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Native Plant Colonization of *Eucalyptus* Plantations in Farmscapes in Relation to Local, Landscape and Historical constraints: Implications for Ecological Restoration

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This is to certify that the thesis prepared by Habte Telila, entitled: *Native Plant Colonization of Eucalyptus Plantations in Farmscapes in Relation to Local, Landscape and Historical Constraints*. *Implication for Ecological Restoration* submitted in fulfillment of the requirements for the degree of Doctor of Philosophy (Plant Biology and Biodiversity Management) compiles with the regulations of the University and meets the accepted standards with respect to originality and quality.

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Abstract

Agricultural expansion has caused much habitat loss across the world. Still there can be substantial levels of biodiversity in such landscapes and we need to understand more about the distribution of biodiversity in such landscapes. In some agricultural landscape there is a reforestation or afforestation trend, often with exotic trees and with the aim of generating wood products. To what extent plantations of exotic trees can harbor native biodiversity is an important question. Generally, the capacity of plantations to contain native species depend both on local, landscape and historical factors. The current study was conducted in a farmscape in central Ethiopia characterized by scattered small Eucalyptus plantations located at different distances from a remnant forest (Chilimo forest). The objectives of this study were to understand to what extent local, landscape and historical factors constrain or foster native woody and herb species regeneration in these Eucalyptus plantations. Additionally the roles of retained, isolated trees of Podocarpus falcatus and Juniperus procera as propagule sources were investigated. All native woody and herb species in 60 small (0.5-1.75ha) Eucalyptus plantations embedded in an open farmscape ranging in distance from 0.1 to 12 km from the remnant continuous forest were collected and identified. Presence/absence data on herbs was also collected in four 100 m x 50 m plots in the Chilimo forest to be used for characterization of the forest herb flora. The ages of the studied Eucalyptus plantations were estimated by using Aerial photosfrom1988-2014. The location of remnant trees of P. falcatus and J. procera were recorded within a circular plot of 500 m radius around each Eucalyptus plantations. A total of 1571 individuals of native woody plants belonging to 55 species were recorded in the plantations. The number of woody species in a plantation increased significantly with the height of the grass sward indicating sensitivity to grazing. Moreover, the number of woody species in the Eucalyptus plantations decreased significantly with distance to the forest showing that they were dispersal limited. The likelihood of presence of P. falcatus in Eucalyptus plantations increased with connectivity to retained mature trees in the surrounding area, but not so for J. procera. For the herbs a total number of 92 species were recorded in the plantations, of which 47 were denoted forest species and 45 non-forest species based on the comparison with the forest plot data. The number of forest herb species in the Eucalyptus plantations decreased with distance from the forest why this not was the case for the non-forest herb taxa. However, both the number of forest and non-forest herb species increased with the height of grass swards illustrating sensitivity to grazing also for this species groups. To conclude, sowing or planting native plants will be necessary in most plantations across the country if biodiversity of the plantations should increase and more resemble forest biodiversity, since only few remnant natural forests that could act as seed sources occur across the Ethiopian highlands. Another main obstacle might be the prohibition of selling timber of native trees, which indirectly discourage farmers from letting native trees regenerate. Thus the increasing cover of Eucalyptus seen across the country will not automatically foster a recovery of native woody plant biodiversity, even if it is managed to optimize local environmental conditions.

Keywords: *Chilimo forest, colonization, dispersal, farmscapes, native flora, restoration, small-scale Eