was the least (Table 4.8).

Table 4. 6 Number and percentage of useful species per each utility group

Utility category	Mafhela Forest reserve				Thathe Vondo Forest reserve			
	S	%S	Ȳ	Range	S	%S	Ȳ	Range
Fuelwood	20	71	0.12	0-0.59	31	73.80	0.18	0-1.00
Construction	17	60.71	0.04	0-0.24	20	47.62	0.04	0-0.35
Grazing and browsing	20	71	0.04	0-0.36	19	45.24	0.09	0-0.95
Wild food	15	53.57	0.09	0-0.63	18	42.85	0.05	0-0.62
Traditional medicine	17	60.71	0.05	0-0.28	29	69.04	0.03	0-0.31
Total	28	39%	0.35	0.02-1.59	42	47%	0.39	0-1.65

S= number of useful species, % S = Percentage of species for a specific utility, Ȳ =Mean use-value,

Table 4. 7 The top ten most important species in Mafhela Forest reserve and their use-value

Scientific name	FW	CON	G&B	WF	TM	Total
Bridelia micrantha	0.53	0.24	0.36	0.35	0.11	1.59
Parinari curatellifolia	0.59	0	0.16	0.18	0.28	1.21
Englerophytum magalismontanum	0.175	0.025	0.04	0.63	0	0.87
Celtis Africana	0.35	0.23	0.04	0.065	0.025	0.71
Brachylaena rotundata	0.38	0.185	0.08	0	0.02	0.665
Syzygium cordatum	0.11	0.065	0.04	0.27	0.085	0.57
Nuxia floribunda	0.275	0.115	0.02	0.04	0.05	0.5
Combretum molle	0.26	0.02	0.02	0	0.165	0.465
Mimusops obovata	0	0	0.02	0.325	0.04	0.385
Aphloia theiformis	0.23	0.09	0.04	0	0.025	0.385

FW=Fuelwood, CON=Construction material, G&B=Grazing and browsing , WF=Wild food, TM=Traditional medicine

Table 4. 8 The top ten most important species in Thathe Vondo Forest reserve and their use-value

Scientific name	FW	CON	G&B	WF	TM	Total
Parinari curatellifolia	1.00	0.09	0.26	0.21	0.09	1.65
Englerophytum magalismontanum	0.46	0.00	0.05	0.95	0.01	1.48
Enterspermum rhodensiacum	0.86	0.35	0.03	0.00	0.03	1.27
Olea capensis	0.89	0.27	0.00	0.00	0.04	1.20
Syzygium gerrardii	0.56	0.03	0.02	0.57	0.01	1.19
Combretum molle	0.56	0.07	0.06	0.00	0.31	1.00
Olea africana	0.68	0.17	0.01	0.12	0.00	0.98
Albizia adainthifola	0.32	0.02	0.62	0.01	0.00	0.98
Mimusops obovata	0.23	0.05	0.00	0.67	0.01	0.97
Syzygium cordatum	0.43	0.06	0.01	0.18	0.01	0.70

FW=Fuelwood, CON=Construction material, G&B= Grazing and browsing, WF=Wild food and TM=Traditional medicine

4.3.4.2. Appearance hypothesis

In MFR, it has been observed that the abundance of species had a positive and moderate correlation with the overall use-value of a species (r_s=0.44, p=0.00). Similarly, abundance of species showed a positively moderate correlation with abundance of a species with the other utilities; fuelwood (r=0.31, p=0.008), construction material (r_s=0.35, p=0.002), wild food (r_s=0.46, p=0.000), livestock (r_s=0.53, p=0.000) and traditional medicine (r_s=0.33, p=0.000) (Annexure 4.3).

In TVFR, the correlation of abundance of a species was positive and moderately correlated with the use-value of a species (r_s=0.37, p=0.00). Except for a moderate correlation between average abundance and use-value of a species for fuelwood (r_s=0.43, p=0.00) and construction material (r_s=0.44, p=0.00); there was no correlation between abundance of species for livestock browsing and grazing, wild food and traditional medicine (P≥0. 05) (Annexure 4.3).

4.4. Discussion

Considering (i) the scepticism on the sustainability and effectiveness of the existing global protected areas to safeguard tropical biodiversity (Oldekop et al. 2016) and (ii) doubts on the conservation behavior of traditional society towards their surrounding ecosystem- on which they directly or indirectly rely on maintaining their lifestyle (Low 1996;2014)), empirical evidence on drivers of natural resource use-behavior are crucial. The current uncertainty on the predictive capacity of ecological appearance hypothesis implies that ecological resource use-behaviour of local people are far more complex than to be exclusively associated with an abundance of ecological resources. In this case study, we examine the predominance of culture in shaping forest and tree use-behaviours' of traditional society in human modified forest landscapes and discuss their implication to the conservation of biodiversity. This case study was done in two Vhavenda communities who share the same culture. However, the two communities reside in forest landscapes that markedly differ in their ecological conditions (species richness, identity, and abundance) (Section 3.1, Table 3.2).

4.4.1. Homogeneity of cultural values towards similar land use regimes in different ecological conditions

The significance of culture is often easier to grasp when the lifestyle of groups are markedly distinct from each other (Head et al. 2005). In hindsight, the findings from this study on the homogeneity of use-values of similar land use regimes/habitats of local people who reside in two distinct forest landscape conditions imply that forest condition does not play a primary role in local people's forest use-behavior. Instead, considering the two Vhavenda communities share the same culture, it highlights that the shared values and norms (a complex set of knowledge, belief, and practices) play a predominant role in actively using or managing their forest landscape than the conditions of forest resources.

The interpretation of the finding may appear at odds with the recent global assessment by Aswani et al. (2018) who highlighted that forced displacement or significant reduction of access to cultural resources and institutional reforms are some of the critical drivers of the global loss of traditional socio-ecological knowledge. Notwithstanding, the complete loss of traditional socio-ecological knowledge or a substantive shift in culturally shared values can only happen in areas where there has been a significant shift in lifestyle due to large scale ecological devastation or complete integration of traditional society to a market economy (Manfredo et al. 2016), in an unsustainable manner. In the case of this study, although the two Vhavenda people were forcefully displaced from their landscapes like many of historically disadvantaged communities in South Africa (Tshidzumba et al. 2018), they still reside within the parameter of their cultural landscapes. The remaining forest and tree resources in Vhavenda people are still intertwined into their livelihood, cultural, emotional, spiritual and symbolic values of their lifestyle (Magwede and van Wyk 2018; Mutshinaylo and Siebert, 2010; Khorombi, 2001). This is not to argue that forced displacement or institutional reforms in traditional society do not affect the culture assets or traditional socio-ecological knowledge. Instead, it is to emphasize the dynamic and adaptive nature of cultural values. Acquiring new knowledge through consistent trial and error to fit the changing social and biophysical environment has been part of human evolutionary history (Reyes-García et al. 2016). With the range of normal cultural change (Head et al. 2005), a shift in cultural value proceeds incrementally and follows a predictable manner. Complete replacement of one set of cultural value of individuals and society by another set of new values does not occur (Manfredo et al. 2016).

4.4.2. Use-value gradient is consistent with a socially perceived land use intensity gradient

Overall, this study found that forest landscape is locally popular as a source of wild food, fuelwood, livestock grazing, and browsing, construction materials, and traditional medicines in descending order. Consistent with the hypothesis, the use-value of a land use regime for those products depends on culturally defined land use intensity gradient with the exception of SIF which is also embedded within the same landscapes. On conformity with traditional rules, local people largely depend on open access common resource use zone for most of the utilities to sustain their livelihood. Few individuals complement their livelihood demands for relatively less destructive resource demands (grazing, and wild food harvesting) from trees along the rivers and streams. Local people do not almost extract any use from traditional protected areas. In contrast, strictly indigenous forests appear to provide almost equal use-value as the open access common resources use zones.

The case of the state-protected indigenous forest may imply that better protection of state- indigenous forest most likely enhanced the ecological abundance of resources that are important for rural livelihood. Local people may not extract large amount of forest products from state-protected indigenous forests due to fear of retribution in comparison with common resource use zone. However, the comparable total use-value with common resource use zone of human modified landscape implies that its effectiveness to protect areas of high ecological importance (e.g., the abundance of species, population or ecosystems) may not be sustained in isolation. This not surprising considering the challenge of enforcement of protection rules in most forest reserves in South Africa (Shackleton 2009).

The prevalence of use-value gradient along socially perceived land use intensity gradient, in contrast with the state-protected indigenous forest, suggests that tree-based traditional land use decisions or forest extraction are culturally bound and non-random. At least in the case of this study, the lower use-value of trees along the rivers and streams and culturally protected areas is a testimony against Low's (1996) assertion on the lack of correlation between forest and tree species extraction with attitude (including compliance with sacred protection). Similar to the findings of this study, Mutshinaylo and Siebert (2010) claim that in most parts where the Vhavenda people reside, certain species and components of forest ecosystems (e.g., Streams and rivers) are still culturally protected due to the rituals, mythical beliefs, and totems. Many studies have been showing cultural protected sites create habitats for rare and threatened species (Paneque-Gálvez et al. 2018, Sutherland et al. 2003; Schneider 2018). The comparable total use-value of State-protected indigenous forest with open access common resources use zone may not imply either local people's lack of conservation attitude or the pre-dominance of abundance in governing their use-behaviour. Instead, it highlights, imposing strict protection measures based on the abundance of ecological resources or without considering the cultural value of local people may not deter local people from breaching conservation rules.

Culturally bound society does not imply all members of society are homogenous (Schneider 2018) and have similar knowledge and attitude towards forest and tree species uses. Still, the difference in household characteristics can affect their use-behaviour (e.g., Soares et al. 2017, Gaoue et al. 2017). For instance, this study found that, while common resource use zone is accessible for use to all members of society, it was small household size who depends more on it than large size households regardless of their gender and marital status. However, it was the reverse when it comes to indigenous forests. The first case may imply that the smallsized households may have a relatively small demand that can easily be satisfied by harvesting from the nearest common resource zone. In contrast, the married and large household may have higher demanded and large manpower to collect forest and tree species products even from relatively afar indigenous forests. However, none of the household characteristics affect the use-behaviour in trees along the river and streams and culturally protected areas. The fact that the effect of household characteristics only manifested in common resource use zone and indigenous forests highlights even these factors generate a difference within the bound of cultural influence. Common resource use zone is culturally open access for every individual member of society while harvesting forest products from state-protected indigenous forests are a new norm; not an exception to anyone.

4.4.3. The use-value of species and the Ecological appearance hypothesis

Although the residents of the two forest reserves share the same culture, the overall number of useful species and their percentage out of total species richness per landscape was higher in TVFR than in MFR. This could imply that the difference of resource availability in the two forests (e.g., Lyver 2019, Macía 2008, Guèze et al., 2004) may have provided a different extent of a knowledge base on potential uses of species through lived experience.

Nonetheless, the mean total use-value of a species and mean use-value for a specific utility remained similar in both areas (Table 4). The most likely explanation could be that, despite the overall difference in species richness and abundance, the actual user preference of species is not exclusively determined by abundance of species. For instance, the Vhavenda communities in MFR and TVFR still share five highly preferred species out of the top ten most useful tree species found in the study area (Table 5 and 6). These are *Parinari curatellifolia, Englerophytum magalismontanum, Syzygium cordatum, Combretum molle R.Br. ex G.Don,* and *Mimusops obovata.* This implies that through traditional knowledge, there is convergence on the species preference of local people for actual use through the different social process. These processes include knowledge sharing through oral tradition, clan gathering, initiation schools and apprenticeship by traditional healers (Constant and Tshisikhawe 2018). Over time, this kind of knowledge intertwines into their cultural and symbolic identities of people (cultural keystone species) (Gaoue et al. 2017). Hence, the keystone species do persist as preferred species, regardless of their abundance. For instance, despite *Parinari curatellifolia* and *Englerophytum magalismontanum* having lower abundance in TVFR than in MFR (Table 2), the two species are still cited as the most important. The same popularity was also observed for *Combretum molle* and *Mimusops obovata*- which are rare species in both forest reserves (Table 4 and 5). However, this does not imply that people solely rely on keystone species for their survival. Local people still use other potentially useful species depending on the local ecological conditions (Macía 2008) (e.g., seasonal availability). Equally, even if, some species are abundantly available for potential uses, actual use can still be constrained by species-specific taboos (Constant and Tshisikhawe 2018). Hence, the similarity of results on the correlation of species abundance and the use-value a species between the two Forest reserves is not surprising.

There was a very weak to moderate correlation (r_s= 0.30-0.50) of the local abundance of a species with use-value, both for overall and specific utility species; both in MFR and TVFR. The only exception was that there was no correlation between the abundance of species with the use-value for grazing and browsing, wild food and medicine in TVFR. Similar to the findings of this study, many studies (e.g., Gonçalves et al. 2016) have been reported on the inconsistent power of ecological abundance to explain human user-behavior to overall forest utilities or specific utilities. In some recent study (e.g., Soares et al. 2017), it has been shown that ecological abundance does not explain use-behavior at all.

4.5. Conclusion

The response to the questions of whether local people consciously manage forests and tree species diversity in their landscape relies on the understanding of how culture and ecological abundance influences resource use-behavior. Based on the findings, at least in the case of the Vhavenda people, culture plays a predominant role to explain use-behavior. Abundance may play a secondary role subject to cultural context. These findings have serious implication on designing conservation intervention that works both for people and biodiversity.

Unlike exclusionary protected area approach to preserve a particular biodiversity hotspot, traditional society manages the sustainability of local biodiversity as a socio-ecological system, on which their livelihood, cultural, emotional, spiritual and symbolic values of their lifestyle depends. Neither land use nor resource use of forest landscape is random decisions nor is the concept of protected areas a new concept to traditional society. Traditional society applies dynamic and adaptive socio-ecological knowledge systems (belief, knowledge, and practices) of a particular habitat or species as an integral part of managing the delicate balance of "use-protection" regimes at a landscape level. The protection of sacred forests and habitats and species-related taboos are typical examples of how traditional society still consciously manage landscape for the multifunctional purpose. Hence, the adherence to social norms and taboos combined, with the resilience of traditional socio-ecological knowledge in human modified landscape, presents potential tools to complement other global biodiversity conservation efforts.

While the presence of higher compliance rate to social norms or lower use-value of cultural protected area acknowledges local people awareness on the need of protection of areas can be useful, it also implies that conventional (state) protected indigenous forests can benefit by integrating those values. Global biodiversity conservation efforts can capitalize on the benefits of the cultural assets of local people through genuine partnership and empowerment, can a significant role in averting a biodiversity crisis.

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CHAPTER 5

Nurturing forest resources in the Vhavenda community, South Africa: factors influencing non-compliance behavior of local people to state conservation rules

Abstract

This comparative study analysed the compliance behaviour of local communities towards culturally (sacred) protected areas and state-protected indigenous forests. Interviews were conducted with 135 households from four villages. Data analysis was done using non-parametric tests. All villages did not significantly differ in the proportion of compliance behaviour in cultural protected areas (χ^2 (3) =5.21; p=0.15) and state-protected indigenous forests (χ^2 (3) =5.10; p=0.16). The majority of local people (76%) were compliers to the cultural protected areas. In contrast, the majority (73%) were non-compliers to the state-protected indigenous forests. Villages did not significantly differ in the distribution of perceived local values (χ^2 = 6.77; df=3; p=0.08). On average, an individual holds about four perceived local values ($\bar{X} = 3.89$, SD=2.35) out of the seven (7) perceived local values. The proportion of people who were motived to protect forest and trees of outstanding utility values (79%), watershed protection (73%), and cultural values (61%), endangered species (n=58%), and wildlife habitat (51%). About 50% of local people perceived that the decline of state-protected indigenous forest cover. About 90% showed a willingness to take part in conservation. There was no consensus on the strength of enforcement of rules over the last 20 years (χ^2(1) =0.67; p=0.796). The regression model (χ^2=50.304; df=7, p=0.00, R^2 =45.5%) revealed that individuals who held fewer perceived local values, perceived negative impact and weaker rules were mostly likely to be non-compliers. People's experiential knowledge might have triggered intention for conservation but did not influence compliance behaviour. We suggest that state-protected indigenous forests become more vulnerable to non-compliance when the necessity of resources for rural livelihood arise due to misalignment of the rules to local values, social norms, and taboos. This can be addressed through inclusive policy reviews to reconcile diverse values and norms and through genuine co-management arrangement.

Keywords: culturally protected area, state-protected indigenous forests, non-compliance behaviour, perceived local values, experiential ecological knowledge, and perceived legitimacy

5.1. Introduction

With the growing global concern of biodiversity loss, nation-states in many parts of the world have been protecting the few remaining intact indigenous forests and biodiversity hotspots (Oldekop et al. 2016, Watson et al. 2014) as a leading strategy through strict conservation rules (Bawa et al. 2011). However, the enforcement (monitoring and sanctioning) of strict protection rules in state-protected indigenous forests (e.g., restriction of access for fuelwood) by forest guards and legal systems have proven to be ineffective (Stern 2008), expensive and antagonistic (Lele et al. 2010, Wilshusen et al. 2002). It has also not achieved the objective of protection of biodiversity fully (Oldekop et al. 2015). Instead, non-compliance behaviour to conservation rules is becoming a universal challenge (Arias 2015, Solomon et al. 2015, Gavin et al. 2010, Robbins et al. 2006).

In contrast, similar to state-protected indigenous forest protection, local people have been voluntarily setting aside part of the landscape for protection of cultural and symbolic significant areas such as sacred forests, species and ecosystem related taboos. In many places where such practices still exist, non-compliance has not been a significant challenge (Ruiz-Mallen and Corbenra 2013, Ormsby and Bhagwat 2010, Colding and Folke 1997). The central question is that if local people have positive biodiversity behaviour, why is it then that non-compliance is becoming ubiquitous in most protected areas around the world (Arias 2015, Solomon et al. 2015)? Higher compliance behaviour does not necessarily mean that all local people respect culture measures. Also, this does not necessarily imply that all members of local people hold Biocentric value orientation or have an underlying belief system in favor of protecting nature for non-human species (Turaga et al. 2010).

Individuals can hold anthropocentric value (i.e., self-centred egoist or human-centred social altruistic) towards their surrounding nature (Ives and Kendal 2014) and still have pro-environmental norms and behaviour (e.g., Schultz 2011). However, the motives for pro-environmental behaviour can be different from those who held Biocentric values. According to value-belief-norm theory, when individuals perceive or have an experiential knowledge of adverse environmental conditions that threaten their egoistic value (e.g., family food security and cultural values), they act to protect or mitigate the impact of environmental conditions on biological resources (Turaga et al. 2010; Ives and Kendal 2014).

Nonetheless, most studies have attempted to predict the actual pro-biodiversity behaviour based on the link between underlying value orientations, attitude and behavioural intentions (Chan et al. 2016). Unsurprisingly, the link between value, attitude, behavioural intention, and actual behaviour has often proven to be weak (Stern 2000, Ives and Kendal 2014). This implies that the underlying value of individuals may represent ideal models but cannot fully capture the socio-cultural and environmental realities that affect the daily livelihood of local people (Infield et al. 2018).

Instead, social norms and taboos are believed to exert a strong influence on individuals' pro-biodiversity action (Gifford and Nilsson 2014). Social norms and taboos are informal institutions or traditional rules that govern human interaction among each other, and with their surrounding environment (Jones et al. 2007). Also, local people establish governing structures that enforce social norms and taboos through different mechanisms (e.g. ceremonies and rituals, sanctions) (Colding and Folke 1997, 2001). Many studies claim that local structures have higher legitimacy and effectiveness in resolving value conflicts and in sanctioning appropriate penalties to non-compliers (e.g., Ruiz-Mallén and Corbera 2013,Kideghesho 2009) than power structures imposed from outside (Infield et al. 2018). Hence, compliance behaviour to traditional rules is quasi-volunteer, and enforcement is most likely to be effective (Hayes and Ostrom 2005).

The above arguments imply that local people make a complex trade-off to assign culturally perceived values about the sustainable use and protection of forest biodiversity depending on their underlying values, experiential knowledge, and traditional rules. In recognition to this, many international agreements (e.g., Convention of Biological Diversity (CBD) advocate for enhancement and integration of culturally perceived local value, knowledge, and practices by nation-states in the biodiversity conservation efforts (Sobrevila 2008). However, there is no consensus in this regard. For instance, Low (1996) and Low and Heinen (2017) stated that our perception of traditional society as being deliberate, cooperative and respectful of their surrounding nature, on which they directly rely for many ecosystem services, is a fallacy. It may thus mislead future conservation strategy. Instead, forest and tree harvesting to sustain rural livelihood are ecologically driven. It is not related to the sacredness of nature.

Similar to the above international literature, there has been a contrasting view of the effectiveness of state-protected indigenous forests and the potential of cultural practices for biodiversity conservation in South Africa. Although all indigenous forests are protected by the National Forest Act (NFA) 1998, as amended in 2005 (Act no. 35 of 2005) of South Africa, enforcement of the act to combat illegal harvesting has been a challenge (DAFF 2011, Shackleton 2009,DAFF 2005). On the contrary, some studies (e.g. Constant and Tshisikhawe 2018, Mutshinylo and Siebert 2010, Khorombi 2001) claim that the Vhavenda rural communities in the Limpopo province of South Africa, are still endowed with rich social norms and taboos including culturally protected areas (sacred forests) that are compatible with biodiversity conservation objectives. The culturally protected areas are under the custody of traditional leaders. However, to the best of my knowledge, neither the compliance behaviour nor the influence of perceived local values, knowledge and legitimacy of state conservation rules are sufficiently studied. Hence, this knowledge gap motivated this study.

The principal objective of this study was to understand the influence of value, experiential knowledge and perceived legitimacy of rules to non-compliance behaviour. The findings of this study will provide an insight on designing policy that will promote work both people and tree species diversity in human dominated landscape. To that effect, the study tested the following hypothesis:

H1: Preferential compliance behaviour: there is a difference in the proportion of local people’s non-compliance behaviour between Culturally Protected Forest Areas (CPA) and State-protected Indigenous Forests (SIF);

H2: Compliance behaviour is positively influenced by locally perceived values, experiential knowledge and perceived legitimacy of conservation of local people.

5.2. Research methods

5.2.1. Study area

As the previous chapters (2-4), this study was conducted in the Vhembe district in the north part of the Limpopo Province of South Africa (Figure 1). However, relevant additional information is furnished to provide a clear picture of traditional forest management practices and the different governing structures of the study area.

The indigenous forests of the Eastern Soutpansberg are fragments which are mostly embedded within the commercial forest plantations (DAFF 2011). Since the establishment of the commercial plantations, after the forced removal of local people in 1949 in Thathe Vondo (von Maltitz and Shackleton 2004) and 1979 in Mafhela (Kabanda and Munyati 2010), the State-protected the remaining indigenous forests for their conservation values. Although the National Forestry Act (NFA) of 1998 prohibits cutting and disturbing any forest or receiving forest products from protected areas without the necessary authorization (DAFF 2011), forest and tree species in VBR have been facing illegal forest activities (Tshisikhawe et al. 2016, Evans 2016).

On the contrary, Vhavenda communities, who reside within close proximity to the nearest commercial plantations, have been reportedly managing the remaining indigenous forests and tree species in their vicinities, as an integral part of their rural-based lifestyle for their economic, cultural, emotional, spiritual and symbolic values (Magwede and van Wyk 2018; Mutshinyalo and Siebert, 2010; Khorombi 2001). The Vhavenda communities have the highest population (69 %) of all the ethnic groups residing in the VBR.

Magwede and van Wyk (2018) reported that Vhavenda rural communities rely on about 189 trees and shrubs species for various products to sustain their rural livelihood. These include fuelwood, construction materials, livestock grazing and browsing, wild food, traditional medicines, and other utilities. They also have taboos concerning specific components of forest ecosystems (e.g., Streams and rivers) (Mutshinyalo and Siebert 2010) and tree species (Constant and Tshisikhawe 2018) for their ritual, mythical beliefs, and totemic values. Many of the communities are well endowed with culturally protected areas (Sacred forests) that have been used as a burial ground for generations of royal families, a place of worship of their ancestors, and as the centre for rituals and traditional ceremonies. Sikhitha (1999) reported that culturally protected areas are still highly regarded by local people for their symbolic and spiritual significance. The traditional leaders, senior chiefs and their juniors, are responsible for overall natural resource management issues including dispute resolution and enforcement of social norms and taboos. However, the perceived local value of forest resources, indigenous knowledge and the legitimacy of traditional rules have been reportedly declining over time (Sikhitha 1999; Constant and Tshisikhawe 2018).

In this case study, two Forest Reserves, Thathe Vondo Forest Reserve (TVFR) and Mafhela Forest Reserve (MFR), were chosen purposefully due to the presence of culturally protected forest areas (CPAs) and state-protected indigenous forests (SIFs) in their respective landscape. The two Forest reserves are located in Thulamela Municipality. The Municipality is occupied almost entirely by Vhavenda communities of which about 90% of them are rural dwellers (Ahunamure 2016).

5.2.2. Sampling and data collection

5.2.2.1. Sampling framework

In this study, a sequentially mixed sampling technique was used to select villages and sample population (Onwuegbuzie and Collins 2007). First, four villages from Thatho Vondo Forest Reserve (TVFR) and Mafhela Forest Reserve (MFR) were purposefully chosen as these villages were located within the perimeter of the Forest Reserves. Tshidzive and Tshilungwi in Thathe Vondo Forest Reserve have 312 and 253 households respectively; while Belemu and Tshiema in Mafhela Forest Reserve have 99 and 113 households, respectively (see Figure 16 above).

This was then followed by determining the sample size of respondents (n) required out of the total population (N=770) using Watson (2001) equation as stated below:

$$n = \frac{\frac{P(1-P)}{\frac{A^2}{Z^2}+\frac{P(1-P)}{N}}}{R} \qquad (1)$$

Where n=sample size required, N =number of people in the population, P=estimated variance of a population, as a decimal, A=Precision desired, as expressed in decimal, Z=Based on confidence level, R=Estimated response rate, as a decimal. Consequently, the required sample size (n) was calculated with an estimated variance in population (P) of 30 %, estimated precision of 5 %, and confidence level of 95% (Z=1.96) and estimated response rate of 95%. Accordingly, the required sample size was approximately 78 households representing a sampling intensity (n/N) of 10.13 %. However, this was increased to 20% when 135 households were interviewed thereby decreasing the sampling error and increasing the reliability of the sample statistic to estimate the population parameter (Bhattacherjee 2012). The actual households for the interviews were selected using systematic random sampling, constituting every 5th household, from the list of residents provided by local chiefs. Where interviews with household heads were not possible, a person above 21 years, available in the household during the interview, was considered with their consent.

5.2.2.2. Household interviews

First, based on a preliminary analysis of the relevant literature and open discussion with officials from the Indigenous Forest Management Division of the Limpopo provincial government, a semi-structured questionnaire was developed. The questionnaire was used to interview six (6) traditional leaders on the history of the villages, establishment, and management of trees, conservation of natural forests, traditional rules and their responsibility. Also, general discussion on forest affairs of the surrounding communities was held with four elderly men and two women (one with traditional healing knowledge). A structured questionnaire for household interviews was then developed based on that information.

The questionnaire was then translated into the local language and pretested on six households in Belemu. On average, the interviews took approximately 1-1.5 hours per household. Various socio-economic data were collected from households. These included household characteristics, perceived local values and forest and tree product harvesting on both protected areas, perceived impacts of forest product harvesting on forest conditions, a willingness to take part in conservation initiatives and perceived strength of conservation rules over the last 20 years.

5.2.2.3. Data analysis

Following Arias (2015), compliers and non-compliers were categorized based on the results of direct questions on the behaviour of tree product harvesting. The use of firewood, timber for construction, wild foods, herbal medicine and grazing and browsing from CPAs and SIFs were denoted as 1 (yes) and 0 (No). A summated score of use of tree products for household livelihoods from SIFs and CPAs was then made separately. Both scores were then transformed into complier (0), for those who do not use any forest product from each protected area and non-complier (1), for those who breached the rule for at least one product. The indirect method of categorizing compliers and non-compliers from direct insensitive questions was chosen as appropriate due to the moral, socio-cultural and legally sensitive of non-compliance behaviour.

i. Compliance behaviour towards Cultural Protected Area and State-protected indigenous forests

Chi-square test of homogeneity (p=0.05) was used to determine if the proportional distribution of compliance behaviour across all villages were the same. The test was done separately for CPA and SIF. This was then followed by a Chi-square test of independence to determine if local people demonstrate the same compliance behaviour towards CPA and SIF.

ii. Drivers of non-compliance behaviour to state-protected indigenous forests

Three explanatory variables were measured to determine the drivers of non-compliance behaviour. These are perceived local values, perceived impact of forest and tree product harvesting for rural livelihood and willingness in conservation, and perceived strength of rules.

Perceived local values

Households were asked to answer yes (1) or no (0) to sets of questions developed which were based on the outcome of key informant interviews and previous local studies by Sikhitha et al. (1999) and Khorombi (2001) on the motives why local people conserve forest and tree species diversity. The following were the motives: the need for conservation of endangered species from over-exploitation, protection of useful tree species for livelihoods, watershed protection (soil conservation, protection of water sources from siltation (e.g. dams and lakes)), protection of cultural values (spirituality, identity and cultural heritage), protection of habitats for wildlife (mammals, birds and insects), recreation and landscape beautification, and tourism.

The internal consistency of the data was examined using Cronbach's alpha reliability coefficient (α=0.814), which was relatively higher than the minimum required for a reliability test (Morgan et al. 2012). A summated score of values per household, ranging from 1-7, was then developed as a measure of perceived local value. Mean, and standard deviation of perceived local value was calculated then from the summed scores. To determine if the distribution of perceived local values are the same across all villages, we used the Kruskal Wallis test. Kruskal Wallis test determines whether more than two independent samples came from the same population and that the variable values (summated scores) did not conform to a normal distribution. Cochran Q test was used to analyze if there was a significant difference in the proportion of local people across the different categories of perceived local value.

Frequency table coupled with the McNemar test was then used for pairwise analysis to determine where the significant difference in the proportion of local people across the perceived local value lies. The results of the above three tests were considered significant at p=0.05.

Experiential knowledge and pro-conservation intention

The perceived impact of forest product harvesting from SIFs on tree species diversity for rural livelihood was assessed as decreasing (1), no change (2) and increasing (3) in tree species diversity over the last 20 years. We tested the perceived impacts using a Chi-square test (p=0.05). Their willingness to take part in conservation initiatives in their landscape was assessed as strongly willing (1), and not willing (2) and undecided (3). Their willingness for reinforcement of the rules in SIFs was assessed as: yes (1) and no (0). Frequency (%) and binomial tests were then used to check if the mean proportion of people who showed a strong willingness to take part in conservation initiatives were higher than the unwilling/undecided ones.

Perceived legitimacy of rules and enforcing agency

The assessment of their perception of change in the strength of state conservation rules was coded as weak as before (1) and getting stricter (2) over the last 20 years. Frequency (%) and a Chi-square test were then used to check if illegal tree product harvesting is independent of their perceived strength of rules. Further information was gathered from local chiefs and key informants using a semi-structured questionnaire.

The influence of value-knowledge- rules in non-compliance behaviour

To examine the factors influencing non-compliance behaviour towards SIFs, a binary logistic regression was fitted where the non-compliance group was regressed against local perceived values, experiential knowledge on the impacts of forest product harvesting for their livelihood and willingness to take part in conservation, and perceived trends of state rules at the 0.05 level of statistical significance. The logistic regression equation is as follows:

$$f(Z) = 1/(1+e^{-z}) \qquad (3)$$

The empirical model of the influence of explanatory variables on the non-compliance behaviour to SIF rules is represented in a linear relationship as follows:

$$Z = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \beta_3 X_3 + \beta_4 X_4 + \beta_5 X_5 \qquad (4)$$

Where:

Z represents the probability of households to be non-compliers of SIF rules. The value Z is 1 if the household is a non-complier and 0 if a complier. X_1 represents perceived local values of forest and tree species diversity. X_2 represents the perceived impact of forest product harvesting for their livelihood on tree species diversity within SIFs (1 = declining; 2 = no change; 3 = increasing). X_3 represents a willingness to take part in conservation initiatives (1 = strongly willing, 2 = not willing, 3= undecided). X_4 represents a willingness to strengthen protection of SIFs (1 = willing; 2 = not willing). X_5 represents the perceived strength of rules (1 = as weak as before and 2 = getting stricter). Logistic regression coefficient (B), Standard error (SE), Wald Chi-square, p-values and odds ratio (exp (b)) were used to report the findings.

The data analysis in this study was done using the Statistical Package for Social Sciences (SPSS) version 20 (SPSS 2011).

5.3. Results

5.3.1. Compliance behaviour towards Cultural Protected Area and State-protected indigenous forests

The chi-square statistical test on the homogeneity of proportional distribution of compliance behaviour across villages was not significantly different both in CPA ($\chi2$ (3) =5.21; p=0.15) and SIF ($\chi2$ (3) =5.10; p=0.16). About 76% (n=102) of the local people in the study area were compliers to the CPA rules. In contrast, about 73% (n=98) of the local people were non-compliers to the SIF (Table 1). The chi-square test result showed that there was not a significant association in the compliance behaviour between CPA and SIF (χ^2 (1) =3.29; p=0.07).

5.3.2. Perceived local values

The mean and standard deviation of perceived local value that motivates an individual to conserve forest and tree species in their surrounding landscape was 3.89 and 2.35, respectively. The Kruskal-Wallis test did not find any significant difference in the distribution among all villages ($\chi2$ = 6.77; df=3; p=0.08). A Cochran Q test determined there was a significant difference in the proportion of local people, who held different motives (perceived local value items) to conserve and protect forest and tree species in their landscape (χ^2 = 129.03; df=6;

p=0.00). Table 5.1 shows that people attached the highest motive for the conservation of forest and tree species to protect trees of outstanding utility values (79%). This was then followed by the motivation for protection of watershed (73%), cultural values (61%)), protection of endangered species (58%), and habitat for wildlife (51%), tourism (39%) and recreation and landscape beautification (35.6%) in descending order. However, the McNemar test result showed there was no significant difference in the proportion of people who were motivated to conserve and protect trees of outstanding utility value from overexploitation and those who were motived to protect trees for watershed protection. Similar, there was no significant difference between the proportion of people who were motivated to protection of forest and tree species diversity for culture reasons and wild habitat, and between conservation for tourism and recreation and landscape beautification.

Table 5. 1 Summary of McNemar test between pairs of motives of local people for conservation of forest tree species

Values items and no. of respondent	Utility species	Watershed protection	Endangered species	Wildlife habitat	Culture	tourism
Utility species (n=107)						
Watershed protection (n= 98)	χ^2=1.16, p=0.28					
Endangered species(n= 78)	χ^2=20.10, p=0.00	χ^2=11.28, p=0.00				
Wild habitat(n= 68)	χ^2=35.52, p=0.00	χ^2=22.13, p=0.00	χ^2=4.05, p=0.04			
Culture (n= 81)	χ^2=11.61, p=0.00	χ^2=8.25, p=0.00	χ^2=0.15, p=0.70	χ^2=4.11, p=0.02		
Tourism (n=53)	χ^2=35.22, p=0.00	χ^2=36.52, p=0.00	χ^2=11.29, p=0.00	χ^2=4.56, p=0.03	χ^2=15.18, p=0.04	
Recreation and beatification(n=48)	χ^2=47.38, p=0.00	χ^2=41.39, p=0.00	χ^2=17.52, p=0.00	χ^2=9.50, p=0.00	$\chi2$=19.32, p=0.00	χ^2=0.84, p=0.35

χ^2 = Chi-square value for Cochran Q test, p= P-value

5.3.3. Experiential knowledge and pro-conservation intention

The chi-square statistical test on perceived impacts of forest product harvesting for rural livelihood was found to be statistically significant (χ^2 (2) =21.1; p=0.000). The observed proportion of people who perceived that the tree species decline over the last 20 years (50 %) was higher than expected, while the proportion of people who did not perceive any change (24 %) and those who perceived an increase in tree species diversity (26 %) was lower than expected.

In terms of their conservation intention, the binomial test found that the observed proportion of those who were more willing to take part in conservation (90%) and those who supported the reinforcement for protection of SIFs (70%) was higher than expected (p<0.00).

5.3.4. Perceived legitimacy

The chi-square test result found that the proportion of people who perceived that the strength of state rules had remained weak over the last 20 years (51%) was not different from those who perceived it as getting stricter (49%) at (χ^2(1) =0.67; p=0.796).

5.3.5. Factors affecting compliance behaviour

The impact of different factors affecting compliance behaviour was evaluated using a binary logistic regression model. When the full model was tested against the intercept-only model, it was statistically significant (χ^2=50.304; df=7; p=0.00) with Nagelkerke R^2 of 45.5 %. Table 2 shows the logistic regression coefficient, Wald test, and odds ratios for each of the explanatory variables. The factors with a statistically significant influence on compliance behavior were the perceived local values, perceived impacts of forest product harvesting, and perceived strength of rules. For every unit decrease in the perceived local values held by an individual, the individual would be 0.7 times more likely to be a non-complier than to a complier to state conservation rules. Perceived impacts of forest product harvesting for rural livelihood on tree species diversity had a significant influence on compliance behaviour (Wald=20.1; df=2, p=0.00). Surprisingly, the odds of those who perceived the decline of forest cover over the last

20 years due to forest product harvesting to be a non-complier was higher by 12.5 times than to be a complier. The odds of those who perceived the rules had remained weak (illegitimate) to be non-complier was higher by 4.8 times than to be a complier.

Table 5. 2 The influence of value, knowledge, and rules in non-compliance behaviour

Parameter	B	SE	Wald	Df	P	Exp(B)
Perceived local values	-0.339	0.127	7.081	1	0.008	0.712
Perceived impact (cat.)			19.986	2	0.000	
Declined	2.524	0.687	13.507	1	0.000	12.479
No change	-0.948	0.653	2.108	1	0.147	0.388
Overall willingness to conservation (cat.)			2.738	2	0.254	
Strongly willing	1.547	0.951	2.648	1	0.147	4.699
Not willing	1.926	1.978	0.949	1	0.330	6.865
Those who showed willingness to strengthen protection of SIF (cat.)	-0.833	0.635	1.723	1	0.189	0.435
Those who perceived the enforcement state conservation rules remained over the last 20 years (cat.)	1.565	0.597	6.868	1	0.009	4.784

B= Logistic regression coefficient , SE= Standard error , Wald=Wald Chi-square, p= P-values, exp (b)=odds ratio

5.4. Discussion

Managing compliance behaviour to rules is at the centre of successful natural resource management and biodiversity conservation initiatives (Stern 2008; Gavin et al. 2010). It demands continuous monitoring and sanctioning of appropriate penalties when non-compliance behaviour is detected (Gibson et al. 2005; Stern 2008). However, both detecting non-compliance (Gavin et al. 2010; Arias 2015) and sanctioning penalties (Solomon et al.

2015) have proven to be difficult in the past. As such, non-compliance behaviour has been affecting many protected areas and endangered species. Biologically, it causes erosion of genetic diversity, species richness, composition, and the decline of ecosystem services. Socially, non-compliance has repercussions on the resources that are vital for the maintenance and sustainability of livelihood of the rural poor (Arias et al. 2015, Solomon et al. 2015, and Gavin et al. 2010). Hence, understanding what influences compliance behaviour is crucial, as compliance with conservation rules is desirable pro-biodiversity behaviour. However, understanding where and why people break the rules are also equally important. It assists in contextualizing the problem and design viable policy interventions to protect biodiversity and natural resources (Arias 2015).

Based on the findings of this study, the higher compliance rate to culturally protected areas (Sacred forests) and higher non-compliance rate to state-protected indigenous forests, the preferential compliance behaviour of local people towards different conservation rules are presented. Also, the influence of perceived local values, perceived/experiential knowledge of local people on the impacts of forest and tree species harvesting on tree species diversity, and perceived legitimacy and strength of rules to non-compliance behaviour towards state-protected indigenous forests are discussed.

5.4.1. Compliance behaviour towards Cultural Protected Areas and State-protected Indigenous Forests

In the case of culturally protected forest areas, the findings from this study attest that compliance is a social norm where the majority of individuals adhere to the traditional rules across all villages. This is consistent with the findings of Sikhitha (1999), who reported that culturally protected areas are highly regarded for their symbolic and spiritual significance in the Vhavenda communities. For instance, the Thathe Vondo sacred forest is the burial site of at least six chiefs of the Tshidzivhe tribe. It is also a place of worship and traditional ceremonies (Sikhitha 1999). Sacred forest is also vital as a shelter for baboons that have totemic values to the clan residing in Mafhela Forest Reserve (NeLwamondo, the local chief, pers. comm. 2015). Also, Mutshinyalo and Siebert (2010) indicated that certain species and components of forest ecosystems (e.g. Streams and rivers) are still culturally protected due to the rituals, mythical beliefs, and totems. These claims are consistent with findings in Indian Sacred Forests (Ormsby and Bhagwat 2010) and the role of “resource and habitat taboos” in many parts of the world

(Gómez-Baggethun 2013, Kideghesho 2009, Colding and Folke 2001). Moreover, the presence of local chiefs, who reside within communities, makes enforcement of traditional rules easy and timely when non-compliance is detected (S Nephiphidi, retired forestry official, Pers. Comm 2015). However, the compliance behaviour of local people towards culturally protected areas does not have any association with their behaviour towards the state-protected indigenous forests.

Instead, in contrast to culturally protected areas, as in many parts of the developing world (Arias 2015; Solomon et al. 2015), this case study supports that non-compliance behaviour with rules that govern the state-protected indigenous forests is a challenge. The majority of local people, who complied with cultural protected areas, were non-complaint to state-protected indigenous forests. The high non-compliance rate to the state-protected indigenous forest has been an ongoing concern in South Africa (e.g. Vermeulen et al. 2019, Shackleton 2009). The legal designation of an area alone is not enough to protect forest and tree species diversity (Hayes and Ostrom 2005). The central question is then: why do those individuals, who comply with rules that govern culturally protected forest areas, fail to adhere to the rules of state-protected indigenous forests?

5.4.2. Perceived local values and non-compliance behaviour

Similar to the homogeneity of compliance behaviour, all villages demonstrated the same pattern of perceived local values. On average, an individual holds about four out of the seven local perceived values that motivated local people to conserve forest and tree species in their landscape. This implies that individuals can hold a multitude of values simultaneously in their desire to conserve forest and tree species diversity in their multifunctional landscape. It is unlikely that their value orientations will be strictly anthropocentric (egoistic or socio-altruistic) or biocentric (Chan et al. 2016, Bengston 1994). Instead, individuals hold a continuum of both value orientations that might appear to be diverse and conflicting. Such value orientation in traditional communities is not surprising considering the historical evidence of the Vhavenda traditional forest and tree species management practice.

The majority of individuals were motivated to protect forest and tree species with outstanding use-value for rural livelihood, protection of watershed from erosion and siltation of water points (e.g. rivers, dams and lake), protection of trees for cultural values, endangered species and wildlife habitat in descending order. The result was expected, considering that the concern of the majority of local people around the study area on the negative impact of forest and tree product harvesting on forest cover (Mabasa and Makhubele 2016). Local people in Vhembe district relies on forest and tree products for more than 80% of their energy demands for cooking and heating. They also use forest and tree products as a source of construction material, wild food, and livestock grazing and browsing, and traditional medicine (Magwede and van Wyk 2018, Makhado et al. 2009). Equally, Khorombi (2012) stated that slash and burn cropping practices on the mountainous areas and clearing of riverine forests for agricultural purpose as one of the concerns for soil erosion and siltation of water points. Hence, consistent with value-belief-norm theory, such perception can trigger personal values and norms towards either to avoid or mitigate the negative impacts of forest harvesting on their food security and other non-timber forest products including their cultural identity (e.g. Ives and Kendal 2014, Schultz 2011). The majority of local people in Vhembe Biosphere Reserve have interest in better use and management of forest resources as a mitigation strategy for their sustainable livelihoods, protection from soil erosion, and improve the quality and availability of water sources (Ofoebgu et al. 2016).

Individuals may consider the value of forest and tree species conservation for the common good beyond maximizing individual benefits. For instance, the same individuals who appeared to hold egoistic values were also holding altruistic social values (watershed protection and cultural values) and biocentric values simultaneously (e.g. protection of rare/endangered species and wildlife habitat). The prevalence of better culturally protected indigenous forests, along with the multitude and diverse individual values, hints that it is not only biocentric values that motivate individuals to conserve biodiversity. Still, egoistic and social altruistic motives can lead to a path of sustainable biodiversity (De Dominicis et al. 2017). While individuals, as individuals and as a member of a Vhavenda community, have been relating with their surroundings to fulfil their livelihood demands, they have also been taking into account the protection of other values of nature in their pursuit of making sense of life in its entirety; culturally, emotionally and spiritually (Khorombi 2001; Mutshinyalo and Siebert 2010).

The number of perceived local values held by individuals has an influence on compliance behaviour towards the rules that govern state-protected indigenous forests. Individuals with lower perceived values were most likely to be non-complying to these rules.

Conversely, people with higher perceived local values to the overall landscape appeared to have positive compliance behaviour. However, the dominance of non-compliers to state-protected indigenous forest rules, in the presence of higher compliance to culturally protected areas in the same landscape, might be an indication of the difference in local people's relationship and sense of place between the two protected areas (Masterson et al. 2017). For instance, people can even hold negative values and views towards parts of their place that may influence both actual individual and social behaviour (Brown and Raymond 2007; Masterson et al. 2017). Given Sikhitha's (1999) evidence of resentment towards the history forceful displacement local people from their place, in pursuit of commercial plantations on which state-protected indigenous patches are embedded, it was expected that local people might attach relatively negative social values with SIFs. Hence, that might have made local people's decision relatively easier to breach the rules that govern state-protected indigenous forests over cultural protected areas when the need for forest and tree products to sustain their livelihood arises.

5.4.3. Experiential knowledge, pro-conservation intentions, and non-compliance behaviour

Half of the local people perceive the negative impact of forest and tree product harvesting for livelihood activities on tree species in the state-protected indigenous forests. However, regardless of their perception, almost all of members of local community showed a willingness to take part in conservation initiatives of the whole landscape. Also, the majority of local people supported the strengthening of protection and enforcement of rules that govern state-protected indigenous forest. In support of Newell et al. (2014), the dominance of the positive perception on the impacts of forest and tree species harvest on the dwindling of tree species diversity within state-protected indigenous forests might have triggered the stronger willingness for conservation and reinforcement of protection of SIFs. However, the willingness of the majority of local people in the study area did not seem to be influencing their actual compliance behaviour positively.

Contrary to the expectations, those who perceived the decline in tree species diversity due to over-exploitation of forest and tree products for rural livelihoods were those who were most likely to be non-compliers. Better experience and judgment by the majority on forest conditions and tree species diversity, due to their day-to-day interactions with forest and tree species in SIFs in pursuit of their livelihoods, alone did not explain their action or actual compliance behaviour. Similar to the findings, there is mounting evidence on the gap between environmental awareness and actual behaviour (Kollmuss and Agyeman 2002).

Consistent with the value-norm-belief theory, the experiential knowledge on the decline of tree species diversity might have triggered environmental concern, and good intention to take part in overall conservation and reinforcement of rules governing state-protected indigenous forests. Similar findings on the attitude and intention of local people of the Vhembe Biosphere Reserve and the support to the enforcement of state indigenous forest protection were reported by Sikhitha (1999), and Ofoebgu and Ifejika (2017) around the same study area.

Ofoebgu and Ifejika (2017) argue that social norms are the lead predictor for the good intention to participate in prospective conservation in the Vhembe Biosphere Reserve. While this assertion may be true, the actual pro-environmental behaviour (e.g. Compliance to state-protected indigenous forest and tree planting) may not be directly influenced by social norms and taboos in the same way as in the culturally protected forest areas. The contrasting level of compliance behaviour between culturally protected areas and state-protected indigenous forests hints at the fact that local context (e.g. poverty, unemployment, alternative livelihoods) may force local people to make a choice, out of livelihood necessity. It was expected that local people would respect social norms which are embedded in their meaning and sense of place and identity (Masterson et al. 2017) over state-protected indigenous forests. Also, there is mounting evidence that personal value and norm are shaped by what the individual perceives on how others in society act (descriptive norms), and what is actually acceptable social behaviour (e.g. injunctive norms) (Schluter et al. 2017; Beyerl and Breckwood 2016) to a specific place (Masterson et al. 2017).

5.4.4. Perceived legitimacy

This study found that there was no consensus around the perceived legitimacy of the rules that govern state-protected indigenous forests. Instead, local people were divided into two categories equally that hints at the ineffectiveness of current state conservation rules. Lack of consensus on the legitimacy of state rules, as opposed to higher compliance for culturally protected areas as found in this study, may hint at the non-coherence between individual values, social norms, state rules and the legitimacy of rule enforcement agencies (Jackson et al. 2012).

This study found that those who perceived state conservation rules as being weak were those who were most likely to be non-compliers. Some members of local people in Thathe Vondo Forest reserve had resentment from the beginning when their land was taken away for the establishment of commercial plantations without consultation and consideration of cultural values (M Netshidzivhe, the local chief, Pers. comm. 2015). Even after the collapse of the homeland policy and the introduction of a democratic government, there has not been strict enforcement of rules and a clear demarcation of authority among various government departments and local institutions who are responsible for conservation (Shackleton 2009). Hence, it can be argued that the weakness of enforcement may have made it easier for individuals to break the rules of the strictly protected state indigenous forests when the necessity arises to fulfil their livelihood than to breach the culturally protected areas.

Although the study found higher non-compliance rate in state-protected indigenous forests, we cannot infer the impact of non-compliance on the status of biological resources. The study was based on self-reported binary data on users' behaviour, and perception of local people on the impact of forest and tree product harvesting on forest cover. Moreover, despite a broader claim on the negative impact of non-compliance on biodiversity (Gavin et al. 2010; Arias 2015), rule-breaking is a non-random ecological disturbance. It follows a specific pattern across space and time. Depending on the extent of illegal harvesting and the resilience of forest biodiversity, non-compliance may maintain, enhance or degrade biodiversity. Nevertheless, there is a paucity of information on the effect of non-compliance for subsistence use of forest products on biodiversity (Robbins et al. 2006) within the parameter of state-protected indigenous forests. This is not to sanction or endorse non-compliance behaviour as an acceptable practice. Instead, this is to highlight the need for ecological research on the impact of non-compliance on biodiversity within protected areas. Considering non-compliance is likely increasingly becoming a threat for the persistence of many ecosystems and species (Solomon et al. 2015), such empirical evidence may inform appropriate management interventions (e.g. regulated local uses or strict enforcement to ensure recovery).

5.5. Conclusion

The future success of protected forest areas within multifunctional landscape lies mainly on designing policy instruments based on the understanding of factors that affects local people's conservation behaviour. Local people hold diverse values, traditional ecological knowledge, and experience that motivate their conservation interest. Conservation of forest and tree species diversity is still an integral part of rural people, at least in Vhembe Biosphere Reserve. The fact that, on average, an individual holds about four values to conserve tree species diversity in multifunctional landscapes shows that they are neither strictly egoistic, socio-altruistic or biocentric. Forest and tree species diversity management in their landscape is intertwined into the livelihood, cultural, emotional, spiritual and symbolic values of their lifestyle. Simply proclaiming designating protected areas with the presumption that local people are not pro-biodiversity conservation will not bring a sustainable solution for the global biodiversity crisis.

The widespread non-compliance behaviour to state-protected indigenous forest does not appear to be as a result of lack of interest and ignorance of the impact of illegal harvesting. Rather, local people have to make tough choices to sustain their livelihoods against their will in conservation. In such situations, local people breach conservation rules that govern strictly protected state indigenous forest areas out of necessity and limited alternative livelihood options. In practice, the presumption that local people will complain to rule if caught by forest guards and convicted may not be as simple as it appears. Hence, stricter protected areas may not be a feasible option as a standalone strategy for the future.

In actuality, culturally protected forest areas have stricter rules than protected state indigenous forests. Such a discrepancy in the effectiveness can be attributed to the fact that culturally protected areas are more aligned to local perceived values, social norms and taboos and a relatively higher legitimacy of the custodian (local chiefs) than state-protected indigenous forests. Hence, conservation policy review must learn a lesson from culturally protected areas by aligning to local values and promoting social norms and taboos. With strong experiential knowledge, concerns, and willingness towards conservation found by this study, such inclusivity may promote successful and enforceable co-management in conserving forests and tree species diversity. Moreover, improving the conservation status and productivity of other parts of forest landscapes may assist in relieving the effect of livelihoods on forest and tree species diversity while improving their livelihood. Innovative and mixed policies and practices to revive traditional multifunctional landscapes, wherein most state indigenous forests reside, through genuine partnership will assist in reversing the course of the biodiversity crisis.

5.6. References

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CHAPTER 6

6.1. Overview and Conclusion

Despite human society modifying ecological and evolutionary process across the globe for millennia (Roberts et al. 2017; Ellis 2015), the current rate of human disturbance on global ecosystems is alarming. Consequently, the rate of biodiversity loss is proceeding faster than the background rate recorded between the previous mass extinction events (Pimm et al. 2014). Although the extent of human modification at the local scale varies (Ellis 2010), the impact on biodiversity ranges from local to regional and global spatial scales (e.g through climate change) that has adversely affected ecosystem processes and functions (Potapov et al. 2017). In this regard, the accelerated rate of deforestation and fragmentation globally, and in the tropics specifically, contributed a significant share as drivers for global biodiversity loss.

Giam (2017) predicted that if the current rate of tropical forest deforestation and fragmentation proceeds unabated, even without considering other human induced global changes (e.g., climate change, habitat loss in the other ecosystems), it will hasten the sixth mass extinction event in a couple of centuries. However, the distribution of disturbance across spatial scales is not uniform and their impacts on tropical biodiversity are not fully avoidable (Ellis et al. 2013). At the same time, neither do all forests nor species respond to disturbance in the same way. Such a complex picture of tropical forests as socio-ecological systems pose scientific and ethical challenges and opportunities for conservation of tree species diversity.

Despite the increase on the extent of protected area networks, there has been growing scepticism on their effectiveness, efficiency, and sustainability in protecting and restoring biodiverse areas from both ecological and social perspectives (Watson 2016; Oldekop 2016). This has dichotomized many conservationists into those who hold biocentric and anthropocentric value orientations towards biodiversity conservation (Gavin 2018). The proponents of protecting the Half Earth (HE) option (Wilson 2016), with their biocentric orientation, still pursue the expansion of protected areas as a preferred biodiversity conservation measure. They advocate protection of the fragile biodiversity from human threats, regardless of their value to humanity. However, there is growing evidence of increasing non-compliance behaviour of local people to conservation rules that govern protected areas (Solomon et al. 2015; Arias 2015) that will eventually undermine the objective of biodiversity conservation.

In contrast, the proponents of the New Conservation Science (NCS) advocate an alternative option that embraces the maintenance and enhancement of biodiversity for anthropogenic value (e.g. ecosystem services) to the benefit of all people equally (Kareiva and Marvier 2012). NCS proponents claim that nature has a remarkable resilience to human disturbance. At the same time, the presumption that all local people are threats to global biodiversity is an oversimplification of human behaviour. In particular, there has been growing evidence that traditional society has rich, local socio-ecological knowledge (TSEK), which is culturally embedded and where conscious management of their landscapes are done in a biodiversity friendly manner. Hence, inclusion of local people and TSEK in systemic conservation planning and management improves compliance to conservation rules. This has led to NCS proponents to argue that effective management of Human Modified Forest Landscapes (HMFL) (e.g. traditional land use) is more important for biodiversity than protecting undisturbed forest ecosystems (Watson et al. 2016). Yet, there is no consensus on the unlimited resilience of nature (Martin et al. 2012) and the potential of culturally modified landscapes for conservation (Melo et al. 2013).

On top of the contrasting underlying value orientations and divergent policy proposals, the realisation of the growing (i) competing demands and conflicting values for land (e.g. expansion of agriculture, biodiversity conservation) (Ellis et al. 2015), (ii) scarcity of large tracts of undisturbed forest land in the tropics to expand protected areas (Melo et al. 2013), and (iii) a growing limitation of resources for conservation investment (Walls 2018), have triggered research interest to understand the link between culture and nature.

Many studies have been showing the inextricable link between traditional socio-ecological knowledge (TSEK)/culture, nature and pro-environmental behaviour of traditional societies (e.g. Lyver et al. 2019; Paneque-Gálvez et al. 2018, Sutherland et al. 2003). To that effect, Aichi targets (the strategic plan for CBD implementation) included the HMFLs beyond protected areas and integration of TSEK as the two pillars of conservation goals (Díaz et al. 2015). However, there has not been sufficient evidence of the superiority of effective management of human modified landscapes against the effectiveness of the currently protected area networks, using a comparable ecological performance matrix (Watson et al. 2016). At the same time, there is no agreement on the validity of the claim to resuscitate TSEK/culture as a conservation tool for fast-paced global biodiversity loss (Low and Heinen 2017; Low 1996).

Based on the findings of this study, this chapter presents a thesis that hypothesizes that the potential of HMFLs for biodiversity conservation over the effectiveness and efficiency of protected areas depends on the prevalent socio-ecological context. The study was carried out in two Forest reserves (FRs) in Eastern Soutpansberg, Limpopo, South Africa; namely Mafhela (MFR) and Thathe Vondo Forest (TVFR). The two FRs are almost exclusively inhabited by the same cultural group - the Vhavenda ethnic group - who relies on forests and tree species diversity for a multitude of utilities to sustain their rural livelihoods. However, the two FRs differ in the condition of their forest landscape complexity. MFR has a relatively simplified human modified landscapes in contrast to the complex TVFR. Both of them falls under Vhembe Biosphere Reserve.

The next sections provide a synopsis of each chapter's key findings and the methodological rigor used to address the specific objectives from the empirical chapters in a synthesized manner (Chapters 2-6). First, based on the key background findings from the different chapters, a socio-ecological framework was presented to link the key findings of each chapter. The chapter also recommends areas for further research and provides general concluding remarks.

6.2. Key findings and methodological rigor developed for this study

Four independent studies were conducted to address the overall aim, specific objectives and associated hypotheses (Section 1.6). These include: the resilience of tree species diversity under contrasting conditions of forest landscape complexity (Chapter 2); disentangling the conservation values of land use regimes from their overall biodiversity value (Chapter 3); the predominant role of culture in shaping forest and tree species use-behaviour over the ecological abundance (Chapter 4), and factors influencing non-compliance behaviour to conservation rules (Chapter 5). Because each study falls under a different discipline (Bennett et al. 2017; Spalding et al. 2017), a separate research method was developed for each objective. These were then collated into an interdisciplinary research framework (Fig 1.4)

First, the existing land use regimes were classified, in consultation with traditional leaders according to their social perceived disturbance intensity (Chapter 2). The existing tree-based traditional land use regimes of the two FRs were categorized into two major groups: (i) *Human Modified Forest Landscapes* (HMFL) under the custody of traditional authorities that comprises highly disturbed Common Resource Use Zones (CRUZ), intermediately disturbed

Trees Along Streams and Rivers (TATR) and minimally disturbed Culturally Protected forest Areas/sacred forests (CPA) and (ii) *State-protected Indigenous Forests* (SIF). This was then followed by enumerating all tree species, and the recording of topographic data of each sampling point using a hierarchical nested sampling design (Figures 2.2 & 2.3).

The enumeration of local ecological data (tree species identity, abundance, and environmental change drivers) from sampling plots from different land use regimes within the separate HMFLs was crucial to avoid the presumption that all land use regimes have the same disturbance intensity, and are uniformly superimposed over the same conditions of pre-existing forests. Testing the validity of such an assumption assists to reveal the uniqueness of each HMFL's complexity that is often ignored for the sake of simplicity of presenting ecological research outputs (Pickett and Cadenasso et al. 1995). Also, the local socio-economical and ethnobotanical data from 135 households for 2 villages (in MFR) and 2 villages (TVFR) were gathered to determine the forest and tree species use-behaviour, and compliance behaviour to conservation rules across different social hierarchy- such as household, communities and cultural group.

6.2.1. Framing the findings from a socio-ecological systems perspective

This study used land use disturbance intensity as a predictor to understand the inextricable link between biodiversity and culture, since land use is often reciprocally affected by both factors over space and time (Bürgi et al. 2015). By implication, the understanding of the factors that determine (i) the pre-existing spatial distribution of biodiversity such as environmental drivers (e.g. elevation, slopes, and position of the terrain) (Chapter 2) as well as (ii) socio-cultural ingredients (e.g. social norms and taboos) that shape resource use-behaviour at different biological organizations (habitat and species) of a landscape (Chapter 4), is crucial to understanding the impact of land use disturbance on biodiversity. Jointly, ecological and social factors determine the condition of current forest landscape complexity by superimposing various land use regimes of disturbance intensity gradients on the pre-existing forest landscape. This affects the condition of forest cover, ecological connectivity (composition and configuration of a species pool), and successional stages of different patches over the landscape (Arroyo- Rodríguez et al. 2017).

The findings in Chapter 2 show that MFR had relatively lower species diversity, and was dominated by fewer disturbance tolerant tree species than TVFR. The two FRs demonstrated different patterns of correlation between land use gradient and elevation despite both FRs having almost the same mountain range. While land use gradient negatively and very strongly correlated with an elevation in MFR, this correlation was weak in TVFR. However, the correlation of land use intensity gradient with the other environmental change drivers (such as distance, slope gradient, position on the terrain and access) were almost the same in both FRs (Table 2.2). In Chapter 4, the study revealed that the local communities of the two FRs possess the same culture that played a more predominant role in shaping forest and tree species use-behaviour of the two communities than the ecological abundance in their respective landscapes.

Concurrently, both findings imply that the current contrasting conditions of forest landscape complexity of the FRs in Vhembe Biosphere Reserve were mainly associated with the difference in the spatial patterns of land use disturbance gradient along the elevation gradient (Chapter 2, Table 1.2); not because of their cultural difference (Chapter 4) and compliance behaviour to different rules (Chapter 5). In MFR, the spatial clustering of the same land use regimes on the same elevation range made the current condition of forest landscape more simplified or ecologically fragmented. In a contrasting manner, the heterogeneous spatial pattern of land use gradient along the elevation gradient kept the existing forest landscape in a better condition. As expected by the landscape moderated insurance hypothesis, the resilience of local assemblages between the two FRs showed ecological contrasting patterns to land use disturbance.

6.2.2. The resilience of local species diversity under contrasting conditions of Forest landscape complexity

Despite the increasing hope around the conservation potential of HMFLs, there has not been a coherent framework to assess the effect of land use disturbance on species diversity (Resasco et al. 2017). Many studies have been signalling that species richness after human disturbance increases in some places (Zulu et al. 2018; Syampungani 2008) and decreases or remains the same in others (Hillebrand et al. 2018). Through time, local species richness recovers fully after anthropogenic disturbance (Vellend et al. 2017). This appears to support the argument of NCS proponents and may suggest the global biodiversity crisis is overblown (Cardinale et al. 2018). However, the assumption of Vellend et al. 2017, on the recovery of tree species after disturbance defies the fact that human disturbance is different from natural disturbance regimes. Human disturbance induced by traditional society may occur uninterruptedly over time as it is associated with the day to day lifestyle. Secondly, there have been many studies that have shown that species composition among local assemblages of different land use regimes can still change (β – diversity), despite the recovery of species richness (Hillebrand et al. 2018). In Chapter 2, the study investigated the resilience of local assemblages under contrasting landscape complexity.

In MFR, the local assemblages were vulnerable to a higher rate of local extermination and colonization, as expected in a simplified landscape. Hence, all land use regimes in HMFL became distinct from each other (difference in mean β – diversity). As a result, the replacement of one species by another meant that local species richness of each land use type remained the same within HMFL in contravention to the expectation of the Intermediate Disturbance Hypothesis (IDH). On average, HMFL had the same overall mean species richness as SIF due to equal local extermination of species and replacement by another species (Hillebrand et al. 2018). The local contribution of HMFL to overall β – diversity of the FR was higher than the local contribution from SIF. Hence, it was not a surprise that land use gradient was the leading change driver to explain the overall β – diversity of the FR.

In TVFR, the local assemblage of land use regimes within HMFL shared the majority of species among each other, despite the similarity in local assemblage declining along the land use gradient. The conformity of the response of species richness to IDH in the presence of a clear gradient in mean β-diversity shows that species replacement was practically insignificant to override an orderly local extermination/gain gradient of the TVFR. The conformity of species richness response to IDH may hint at the fact that the local assemblage of land use regimes (along a land use intensity gradient) in the HMFL of TVFR are more resilient to land use disturbance. However, resilience does not mean the absence of dynamism. Even in the absence of human disturbance, the local neighbourhood effect, together with biotic and abiotic elements, may still incur small scale changes in species composition (Ghazoul et al. 2015). Overall, the mean species richness and the LCBD of HMFL and SIF were the same. The fact that natural factors, such as elevation, slope gradient and position of the terrain were the leading drivers to explain the overall β-diversity of TVFR shows that human modification did not override the natural gradient of γ-diversity (landscape species pool). Instead, it kept the species diversity in dynamic equilibrium across space in the landscape.

The implication of the above findings is that, unlike the NCS claim of the remarkable resilience of nature, the response of local species diversity to disturbance is complex and non-linear. It depends on the species diversity matrix used, the spatial scales of measurement, and the condition of forest landscape complexity. The overall average local species richness (Margalef index) at landscape scale in both HMFLs of the FR remained the same as their adjacent SIF. This appears to conform to the current global meta-analysis of local species richness in HMFLs that remain the same (Vellend 2017), regardless of the response of the species richness of a particular land use regime subject to the landscape context. However, the resilience of species composition of the local assemblage of all land use regimes may shift from each other as was the case in MFR, or remain resilient at HMFL as was the case in TVFR.

Overall, this study shows that the resilience of local assemblages due to land use pressure can neither be fully explained by one theory nor be captured by one species diversity matrix (e.g., Richness or β-diversity). The conditional conformity of species richness in responding to IDH is not a unique phenomenon in the application of natural laws to advance scientific debates. This is synonymous to the *principle or the theory of least action* that proves, for instance, why a ball thrown from a tower falls freely along the shortest path to the ground following the path with least resistance under the force of gravity. If the ball spirals around in widening loops and bounces back in the air, it is given that there are hidden forces in operation, such as hidden strings or gusts of air (Levin 2017). This does not mean the law of gravity has ceased to work due to a change of conditions. With the same logic, the conditional conformity of species richness to IDH predication is also moderated by the conditions of landscape complexity. That does not imply that IDH is obsolete, as stated by some authors who demand the obliteration of the hypothesis (e.g. Fox 2013). Instead, it implies understanding that the condition of forest landscape complexity is as important as understanding the direct impact of land use gradient on species diversity during systemic conservation planning and management of tree species diversity in HMFLs.

However, using the species richness index alone may statistically obscure the effect of different land use regimes of a larger area of HMFL by averaging their variability in comparison with the relatively small area of SIF. This is mainly due to the effect of area on species richness (Lawton 1999). On top of that, the species richness matrix cannot reflect the effect of land use on the composition of co-existing species identity (β-diversity) at different spatial scales that may be affected by different outcomes of a trade-off between local extermination and colonization. Equally, β-diversity that relies only on presence/absence data, as in the case of Chapter 2, cannot capture the overall facets of biodiversity loss, such as change in the abundance distribution of species (Hillebrand et al. 2018). Hence, assessing the impact of disturbance on local assemblage demands the application of complementary theoretical frameworks and multilevel modelling while controlling the effect of area on species richness.

6.2.3. Disentangling conservation values from overall species diversity values

One of the disagreements between the proponents of the HE and NCS options for conservation is whether all species respond similarly to human disturbance. Some tree species can be inherently rare and at a higher risk of extinction than others either due to their low population density, as in most of the old-growth forests in the tropics (Barlow et al. 2010), narrow geographic range (endemicity) (Pimm et al. 2014), or both. Land use disturbance can also be aggravated by directly removing rare species or by creating favourable conditions for their replacement by wide geographical range species (Waltert et al. 2011). The findings in Chapter 2 show that many species can still adopt and survive in HMFLs in both FRs despite the fact that the identity or rarity of species in land use regimes and HMFLs were not exposed explicitly. Given the vulnerability of rare species, one of the questions in conservation should then be to answer whether protecting areas with high concentrations of rare/vulnerable species should be a top priority or not. The answer to this question will assist in the efficient allocation of limited global conservation resources (Mittermeier et al. 2011).

Indeed, the expansion of protected area has been promoted mostly on the grounds that it will provide a safe haven for those vulnerable rare species that cannot persist under a recurrent human disturbance (Fabricius et al. 2003), even before the invention of the HE option. If conservation fails to succeed to protect vulnerable rare species now, we may not get a second chance to see those species in the future (Mittermeier et al. 2011), or at least their recovery will be expensive. At the same time, presuming all local people as a threat to biodiversity may deprive us of an opportunity to maximize the potential of different land use regimes to the conservation of rare species. Hence, in Chapter 3, this study evaluated the efficiency of protected areas for the conservation of rare species against each land use regime in HMFLs.

Unlike the species richness matrix and *β*-diversity which is based only on the presence/absence of species data used in Chapter 2, Chapter 3 relied on species abundance (frequency) data to discern the different facets of beta (*β*)–diversity. Species traits that contributed most to the dissimilarity between land use regimes and SIF were identified using SIMPER analysis and then linked to change in species abundance distribution (e.g. species richness at the scale of land use, species abundance). This was crucial in disentangling the conservation value of alternative options for rare species (such as canopy and sub-canopy trees of old-growth forest and endemic species) from the overall biodiversity value.

The results exposed that SIF in MFR supported the highest overall tree species diversity. All land use regimes differed in mean *β*–diversity (dis/similarity between local assemblages) and variance *β*–diversity (homogeneity of within-group dissimilarity among different local assemblages). Overall, all land use regimes in MFR lost species, became more uneven, and the dominance of rare old-growth and endemic trees declined and was replaced by shrubs along the land use gradient. The findings in MFR underline the fact that SIF is a superior conservation strategy - be it to overall species diversity or canopy and sub-canopy tree species. On the contrary, old-growth forest and endemic trees in relatively simplified or highly fragmented HMFLs are more vulnerable to local extermination due to land use disturbances. In particular, the change in overall tree species diversity into a distinct assemblage of shrubs in CRUZ demonstrates a regime shift to a different stable state. However, the presence of some rare tree species in TATR and CPA in higher abundance, despite losing many species, implies that those land use regimes in human modified landscapes can still play a significant role as a supplementary strategy.

In contrast, TVFR showed a different pattern of response in species diversity to disturbance. Unlike MFR, SIF in TVFR was not exceptionally efficient in comparison with the other traditional land use, except with intensively disturbed CRUZ. The findings in TVFR highlight the fact that the canopy and sub-canopy tree species in HMFL, with better conditions of forest landscape complexity, are more resilient to disturbance than in a simplified landscape. Except for the decline in abundance in CRUZ, rare tree species were more abundant in relatively less disturbed traditional land use regimes of HMFL than in SIF of TVFR.

It is also worth noting, similar to the case of CPA in MFR, some tree species were more abundant than in SIF, despite the fact that many tree species may have still been lost or declined in abundance as part of a natural process. The decline in abundance or absence of many tree species in CPA, which were found in SIF, may be related to differences in their degree of protection. There have been many studies showing the loyalty of local people to traditional rules that restrict human disturbance in CPA (sacred areas) due to spiritual and symbolic reasons (Ormsby and Bhagwat, 2010). According to IDH (Connell 1979), the rarity of disturbance drives out competitively inferior species and promotes the dominance of late successional species. On the contrary, non-compliance to conservation rules that govern SIF is becoming a universal challenge (Solomon et al. 2015) that might trigger higher species richness as expected by IDH.

It is worth noting that the findings in Chapter 3 show that, unlike the claim of some studies (e.g., Waltert et al. 2011), land use disturbance does not promote the replacement of endemic species by wide range species. Instead, it is most likely that endemic and wide range species are inherently segregated across environmental spaces due to their difference in habitat requirements. However, given higher vulnerability of rare species to local extermination due to demographic and environmental stochasticity, lower genetic diversity and disruption of biotic interaction, trees outside of the SIF can still play a significant role in the persistence of rare species at the landscape scale (Hooftman et al. 2003).

6.2.4. The predominance of culture over ecological abundance on local people's forest and tree species behaviour

The underlying difference between HE and NCS proponents lies in their views on the relationship between nature and culture (Mace 2014). In particular, many conservationists show that traditional society does not only have inextricable links with biodiversity (Paneque-Gálvez et al. 2018, Sutherland et al. 2003) but also often modifies their landscapes in a manner that supports biodiversity conservation (Garnett et al. 2018; Sobrevila 2008). They do so using a dynamic and adaptive TSEK (Yang et al. 2018; Sobrevila 2008, Berks and Turner 2006; Gadgil et al. 1993). In contrast, some authors (e.g. Low and Hein 2017, Low 1996) have been counter arguing that the environmental friendliness of traditional society is a fallacy that may misinform prospective conservation strategy. The forest and tree species use-behaviour of traditional society is ecologically driven (resource abundance) and not from a collective consciousness of conserving nature for future benefit. Others (e.g. Holmes 2013) have been suggesting that local people's support is not a necessity for successful conservation measures. Holmes (2013) further argued that local people have many barriers to challenge even if the unpopular protected areas measure is imposed on them as far as it attains its objective.

The above disagreement hints at the need to understand what really drives use–behaviour of traditional society: culture or ecological abundance of resources? In Chapter 4, this study tested if the ecological appearance hypothesis can sufficiently explain the use-behaviour of traditional society for various uses; fuelwood, construction materials, livestock grazing and browsing, wild food, and traditional medicines. Both ecological and ethnobotanical data gathered were analysed using parametric and non-parametric tests. The results from two communities of the Vhavenda people revealed that there was homogeneity of cultural values pertaining to resource use-behaviour in modifying their landscape for multifunctional uses, although they reside in ecologically different forest conditions (Chapters 2-4).

The most popular utilities were wild food, followed by fuelwood, livestock grazing, and browsing, wood for construction materials and traditional medicine, in descending order. The overall use-value of land use regimes increases with the increase of the land use intensity gradient in multifunctional landscapes as defined by cultural norms and taboos. SIF was an exception. Despite the SIF presumably having the strictest protection status, it had the same use-value as with the open access common resources use zone. While local people complement their household demands of forest and tree species products from the TATR to some extent, there was almost no forest resource harvesting from culturally protected (scared) forests. The study also found that the effect of household characteristics on the differences of user-behaviour was within the bounds of cultural norms and taboos that govern land use decisions (Table 4.5).

The above findings show that traditional society applies dynamic and adaptive TSEK systems (belief, knowledge, and practices) of a particular habitat or species as an integral part of managing the delicate balance of “use-protection” regimes at a landscape level. The protection of sacred forests and habitats and species-related taboos are typical examples of how traditional society still consciously manages landscapes for multifunctional purposes (Berkes et al. 2004).

Based on the findings of the resilience of species diversity (Chapter 2) and change in species abundance distribution (Chapter 3), and the dominance of certain canopy and sub-canopy tree species that are either not available or rarely found in SIF, this study seems to suggest that CPA (sacred forests) areas enjoy practically stricter protection than SIF. In other words, as explained in section 6.2.3, the adherence to social norms and taboos in CPA may have driven the competitive inferior species, by competitive dominant species, either to local extermination debt (decline in population of some species) as in the case of MFR, or local extermination of many species as in the case of TVFR. On the contrary, the findings from SIFs show that the mere designation of forest areas as state-protected may not fully stop illegal forest harvesting. However, the higher species diversity of SIF in Chapters 2 and 3 implies that local people may not harvest as freely as they do in open access CRUZ due to fear of retribution. Hence, the higher species diversity can be due to non-compliance that might induce an intermediate level of disturbance.

Moreover, the abundance of species at the scale of forest landscape did not sufficiently explain the use-value of species as expected by ecological appearance hypothesis. Instead, the two communities share similar cultural keystone species for similar utilities, regardless of the difference in their abundance of species within and between the two HMFLs, and despite the difference in the richness of TSEK (Table 4.5-4.7). Generally, the main findings show that culture plays a predominant role in explaining use-behaviour - be it on superimposing different land use disturbance in multifunctional landscapes or use of species for species utilities. Neither is resource use decision of forest landscapes random, nor is the concept of protected areas a new concept to traditional society. The central question is then, if the Vhavenda cultural groups have rich TSEK and respect for their CPA, why do they harvest forest products illegally from SIF?

6.2.5. Factors influencing non-compliance behaviour towards conservation rules

Amid the current push to further expand strictly protected areas by HE proponents, the Vhavenda communities show that simply designating an area as SIF does not amount to sustainable protection of indigenous forests and tree species (Hayes and Ostrom 2005). The enforcement of compliance rules is crucial for the success of protected areas (Stern 2008; Gavin et al. 2010). However, the high rate of forest and tree product harvesting in both FRs shows that non-compliance behaviour of the Vhavenda communities to state conservation rules is not different from most parts of the world (Robbins et al. 2006, Gavin et al. 2010; Arias 2015; Solomon et al. 2015). Despite the good intentions of most global protected areas, the enforcement of strict protection rules by forest guards and legal systems have proven to be ineffective (Stern 2008), expensive and antagonistic (Wilshusen et al. 2002; Lele et al. 2010). Hence, pursuing the HE option without understanding why local people break state conservation rules does not only jeopardize the ultimate goal of protecting biodiversity, but also amounts to wasteful expenditure of the growingly limited conservation resources.

On the contrary, abandoning the protected area approach with the presumption that (i) the inclusion of TSEK/culture in conservation will boost local support, and (ii) ensuring the persistence of biodiversity in HMFLs without scientific evidence is tantamount to ignoring the fact that unregulated disturbance has a negative impact. The findings of this study show that the deteriorated conditions of species diversity in CRUZ was due to open access to forest and tree resource harvesting (Chapters 2 and 3). Hence, given the highest species diversity of SIF in the vulnerable MFR, protected areas can still play a significant role as *refugia* to retain rare species and a source of propagules for the recovery of the local lost species of human simplified forest landscapes. Although, caution is needed because the effectiveness and efficiency of SIF in MFR may have also been related to their inaccessibility (Table 1.1), rather than the effectiveness of enforcement of conservation rules. Distance from settlement areas and higher location in rugged mountain tops have been related to better performance of many protected areas of the world (Joppa and Pfaff 2009). However, such areas are increasingly becoming rare, and the fate of much of the biodiversity in tropical protected areas is interlinked with the quality of their surrounding HMFLs. Thus, devoting efforts exclusively in the pursuit of the NCS option demands the understanding of the where, why and who (Arias 2015) in traditional societies demonstrate compliance behaviour to conservation rules.

In Chapter 5, the study compared the compliance behaviour of local communities towards rules that govern CPA and SIF. Socio-economic data was collected and analysed using non-parametric tests. All villages did not significantly differ in their proportion of compliance and non-compliance behaviour in CPA and SIF. This is not surprising, considering the homogeneity of the cultural values of the Vhavenda communities demonstrated in Chapter 4. The findings also showed that compliance with CPA was a social norm while non-compliance to SIF appeared to be a new norm. These contrasting findings are in line with Sikhitha's (1999) findings on the symbolic and spiritual significance of sacred forests in Vhavenda communities, and the commentary of Shackleton (2009) on the challenge of enforcement of state forest conservation rules in many parts of South Africa.

The Vhavenda communities, as in India (Ormsby and Bhagwat 2010) and many other countries (Colding and Folke 2001; Kideghesho 2009; Gómez-Baggethun 2013), protect certain species and components of forest ecosystems (e.g. streams and rivers) for rituals, mythical beliefs and totemic values (Mutshinyalo and Siebert 2010). However, higher compliance does not mean the absence of non-compliant people in society. Instead, local governing structures have higher legitimacy and effectiveness in resolving value conflicts and in sanctioning appropriate penalties to non-compliers (e.g., Kideghesho 2009; Ruiz-Mallén and Corbera 2013) than power structures imposed from outside (Infield et al. 2018). Hence, compliance with traditional rules is quasi-volunteer, and enforcement is most likely to be effective when local people establish their own local governing structure (Hayes and Ostrom 2005).

All villages did not differ in the distribution of perceived local values. On average, an individual holds about four out of the seven perceived local values. The majority of local people cited different motives to conserve forest and tree species diversity. These include the need to protect forest and trees of outstanding utility value, watershed protection, and cultural values (identity and symbolic value), protection of endangered species, and wildlife habitat in descending order; although the difference in the proportion of people who held values between the need to protect forest and trees of outstanding utility value and watershed protection were not significantly different (Table 4.1). This shows that, unlike the dichotomy of value orientations between HE and NCS proponents, local people held a continuum of biocentric and anthropocentric value orientation in managing their landscapes for multifunctional purposes. According to value-belief-norm theory, it is not only biocentric value orientation that motivates the need for conservation. When individuals perceive or have an experiential knowledge of adverse environmental conditions that threaten their egoistic value (e.g., family food security and cultural values), they are pushed to act consciously in the protection or mitigation of the impact on biological resources (Turaga et al. 2010; Ives and Kendal 2014).

Half of the local people perceived that the decline of tree species diversity was due to forest and tree species harvesting for rural livelihoods. However, almost all people showed a willingness to take part in the conservation of their landscapes. Although there was no consensus on whether the enforcement of state conservation rules over the last 20 years was weak or strong, almost all members of the communities supported further strengthening of the conservation rules. The regression model (Table 5.2) revealed that individuals who held fewer perceived local values, who perceived negative impacts of forest and tree species harvesting, and weaker rules were mostly likely to be non-compliers.

While perceived experiential knowledge played a major role in explaining non-compliance behaviour, it was those who were aware of the negative impacts of their actions that were most likely to be non-complaint. This appeared to be at odds with a high level of willingness towards conservation initiatives. The most likely explanation of such a contradiction could be that their day to day exposure to forest and tree species in pursuit of their rural livelihoods might have given them better experiential knowledge, which in turn triggered their intention to take part in conservation. However, the gap between environmental awareness, good intention and actual behaviour (Kollmuss and Agyeman 2002) is not unique to the Vhavenda people.

The link between experiential knowledge-willingness to take part in conservation reflects that the widespread non-compliance behaviour towards SIF does not appear to be as a result of lack of interest and ignorance of the impact of illegal harvesting. Rather, local people have to make tough choices to sustain their livelihood against their goodwill in conservation. In such a situation, the majority of local people breach conservation rules that govern SIF out of necessity and limited alternative livelihood options. Overall, the dominance of non-compliers to SIF rules, in the presence of higher compliance to CPA in the same landscape, might be an indication of the strength of preferential sense of place attachment and meaning by local people between the CPA and SIF. Recently, the preferential application of cultural norms and taboos among different places within a landscape has been documented by many studies (Masterson et al. 2017).

6.3. Future research direction

The growing human dominance of nature rendered the distinction between nature and socio-cultural realms in scientific inquiry obsolete (Kueffer et al. 2015). However, the current debate whether to choose HE or NCS appears to focus too much on ideological standpoints and lags behind to embrace biodiversity conservation as a socio-ecological issue. As discussed in the introduction section (Figure 1.2), this study relied on a socio-ecological framework by using different ecological and social theories and hypotheses. However, considering the life span of forests and trees that often take decades or centuries to recover from human disturbance, this makes short term planning uncertain (Fischer 2018). The study, therefore, acknowledges the need for long term research. Based on this study, it was established that there has been neither long term data nor permanent sample plots aligned to the objectives investigated in this research study. Therefore, it is recommended that long term studies on the multifaceted dimensions of biodiversity dynamics overtime, using the sample plots of this study, may assist to understand the long term impacts, thereby facilitating adaptive management when required (Annexure 2.1). The following are the recommended research areas:

Ecological sustainability: Unlike the recent studies on the effectiveness and efficiency of alternative conservation measures, the sustainability of any conservation measure demands additional data gathered over a long period of time under recurrent human disturbance conditions. These include the resilience of structural diversity and dynamics (height and diameter) and the rate of natural regeneration of different species in addition to abundance of co-existing species measured in this study. In particular, the fact that wild food harvesting is the most popular utility (section 6.2.4), assessing its impact on natural regeneration of rare species is recommended.

Bio-cultural hysteresis hypothesis: Many studies have indicated that a cultural shift can occur with severe environmental change or due to culture-unfriendly conservation measures e.g. by detaching cultural societies from their natural surroundings, consequently negatively affecting the adaptive capacity of traditional society to cope with fast-paced global environmental change (Lyver et al. 2019). However, this study could not establish the threshold of forest and tree species diversity change that may cause a change in cultural values, norms, and taboos of traditional society and their coping mechanisms. It is recommended that this be undertaken, considering the current rapid biodiversity loss so as to devise strategies to conserve culture alongside biodiversity conservation.

The context-dependence of nature and culture: The fact that the study relied on the same culture does not represent the influence of cultural diversity of the traditional society in different parts of the world. Hence, investigating the influence of cross-cultural dis/similarity on resource use-behaviour may increase the predictability of human action on larger spatial scales. Hence, it is recommended that similar studies be replicated in different cultural setups.

Non-compliance as disturbance agent: Non-compliance is a non-random ecological disturbance. It follows a specific pattern across space and time. Depending on the extent of illegal harvesting and the resilience of forest biodiversity, non-compliance may maintain, enhance or degrade biodiversity. Nevertheless, there is a paucity of information on the effect of non-compliance for subsistence use of forest products on biodiversity (Robbins et al. 2006) within the parameter of SIF. Considering non-compliance is increasingly becoming a threat for the persistence of many ecosystems and species (Solomon et al. 2015), such empirical evidence may inform appropriate adaptive management interventions (e.g. regulated local uses or strict enforcement to ensure recovery) of existing protected areas and future considerations.

Although this study found higher non-compliance rates in SIF, the study was based on self-reported binary data on users' behaviour and perceptions of local people on the impact of forest and tree product harvesting on forest cover. Given individuals differ on the amount of forest and tree species product needs depending on their socio-economic status, further research is needed to determine the extent of variation among households, communities and cultural groups. This must be linked to their impact on species diversity at different spatial scales in one hand and it consequence on rural livelihood or ecosystem service in general. It is important to emphasise that this study did not assess the consequence of species diversity loss on rural livelihood in particular and ecosystem service due to resource and time limitation. However, the societal consequence of species diversity loss crucial either to develop a viable conservation/ restoration policy. Thus, it is also further recommended to conduct a study on the link of the impact of human disturbance induced species diversity loss on rural livelihood and other ecosystem service.

6.4. Concluding remarks

Based on the findings of the study, the potential of HMFLs for the conservation of forest and tree species diversity over the effectiveness and efficiency of protected areas depends on the prevalent socio-ecological context. It demands context-specific policies that are informed by interdisciplinary science rather than a mere dichotomy of NCS and HE options. The recent claim by the NCS on the remarkable resilience of species diversity is an oversimplification of a complex and non-linear response of local assemblage to disturbance that might misinform future conservation policies and strategies. Equally, the expansion of protected areas of the HE option, with the wholesale presumption that local people are threats to biodiversity, may deem to fail.

This study shows the inextricable link between culture and forest and tree species diversity. The response of local forest and tree species diversity to cultural influence, through land use, depends on the prevailing condition of forest landscape complexity. From a conservation planning perspective, the overarching goals of conservation in human dominated landscapes should be either to restore complexity, in the case of simplified landscapes, to their natural condition or to sustain the better condition of complex landscapes using different use-protection techniques. Ultimately, the sustainability of forest and tree species diversity depends on what we do to improve the condition of the whole landscape, as it improves the resilience of species diversity under recurrent human pressure in the future. However, given a choice to allocate limited resources more efficiently, the immediate action should be to evaluate all possibilities that will maximize the benefit of conserving rare species. Such decisions are simultaneously ecological and social.

Ecologically, the findings in MFR highlight the fact that a state-protected indigenous forest is a superior conservation strategy - be it to overall species diversity or canopy and sub-canopy tree species. On the contrary, wide range and endemic trees in relatively simplified or highly fragmented human forest landscapes are more vulnerable to local extermination due to land use disturbance. Hence, investing the limited conservation resources to reinforce state-protected indigenous forests along with the restoration of the degraded human modified ecosystem should be a top priority. However, the presence of some rare species in TATR and CPA in greater abundance than in SIF hints to the fact that those land use regimes in human modified landscapes can play a significant role as a supplementary conservation strategy.

In contrast with MFR, the findings in TVFR highlight the fact that the canopy and sub-canopy wide range species in the human modified landscape with better conditions of forest landscape complexity are more resilient to disturbance than in a simplified landscape. Except for the decline in abundance in CRUZ, conservation priority species were more abundant in relatively less disturbed traditional land use regimes than in the state-protected forests of TVFR. However, rare endemic canopy and sub-canopy trees in the whole TVFR landscape are as vulnerable to extinction as in MFR. In this regard, effective management of the whole human modified landscape may play a complementary role in biodiversity conservation. Hence, empowering local people to sustain their cultural practices through collaborative forest and tree species conservation projects may ensure the persistence of the full of set of vulnerable species with possible least cost. In sum, the findings hint that a “one size fits all” approach may not work in conservation decisions.

Socially, the future success of protected forest areas or effective management of landscapes for multifunctional landscapes lies mainly on designing policy instruments based on the understanding of factors that affect local people’s forest and tree species use, and conservation behaviour. Local people hold diverse values, traditional ecological knowledge, and experiences that motivate their conservation interest. Conservation of forest and tree species diversity is still an integral part of the mind-set of rural people, at least in the Vhembe Biosphere Reserve. The fact that, on average, an individual holds about four values to conserve tree species diversity in multifunctional landscapes shows that they are neither strictly anthropocentric (egoistic and socio-altruistic) as advocated by new conservation science, nor biocentric as advocated by proponents of protecting the Half Earth movement.

Local people consciously manage forests and tree species diversity in their landscape. Neither land use nor resource use of forest landscapes is random, nor is the concept of protected areas new to traditional society. Unlike the exclusionary protected area approach to preserve a particular biodiversity hotspot, traditional society manages the sustainability of local biodiversity as a socio-ecological system, on which their livelihoods, as well as cultural, emotional, spiritual and symbolic values of their lifestyle depends. The protection of sacred forests and habitats and species-related taboos are typical examples of how traditional society still consciously manages landscapes for multifunctional purposes. Hence, the adherence to social norms and taboos, combined with the resilience of traditional socio-ecological knowledge in human modified landscapes, presents potential tools to complement other global biodiversity conservation efforts. In this regard, inclusive policy reform that integrates local people and their traditional socio-ecological knowledge in landscape management for biodiversity, plays a significant role.

6.6. References

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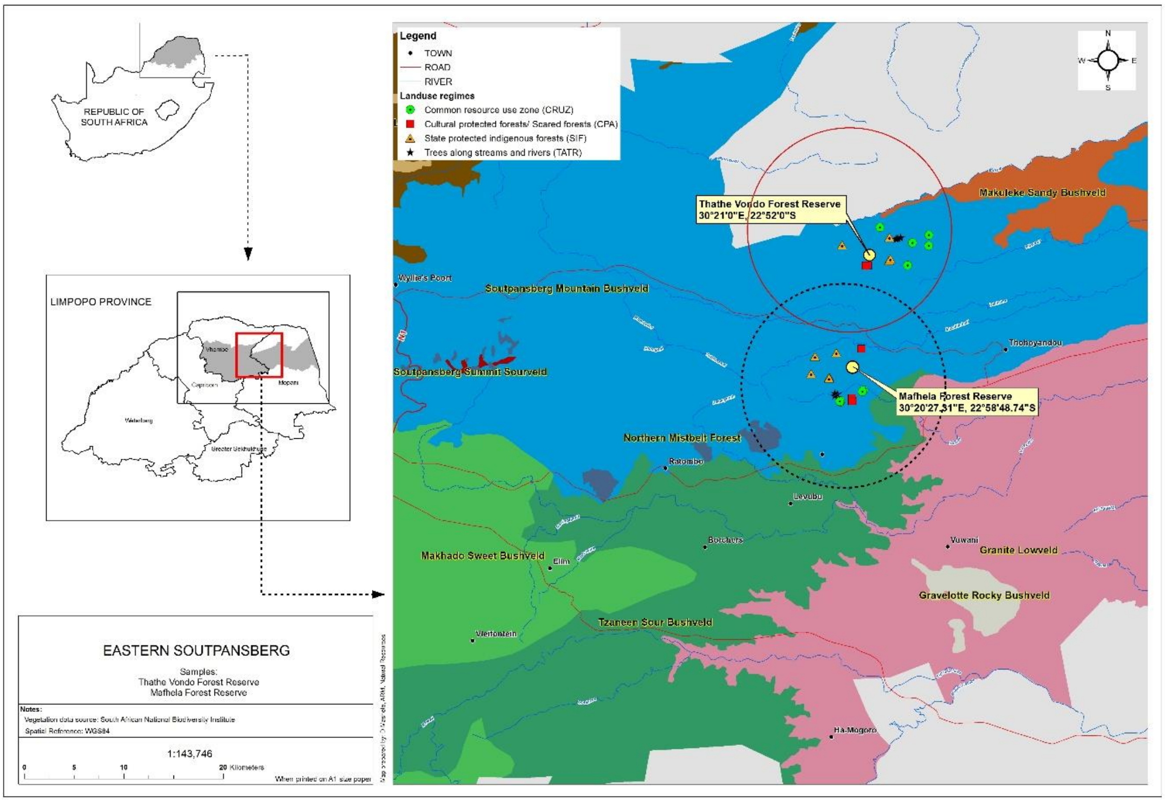
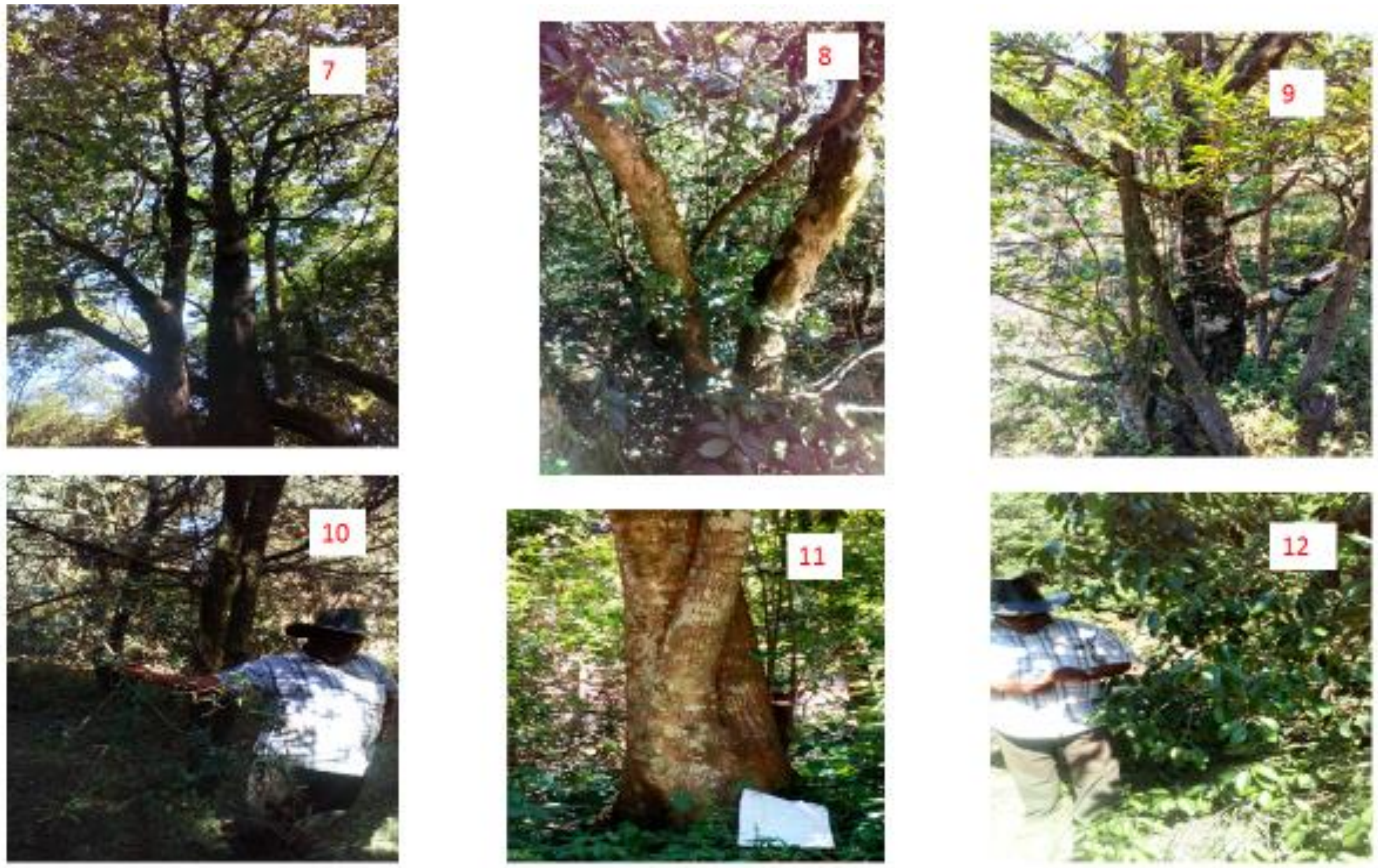
ANNEXURE

Annexure 2.1: sample plots of the Mafhela and Thathe Vondo Forest reserves overlaid on Vegetation map (source: South African National Biodiversity Institute- spatial reference: WGS 84)



Annexure 2.2 Pictures of dominant tree species in the study area

1= *Englerophytum maglismontanum*, 2= *Parinari curatellifolia*, 3= *Psidium guajava*, 4= *Bridelia micrantha*, 5= *Annona senegalensis*, 6= *Aphloia theiformis*



7= *Syzygium gerrardii*, 8= *Xymalos monospora*, 9= *Eugenia natalitia*, 10= *Eugenia natalitia*, 11= *Schefflera umbellifera*, 12= *Mimusops obovata*

Annexure 2.3. List of species observed in the MFR (Mafhela Forest Reserve) and TVFR (Thathe Vondo Forest Reserve). N.B. 1=Present and 0= Absent

Species name	MFR				TVFR			
	CRUZ	TATR	CPA	SIF	CRUZ	TATR	CPA	SIF
Acacia ataxacantha	1	1	1	0	1	1	0	1
Acacia caffra	0	0	0	0	1	0	0	0
Acacia karroo	1	0	0	0	0	0	0	1
Aeschynomene nodulosa	0	0	0	0	1	0	0	0
Albizia adianthifolia	1	1	1	1	1	1	0	1
Annona senegalensis	1	0	0	0	0	0	0	0
Anthocleista grandiflora	0	1	0	0	0	1	0	1
Antidesma vernosum	1	0	0	0	1	1	0	0
Aphloia theiformis	0	0	0	1	1	1	1	1
Apodytes dimidiate	0	0	0	1	1	1	0	1
Bauhinia galphinii	0	0	0	0	1	0	0	0
Bersama tysoniana	0	0	0	1	1	0	1	1
Brachylaena discolour	1	1	1	0	1	1	0	1
Brachylaena rotundata	1	0	0	0	0	0	0	0
Bridelia micrantha	1	1	1	1	1	1	0	1
Buddleja salviifolia	0	0	0	0	1	0	0	0
Canthium ciliatum	0	0	0	0	0	0	0	1
canthium inerme	0	0	0	0	0	0	0	1
Canthium moudianum	1	0	0	0	0	1	0	1
Cassina aethiopica	0	0	1	1	0	0	0	0
Cassine eucleiformis	0	0	0	0	1	1	1	1
Celtis Africana	0	1	1	0	0	0	0	0
Cephalanthus natalensis	1	1	0	1	0	0	0	1
Choristylis rhamnoides	0	0	0	0	0	1	1	0
Citrus lemon	0	1	0	0	0	0	0	0
Clerodendrum glabrum	1	0	0	0	0	0	0	0
Combretum erythrophyllum	0	1	1	1	1	1	0	1

Combretum collinum	1	0	0	0	1	1	0	0
Combretum kraussii	0	0	0	0	0	1	1	1
Combretum molle	1	0	0	0	1	1	0	0
Croton megalobotrys	0	0	0	0	0	1	0	0
Croton sylvaticus	0	1	1	0	1	1	1	1
Cryptocarya liebertiana	0	0	0	0	1	1	1	1
Curtisia dentate	0	0	0	0	1	1	0	0
Cussonia sphaecephala	0	0	0	1	0	0	0	0
Cussonia spicata	1	0	0	1	1	1	0	1
Cyathea capensis	0	0	0	0	0	1	0	0
Cyathea dregei	0	0	0	0	1	1	0	0
Dichrostachys cincrea	1	0	0	1	1	0	0	0
Diospyros lycioides	1	0	0	0	0	0	0	0
Diospyros whyteana	0	0	0	0	0	0	0	1
Ekebergia capensis	0	0	1	0	0	0	0	0
Englerophytum maglismontanum	0	1	1	1	1	1	0	1
Enterospermum rhodesiacum	0	0	0	1	1	1	0	1
Erythrina lysistemon	1	0	0	1	1	0	0	0
Euclea divinorum	1	0	0	0	1	0	0	1
Eugenia natalitia	0	0	0	1	1	1	1	1
Faurea galpinii	0	0	0	0	1	0	0	0
Ficus capensis	1	1	1	1	0	1	0	1
Ficus burkei	0	0	0	0	1	1	1	1
Garcinia livingstonei	0	0	0	0	1	0	0	0
Gymnosporia mossambicensis	0	0	0	1	1	1	0	1
Halleria lucida	0	0	0	1	0	1	1	1
Heteromorpha arborescens	0	0	0	0	0	0	0	1
Heteropyxis natalensis	1	0	0	0	0	0	0	0

Hymenocardia ulmodes	0	0	0	1	0	0	0	0
Ilex mitis	0	0	0	1	1	1	1	1
Indigofera lyalli	0	0	0	0	0	1	0	0
kiggelaria africana	0	0	0	0	0	0	0	1
Lannea discolour	0	0	0	0	1	0	0	0
Lippia javanica	1	0	0	0	0	0	0	1
Maesa lanceolate	1	0	0	0	1	1	0	0
Maytenus pendicularis	0	0	0	0	1	0	0	1
Maytenus senegalensis	1	0	0	0	0	0	0	0
Mimusops obovata	0	1	1	1	1	1	0	1
Mundulae sericea	0	0	0	0	1	1	0	0
Nuxia congesta	0	0	0	1	0	1	0	0
Nuxia floribunda	1	1	0	1	1	1	1	1
Ochna arborea	0	0	0	1	1	1	1	1
Ochna holstii	0	0	0	0	1	1	1	1
Ocotea bullata	0	0	1	0	0	0	0	0
Olea africana	1	0	0	0	1	0	0	1
Olea capensis	0	0	0	0	1	1	1	1
Pachystigma macrocalyx	0	0	0	0	0	0	0	0
Parinari curatellifolia	1	0	1	1	1	1	0	0
Pavetta lanceolate	0	0	0	1	0	1	1	0
Pittosporum viridflorum	0	0	0	0	1	0	0	0
Podocarpus falcatus	0	0	1	1	1	1	1	1
Podocarpus latfolius	0	0	0	0	0	1	0	0
Protea roupelliae	0	0	0	0	0	0	0	1
Prunus africana	0	0	1	0	0	0	0	0
Psidium guajava	1	0	0	0	0	0	0	0
Psychotria capensis	0	0	0	1	1	1	0	1
Psychotria zambamontana	0	0	0	1	1	0	1	1
Pterocarpus angolensis	0	0	0	0	0	0	0	1
Rapaena melanophloeos	0	0	0	0	0	1	0	0

Rauvoflia caffra	0	1	1	1	1	1	0	0
Rhamnus prinoides	0	0	0	0	0	0	1	0
Rhus chiridensis	1	1	1	1	0	0	0	1
Rhus gueinzii	0	0	0	0	0	1	0	0
Rhus rehmanniana	0	0	0	0	1	1	0	0
Rinorea angustifolia	0	0	0	0	0	0	0	1
Rothmannia capensis	0	0	1	0	0	1	1	1
Rothmannia globose	0	0	0	1	0	0	0	1
Schefflera umbellifera	0	0	0	1	1	1	1	1
Senna petersiana	1	0	0	0	0	0	0	0
Strychnos spinose	0	0	0	0	1	0	0	0
Syzygium cordatum	1	1	1	1	1	1	0	1
Syzygium gerrardii	0	0	0	1	1	1	1	1
Trema orientalis	0	1	1	1	0	0	0	0
Trichilia dregeana	0	0	1	1	0	0	1	1
Trilepisuim madagascariense	0	0	1	0	0	0	0	0
Trimeria grandifolia	0	0	0	1	0	0	0	0
Vangueira infausta	1	0	1	0	1	1	0	0
Vernonia colorata	1	0	0	0	0	0	0	0
Vernonia stipulacea	1	0	0	0	0	0	0	0
Warburgai salutaris	0	0	1	0	0	0	0	0
Xylopia parviflora	0	0	1	0	1	1	1	0
Xymalos monospora	0	0	0	1	1	1	1	1
Zanthoxylum capensis	0	0	0	0	1	0	0	0
Zanthoxylum davyi	0	0	0	0	0	0	1	1

Annexure 3.1 PERMANOVA results on the effect of land use induced replacement of endemic species by wide-range geographic species in Mafhela Forest reserve and Thathe Vondo Forest reserve; denoted as MFR and TVFR respectively

Forest reserve	Source	Df	SS	MS	Pseudo-F	p(Perm)	unique terms
MFR	WR	1	1.84	1.84	2.14	0.09	999
	La	3	3.46	1.15	1.29	0.17	996
	Tr	14	12.5	0.89	1.14	0.35	999
	WRXLa	3	1.87	0.62	0.79	0.64	999
	WRXTr	8	5.82	0.72	0.92	0.58	999
	Res	8	6.29	0.78			
	Total	37	31.87				
TVFR	WR	1	0.63	0.63	0.92	0.47	998
	La	3	8.15	2.71	3.08	0.00	998
	Tr	15	12.9	0.86	2.21	0.00	998
	WRXLa	3	1.92	0.64	1.63	0.08	998
	WRXTr	13	7.95	0.61	1.56	0.025	998
	Res	14	7.95	0.39			
	Total	49	5.49				

Land use gradient (La, fixed factor, three levels), Transect (Tr, random factor) were nested in La. WR (Abundance of wide geographical range species), X (interactions), Degree of Freedom (df), Sum of square (SS), F ratio (Pseudo-P), Permuted probability values (P) are shown.

Annexure 4.1: Household questionnaire survey (English version)

I am conducting a research on "*Circa situm* conservation of tree species diversity study of Vhembe Biosphere Reserve in Limpopo, South Africa". The research requires conducting interviews with household members and in various rural communities in Limpopo. The information hereby obtained is solely for academic purposes and all your response will remain confidential. We will share the results with you once completed. The questions are designed to help us understand how your traditional tree-based land use practices can be of paramount significance to conservation of tree species diversity of your landscape and rural livelihood. We thank you for your time.

General information

a. Name of the village ………………………………………………….
b. Date…………………………………………………………………
c. Name of Household head…………………………………………......

I. Household socio-economic characteristics

1. HH size:____
2. Gender:____
3. Age:____
4. Marital status

Marital status	Single	Married	Widow	Divorced	Other
Tick	1	2	3	4	5

5. Level of education

Qualification	No qualification	Primary	Secondary	Tertiary
Tick	1	2	3	4

6. Do you practice crop farming? Yes ☐ 1 No ☐ 2

Please indicate your farm size in the table below

Location	Ha	Mention the three major agronomic/ fruit orchards/vegetables/ pasture	Household consumption, sale or both?
Homestead			
Farm in the Communal land			

7. Do you own livestock? Yes ☐ 1 No ☐ 2

8. Please indicate the type of livestock

Type	Cattle (1)	Goats (2)	Sheep(3)	Chicken (4)	Other(specify) (5)
Number?					

9. What is your employment status? Please indicate the employment status and your place of work below

Status	Full time	Part-time	Casual	Self employed	Pensioner	Student	Unemployed
Please tick	1	2	3	4	5	6	7
Where?							

10. Is there any other member of the family who is employed? What is the status of the employment and where does the person work?

Status	Full time	Part-time	Casual employee	Self employed	Pensioner	Student	Unemployed

Please tick	1	2	3	4	5	6	7
Where?							

ii. Establishment and management of trees in the landscape matrix

11. How do you establish trees in your landscape? Planting ☐ 1 Retaining ☐ 2 Both ☐ 3

12. Where do you keep your trees in the landscape? Please tick in the appropriate column

Technique	**Spatial arrangements of tree species in the landscape matrix**					
	Tree-based land use practice in the human modified landscape				**Forest protection**	
	Homestead	**Common resource use zone**	**River bank?**	**Community Woodlot?**	**Sacred forest /traditionally protected areas**	**State indigenous forest reserve**
Planted	1	1	1	1	1	1
Retained	2	2	2	2	2	2
Both	3	3	3	3	3	3

Source	**Collection of wildenings**	**Own farm/backyard nursery**	**Community nursery**	**State nursery**	**Buying from private nursery**
Tick	**1**	**2**	**3**	**4**	**5**

13. Where do you find your seedlings? (Please tick where appropriate)

14. Do you get extension services related to tree planting and forest management?

Yes ☐ 1 No ☐ 2

15. From which organization do you get extension service? Please indicate in the table below.

From which organization?	**Yes**	**No**	**Specify the kind of services (skills, input, funding and so on)**
Government	1	2	
NGO(e.g. Mpho)	1	2	
University	1	2	
Others(specify)	1	2	

16. Do your community intentionally plant, conserve and/or protect trees outside of the state indigenous forest? Yes ☐ 1 No ☐ 2

17. Why is that important for you to plant, conserve and protect trees in your landscape? Please tick in the table below all that applies

Reason		**Reason**	
Trees are becoming locally rare/endangered in the forest due to over exploitation	1	Cultural values(spiritual values, identity and cultural heritage)	4
They provide tree products for livelihood (utility species)	2	They provide habitat to wildlife protection (mammals, birds , insects and soon) (Keystone species)	5
They provide major watershed protection service (soil and water conservation, protection of water sources, prevention of siltation of dams and lakes) (keystone species)	3	Recreation and landscape beautification	6
		Tourism	7

18. Is there any rules and regulations that governs the utilization, conservation and protection of trees and forests in your village? Yes 1 No 2
19. Who enforce the rules? Traditional leaders 1 local councillors Government Department 2 All 3
20. Do you request to permission either from the chief or the head man to harvest trees and tree products from your pasture and communal lands? Yes 1 No 2
21. Are those rules have changed over the last 20 years? Yes 1 No 2 I do not know 3
22. What is the trend of change in enforcing the rules and regulations for forest and tree utilization and conservation?

Status (tick in the appropriate column)	**Getting stricter**	**Getting weaker**	**No change**
Traditional by law	1	2	3
State rules and regulations	1	2	3

iii. Tree ecosystem provisioning service for rural livelihood

23. Do you harvest tree products from the landscape to sustain your livelihood? Yes 1
No 2

24. From which part of the landscape do you harvest those tree products?

Utility property/ group	Tree-based land use in human modified landscape				Commercial plantation	Forest protection	
	Home Stead	Common resource use zone	River Bank	Community woodlot		Scared / traditionally protected forest	State indigenous forest reserve
Fuel wood	1	2	3	4	5	6	7
Timber harvesting	1	2	3	4	5	6	7
Food	1	2	3	4	5	6	7
Livestock Feed/pasture	1	2	3	4	5	6	7
Medicine	1	2	3	4	5	6	7
Other (specify	1	2	3	4	5	6	7

25. Do you harvest them for household consumption, sale or both? Please tick in the table below

Utility group	**For what purpose?**			**Remarks**
	Household consumption	**Sale**	**Both**	
Fuel wood	1	2	3	
Timber harvesting	1	2	3	
Food	1	2	3	
Livestock feed /pasture	1	2	3	
Medicine	1	2	3	
Other (specify)	1	2	3	

iv. Farmers perception on the tree species diversity and future prospect of tree-based land use for conservation tree species diversity

26. How is the trend of tree cover in you vicinity over the 20 years? Please indicate in the table below

	Status over the last 20 years			**Remarks**
	Increasing	**Decreasing**	**No change**	
State indigenous forest reserves?	1	2	3	
Sacred/traditional protected forest	1	2	3	
Homesteads	1	2	3	
Common resource use zone	1	2	3	
Riverbanks	1	2	3	
Community woodlots	1	2	3	

27. What do you anticipate the trend of tree cover in you vicinity over the coming 20 years? Please indicate in the table below

	Anticipation over the coming 20 years			**Remarks**
	Increasing	**Decreasing**	**No change**	
State indigenous forest reserves?	1	2	3	
Sacred/traditional protected forest	1	2	3	
Homesteads	1	2	3	
Common resource use zone	1	2	3	
Riverbanks	1	2	3	
Community woodlots	1	2	3	

28. What are the threats affecting the forest cover and tree species diversity around your landscape over the last 20 years? Please tick in all that your perceive are threats

Threat	**Tick**	**Threat**	**Tick**
Land clearing for cropping /orchards	1	Over exploitation by community	6
Expansion of settlement	2	Fire	7

Overgrazing	3	Mining	8
Expansion of forest plantations	4	Climate change	9
Invasive alien species	5	Other (specify)	10

29. How do you perceive tree products harvesting by the local community affect tree species diversity in the landscape over the last 20 years?

Effect of harvesting on tree species diversity on the following	**Status over the last 20 years**			**Remarks**
	Increasing	**Decreasing**	**No change**	
Your homestead?	1	2	3	
Common resource use zone	1	2	3	
Along the river bank?	1	2	3	
Community woodlot?	1	2	3	
Scared/traditionally protected forest	1	2	3	
State indigenous forest reserves?	1	2	3	

30. Do change of tree cover and tree species abundance in the landscape directly affect your livelihood? Yes 1 No 2

31. How does it affect you in terms of availability of forest products for day to day consumption? Please tick in the appropriate column below

Utility property	**No effect**	**Scarcity during normal time**	**Acute shortage during critical time (e.g. drought and disasters)**	**Not available anymore**
Fuel wood	1	2	3	4
Construction	1	2	3	4
Wild food	1	2	3	4
Livestock feed	1	2	3	4
Medicine	1	2	3	4
Other (specify)	1	2	3	4

32. How do you cope at times of scarcity or absence of tree products that are vital for your livelihood in your vicinity?

Coping mechanism	**Yes**	**No**	**Remark**
Do you harvest from sacred / traditional protected forests?	1	2	
Do you harvest them from the state indigenous forest reserves?	1	2	
Do you harvest outside of your vicinity?	1	2	
Do you collect from commercial plantations?	1	2	

Procuring from the market?	1	2	
Other mechanisms (specify)	1	2	

33. Do you believe trees in your homestead, common resources zones, and river banks in your landscape under traditional authority help for the sustainability of tree species in state indigenous forest reserves? Yes 1 No 2 I do not know 3

34. Do you believe sacred/traditionally protected forests in your landscape help for the sustainability of tree species in state indigenous forest reserves? Yes 1 No 2 I do not know 3

35. Are you willing to participate in any initiative to conserving and enhancing tree species diversity in your village? Yes 1 No 2

36. To what extent would you be willing to participate in this initiative of conserving tree species in your local area?

Degree of willingness	**Strongly willing**	**Willing**	**Undecided**	**Remark**
Tick	1	2	3	

37. How do you propose them to be done? Tick in all that applies

Proposed mechanism of conserving and enhancing	**Tick**	**Remark**
By planting indigenous trees in the homestead and farm	1	
By planting fruit, fuel wood and timber trees in the homestead and farm	2	
By establishing community woodlots	3	
By planting trees wherever possible	4	
By protecting sacred/traditional protected forests	5	
By protecting state indigenous forest reserves	6	
Other (specify)	7	

38. Please state any general issue in your opinion that is very important, but not discussed above relating with conservation and protection of tree species in your vicinity

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Annexure 4.2. Household questionnaire survey (Tshivenda version)

Ndi kho ita tsedzuluso nga **“*Circa situm* conservation of tree species diversity study of Vhembe Biosphere Reserve in Limpopo, South Africa”(Tsedzuluso ya tshikolo ngaha kuvulungele kwa madaka fano afrika tshipembe)**.tsedzuluso heyi I kho toda u vhudzisa vhadzulapo vha mashango a mahayani a limpopo. Tsedzuluso heyi ndiya tshikolo ende phindulo dzavho dzi do fhariwa nga ndila ya vhudi. Ri do vha divhadza zwe rawana tsedzuluso musi I tshifhela.mbudziso hedzi dzo dzudzanyiwa nga ndila ine ya do ri thusa u divha ku pfesesele kwavho kwa madaka kha vhuthogwa ha u vhulunga miri ya madaka yo fambanaho na u shela mulenzhe kha ku tshilele kwavho. Ri do livhuwa tshumisano na tshifhinga tshavho.

General information

Dzina la shango …………………….……………………………………….

Duvha ………………………………………………………………………

Dzina la thoho ya muta……………….……………………………….........

A. Ku tshilele kwa mudini

1. Ni dzula ni vhangana mutani:____________________
2. mbeu:_______________________________________
3. minwaha:____________________________________
4. tshiimo tsha mbingano

Tshiimo tsha Mbingano	Usa mala Kana usa maliwa	U maliwa Kana U mala	Ndo lovheliwa	Ndo Taliwa	Zwinwe
kha vha swae	1	2	3	4	5

5. vho dzhena tshikolo u swika gai?

Vho dzhena u Swika gai	A thi ngo dzhena	phurayimari	sekondari	theshiari
Kha vha swae	1	2	3	4

6. Vha ita zwavhulimi? Vhalima zwi fhio?Ee ☐ 1 Hai ☐ 2

Fhethu	**Ha**	Kha vha sumbedzise zwilimiwa zwavho zwi raru zwine vha zwilimesa,miroho midala kana mitshelo	Vha shumisa zwilimiwa ula? U rengisa? Kana zwothe?
U lima hayani			
Vha lima tshimuni			

7. Vhana zwifuwo? Ee ☐ 1 Hai ☐ 2

8. Kha vha sumbedzise tshaka dza zwifuwo zwine vha zwifuwa

Tshaka	kholomo (1)	Mbudzi (2)	Nngu(3)	Khuhu (4)	Zwinwe (5)
Nomboro?					

9. Vha a shuma?kha vha sumbedzise hune vha shuma hone?

Tshiimo	Wa tshothe	Wa tshifhinga nyana	u shuma nga zwifhinga	U di shuma	Vha hola Mudende	U jhena tshikolo	Usa shuma
Kha vha swae	1	2	3	4	5	6	7
Gai?							

10. Huna munwe ane a shuma hafha mutani? A vha sumbedzise uri u shuma gai

Tshiimo	Wa tshothe	Wa tshifhinga nyana	u shuma nga zwifhinga	U di shuma	Vha hola Mudende	U jhena tshikolo	Usa shuma
Kha vha swae	1	2	3	4	5	6	7
Gai?							

B. U thoma ha u langula ha madaka

11. Vha thoma hani miri shangoni lavho? U tavha 1 I dzivha hone 2 Zwothe 3

12. Vha thoma gai a u thogomela miri? Kha vha nange

Thekiniki	**Ku dzudzanyelo kwa miri**					
	Vha thogomela miri gai				**U tshileledza daka**	
	Ha yani	**Hune ha rhediwa khuni ene kholomo Dzala hone**	**Tshini na mulambo**	**Bulasini la Miri ya u ita Khuni**	**Daka la Mvelelo**	**Daka la muvhuso**
Miri yo tavhiwaho	1	1	1	1	1	1
Vha wana i hone	2	2	2	2	2	2
Zwothe	3	3	3	3	3	3

13. Vha wana gai zwimela?(kha vha swae)

Hune vha wana hone	**Vha dzi wana Madakani**	**Vha dzi wana burasini kana gadeni**	**Gadeni ya tshi Tshavha**	**Gadeni ya muvhuso**	**Vho to renga**
Tick	**1**	**2**	**3**	**4**	**5**

14. Vha wana thuso mayelana na u tavha miri na u langula miri ya daka?

Ee ☐ 1 Hai ☐ 2

15. Vha wana thuso kha madzangano a fhio?

Madzangano	**Ee**	**Hai**	**Kha vha sumbedzise zwine vha thusiwa ngazwo(tshenzhemo,thuso ya masheleni,ya zwi shumiswa kana zwinwe?)**

Muvhuso	1	2	
Madzangano asi a muvhuso(NGO)	1	2	
Gudidzini lihulwani	1	2	
Hunwe?	1	2	

16. Tshi tshavha tshavho tshi a tavha,u vhulunga na u tsireledza miri nnda ha fhethu ho tsireledziwaho miri? Ee ☐ 1 Hai ☐ 2

17. Ndi ngani zwi zwa vhuthogwa u tavha , u tsireledza na u vhulunga miri ya daka? Kha vha sumbedzise

Tshiitisi		**Tshiitisi**	
Miri I kho thoma u fhela ngau shumiseswa	1	U itela zwa mvelelo na vhurereri	4
U wana zwibveledzwa zwine zwia thusa kha ku tshilele	2	Tsireledza na u disa vhudzulo ha zwipuka zwa daka	5
Tsireledza kha madi na mavu	3	Zwa vhudi mvumvusi	6
		Vha endela mashango	7

18. Huna milayo kana ndila dzine dza shumiswa u langula ku shumisele, u vhulunga na u tsileledza miri vhuponi havho? Ee ☐ 1 Hai ☐ 2

19. Ndi vhonnyi vhano vheya milayo? Vha musanda ☐ 1 vha mukhaselara ☐ 2 vha madaka ☐ 3 vhothe ☐ 4

20. Vhaya humbela thendelo ya u rhema daka kha vho musanda kana vhakoma?

Ee ☐ 1 Hai ☐ 2

21. Milayo yo tshentsha kha minwaha ya 20 yo fhelaho? Ee 1 Hai 2 a ti divhi 3

22. Nyimelo ya milayo kha ku shumisele nau vhulunga madaka kha tshifhinga tsha zwino?

Tshiimo	**I kho konda**	**I kho leluwa**	**A huna tshanduko**
Milayo ya vha musanda	1	2	3
Milayo ya muvhuso	1	2	3

C. Miri i thusa hani kha ku tshilele kwa mahayani

23. Vha wana zwibveledzwa zwa miri kha u thusa ku tshilele kwavho?

Ee 1 Hai 2

24. Vha zwi wana gai zwibveledzwa zwa miri?

Gurupu	**Ku dzudzanyelo kwa miri**				**Burasini la mibomo na mupayini**	**U tsileledza daka**	
	hayani	**Hune ha rediwa khuni ene kholomo** **Dzala hone**	**Tshini na mulambo**	**Bulasini la khuni**		**Daka la mvelelo**	**Daka la muvhuso**
Khuni	1	2	3	4	5	6	7
Matanda	1	2	3	4	5	6	7

Zwliwa zwa daka	1	2	3	4	5	6	7
Zwiliwa zwa zwifuwo	1	2	3	4	5	6	7
Mishonga	1	2	3	4	5	6	7
Zwinwe	1	2	3	4	5	6	7

25. Vha shumisa zwi bveledzwa u itela muta wavho? U rengisa? Kana zwothe? Kha vha swae

Gurupu	**Vha itela mini?**			**Vha zwi vhonisa hani?**
	U shumisa mutani	**U rengisa**	**zwothe**	
Khuni	1	2	3	
matanda	1	2	3	
Zwiliwa zwa daka	1	2	3	
Zwiliwa zwa zwifuwo	1	2	3	
Mishonga	1	2	3	
Zwinwe	1	2	3	

D. Vhonele kwa vho ramabulasi kha ku shumisele na ku vhulungele kwa miri yo fambanaho

26. Ku dzulele kwa miri kha minwaha ya 20 yo fhelaho

	Nyimelo kha minwaha ya 20 yo fhelaho			**Vha zwi vonisa hani**
	U engedzea	**U fhungudzea**	**Ahuna tshanduko**	
Daka la muvhuso	1	2	3	
Daka la mvelelo	1	2	3	
Hune vha rediwa khuni ene kholomodzala hone				
Hune zwifuwo zwala hone	1	2	3	
Tshini na mulambo	1	2	3	
Bulasini la miri ya khuni	1	2	3	

27. Ku vhonele kwavho kha ku dzulele kwa madaka kha minwaha 20 idaho?

	Nyimelo kha minwaha ya 20 idaho			**Vha zwi vonisa hani**
	U engedzea	**U fhungudzea**	**Ahuna tshanduko**	
Daka la muvhuso	1	2	3	
Daka la mvelelo	1	2	3	
Hune vha rediwa khuni ene kholomodzala hone	1	2	3	

Hune zwifuwo zwala hone	1	2	3	
Tshini na mulambo	1	2	3	
Bulasini la miri ya khuni	1	2	3	

28. Ndi mini zwono tshinyadza madaka na miri kha minwaha ya 20 yo fhelaho? Kha vha swae

Zwi tshinyadzi	**Kha vha swae**	**Zwi tshinyadzi**	**Kha vha swae**
U rema hukho iteliwa zwa vhulimi		U shumiseswa ha madaka nga vhadzulapo	
U fhatiwa ha zwifhato kha tshitshavha		Mulilo	
U fula ha zwifuwo		Mugodi	
U tavhiwa ha miri ya zwavhumbidudzi		U tshentsha ha mutsho	
U engedzea ha miri yo bva kha manwe mashango		Zwinwe	

29. Vha vhona hani ku shumisele kwa zwibveledzwa zwa madaka kha vhupo havho kha minwaha ya 20 yo fhelaho?

Tshinyalelo nga u rema madaka	**Nyimelo kha minwaha ya 20 yo fhelaho**			**Vha zwi vhonisa hani**
	U engedzea	**U fhungudzea**	**Ahuna u tshanduko**	

Hayani?	1	2	3	
Hune vha rediwa khuni ene kholomo Dzala hone	1	2	3	
Tshini na mulambo?	1	2	3	
Hune ha tavhiwa miri ya khuni?	1	2	3	
Daka la mvelelo?	1	2	3	
Daka la muvhuso?	1	2	3	

30. U fhungudzea ha madaka na ku wanele kwa miri zwia tshinya kha ku tshilele kwavho ?

Ee ☐ 1 Hai ☐ 2

31. Ku wanelo kwa zwi shumiswa zwa madaka zwi a kwama ku tshilele kwavho kwa duvha na duvha? Kha vha swae

Vha shumisa mini	**Zwia wanaleya**	**Zwia konda kha tshinwe tshifhinga**	**U konda u wanala kha zwifhinga zwo konda sa tshumbo huna gomelelo**	**A zwi tsha wanala**
Khuni	1	2	3	4
Zwo fhata	1	2	3	4
Zwiliwa zwa daka	1	2	3	4
Zwiliwa zwa zwifuwo	1	2	3	4
mishonga	1	2	3	4

Zwinwe	1	2	3	4

32. Vhan kona u tshila hani husina zwinwe zwa zwibveledzwa zwa madaka kana zwi tshi kho konda u wanala?

ndila ine vha kona ngayo	**Ee**	**Hai**	**Vha zwi vhonisa hani**
Vha wana zwi bveledzwa fhethu ho tsileledziwaho nga zwa mvelelo?	1	2	
Vha wana zwi bveledzwa fhethu ho tsileledziwaho nga muvhuso?	1	2	
Vha zwi wana tsini na hune vha dzula hone?	1	2	
Vha zwi wana madakani a zwa vhubindudzi?	1	2	
Vha renga mimakete?	1	2	
Dzinwe ndila?	1	2	

33. Vhaakholwa uri miri ino wanala hayani,mabulasini na hunwe uri iya thusa kha u vhulunga miri ino wanala madakani o tsileledziwaho nga muvhuso?

Ee 1 Hai 2

34. Vhaakholwa uri daka la mvelelo liya thusa kho tshileledza miri dakani la muvhuso?

Ee 1 Hai 2

35. Vho di imisela u shela mulenzhe kha u vhulunga na u khwinisa miri yo fambanaho vhuponi havho?

Ee ☐ 1   Hai ☐ 2

36. Vho di imisela u swika gai kha u shela mulenzhe u vhulunga miri ya daka vhuponi havho?

Vhu di imisela	Ndo di imisela vhukuma	A thingo di imisela	Ndi vhukati	Vha zwi vhonisa hani
Kha vha swae	1	2	3	

37. Vha nga kona hani uri zwi itiwe? Kha vha swae

Ndila dzine vhanga dzi shumisa u vhulunga na u khwinisa	Kha swae	Vha zwi vhonisa hani
Vha nga tavha miri ya daka mahayani na mabulasini	1	
Vha nga tavha miri ya mitshelo na miri ya khuni mahayani na mubulasini	2	
Vha nga tavha hune vha do reda hone khuni	3	
Vha nga tavha miri hunwe na hunwe hune vha nga kona	4	
Vha nga tsileledza daka la mvelelo na vhurereri	5	
Vha nga tshireledza madaka o tsireledziwaho nga muvhuso	6	
Zwinwe	7	

38. Kha vha ambe zwinwe zwa ndeme zwine a zwi ngo ambiwa kha mbudziso dzothe dzo vhudzisiwaho hafha nntha kha u vhulunga kana u tsireledza madaka kha vhupo havho.

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Annexure 4.3. List of important tree species and their use value. FW=Fuelwood, CON=Construction, WF=Wild food, G&B =grazing and browsing, TM=Traditional medicine, ha=hectare. N.B. The Av.abu/ha is calculated from Av.abu/plot which is 20mX10m

(i)Mafhela Forest Reserve

Tree species	Tshivenda name	Av.abu/ha	FW	CON	WF	G&B	TM	Total
Bridelia micrantha	Mukumbakumbane	20.37	0.53	0.24	0.35	0.36	0.11	1.59
Parinari curatellifolia	Muvhula	11.11	0.59	0	0.18	0.16	0.28	1.21
Englerophytum magalismontanum	Munombelo	27.78	0.175	0.025	0.63	0.04	0	0.87
Celtis Africana	Mubvubvu	5.56	0.35	0.23	0.065	0.04	0.025	0.71
Brachylaena rotundata	Mufhata	0.93	0.38	0.185	0	0.08	0.02	0.665
Syzygium cordatum	Mutu	11.11	0.11	0.065	0.27	0.04	0.085	0.57
Nuxia floribunda	Munanotshi	11.11	0.275	0.115	0.04	0.02	0.05	0.5
Combretum molle	Mugwiti	1.85	0.26	0.02	0	0.02	0.165	0.465
Mimusops obovata	Mububulu	7.41	0	0	0.325	0.02	0.04	0.385
Aphloia theiformis	Mufhefhera	10.19	0.23	0.09	0	0.04	0.025	0.385
Cephalanthus natalensis	Mutsanda	4.63	0	0.02	0.2	0.04	0	0.26
Albizia adianthifolia	Muelela	5.56	0.06	0.08	0	0.08	0	0.22
Acacia karroo	Muunga	10.19	0.15	0	0	0.02	0.05	0.22
Euclea crispa	Mutangule-nyele	7.41	0	0	0.02	0.02	0.17	0.21

Annona senegalensis	Muembe	9.26	0.02	0.02	0.105	0	0.04	0.185
Canthium mundianum	Muvhibvhelashadani	1.85	0.05	0.02	0.09	0.02	0	0.18
Lippia javanica	Musudzungwane	4.63	0	0	0	0.04	0.13	0.17
Hetropyxis natalensis	Mudede	1.85	0.02	0	0	0	0.13	0.15
Ficus natalensis	Tshikululu	12.96	0	0.02	0.085	0.04	0	0.145
Enterospermum rhodensiacum	Muhasha-phande	0.93	0.12	0	0	0	0	0.12
Warburgia salutaris	Mulanga	0.93	0	0	0.025	0	0.07	0.095
Dichrostachys cinerea	Murenzhe	7.41	0.02	0	0	0.02	0.04	0.08
Schefflera umbellifera	Mukho	5.56	0.04	0.02	0	0.02	0	0.08
Antidesma vernosum	Mukwalikwali	6.48	0.02	0	0.02	0.02	0	0.06
Eugenia natalitia	Tshitawatawane	1.85	0	0	0.025	0	0.025	0.05
Prunus africana	Mulalamanga	0.93	0.025	0.02	0	0	0	0.045
Aeschynomene nodulosa	Muvumbaredzi	0.00	0.02	0.02	0	0	0	0.04
Podocarpus falcatus	Mufhanza	10.19	0	0.02	0	0	0	0.02

(ii) Thathe Vondo Forest Reserve

Tree species	Tshivenda name	Av.abu/ha	FW	CON	WF	G&B	TM	Total
Parinari curatellifolia	Muvhula	10.17	1	0.09	0.21	0.26	0.09	1.65
Englerophytum magalismontanum	Munombelo	21.19	0.46	0	0.95	0.05	0.01	1.48
Enterospermum rhodensiacum	Muhasha-phande	6.78	0.86	0.35	0	0.03	0.03	1.27
Olea capensis	Musiri	16.95	0.89	0.27	0	0	0.04	1.2
Syzygium gerrardii	Mutawi	21.19	0.56	0.03	0.57	0.02	0.01	1.19
Combretum molle	Mugwiti	6.78	0.56	0.07	0	0.06	0.31	1
Olea Africana	Mutlhwari	2.54	0.68	0.17	0.12	0.01	0	0.98
Albizia adainthifola	Muelela	13.56	0.32	0.02	0.01	0.62	0	0.98
Mimusops obovata	Mububulu	12.71	0.23	0.05	0.67	0	0.01	0.97
Syzygium cordatum	Mutu	11.86	0.43	0.06	0.18	0.01	0.01	0.7
Brachylaena discolour	Mufhaṱathavha	9.32	0.24	0.25	0	0.05	0.02	0.57
Pterocarpus angolensis	Mutondo	0.85	0.13	0.02	0.12	0	0.16	0.43
Bridelia micrantha	Mukumbakumbane	7.63	0.1	0.04	0.09	0.13	0.05	0.42
Xylopia parviflora	Muvhulavhusiku	5.93	0.24	0.13	0	0	0.03	0.4
Canthium mundianum	Muvhibvhelashadani	1.69	0.08	0.1	0.13	0.07	0.01	0.39
Anthocleista grandiflora	Mueneene	7.63	0	0	0	0.3	0.03	0.33
Acacia karroo	Muunga	1.69	0.04	0	0	0.1	0.16	0.3
Vangueria infausta	Mavelo	4.24	0.01	0	0.26	0	0	0.27
Lippia javanica	Musudzungwane	0.85	0	0	0	0.23	0.04	0.27
Nuxia floribunda	Munanotshi	11.86	0.21	0.02	0	0	0	0.23
Faurea saligna	Mutango	0.85	0.13	0	0.05	0	0	0.18
Ficus sur	Muhuya gale	3.39	0.01	0	0.13	0.01	0.02	0.17
Trichilia dregeana	Mutuhu	1.69	0.06	0	0	0	0.1	0.16
Ochna holstii	Tshipfure	9.32	0.02	0.01	0.06	0.03	0	0.12

Garcinia livingstonei	Mupimbi mupimbi	0.85	0	0	0.08	0	0.01	0.09
Croton sylvaticus	Mulathoho	11.02	0.07	0	0	0	0.01	0.08
Aphloia theiformis	Mufhefhera	18.64	0.07	0.01	0	0	0	0.08
Aeschynomene nodulosa	Muvumbaredzi	0.85	0.07	0	0	0	0	0.07
Eugenia natalitia	Tshitawatawane	16.95	0	0.03	0.03	0	0.01	0.06
Apodytes dimidiate	Tshiphophamadi	4.24	0.06	0	0	0	0	0.06
Mundulea sericea	Mukundan dou	1.69	0.03	0	0	0	0.01	0.05
Euclea crispa	Mutangule	1.69	0	0	0.01	0.02	0.02	0.05
Bauhinia galpinii	Mutswiriri	0.85	0	0.01	0	0.01	0.02	0.04
Maytenus peduncularis	Mukwatule	1.69	0.02	0	0	0	0.02	0.04
Dichrostachys cinerea	Murenzhe	0.85	0	0	0	0.02	0.01	0.04
Antidesma vernosum	Mukwalikwali	1.69	0	0	0	0	0.03	0.03
Cassine eucleiformis	Munamu	19.49	0.01	0.01	0	0	0	0.02
Rauvolfia caffra	Munadzi	1.69	0	0	0	0	0.02	0.02
Diospyros mespiliformis	Mudoma	0.85	0	0	0.01	0	0	0.01
Xymalos monospora	Tshipengo	23.73	0	0	0	0	0.01	0.01
Cryptocarya liebertiana	Munenze	6.78	0.01	0	0	0	0	0.01
Schefflera umbellifera	Mukho	16.10	0.01	0	0	0	0	0.01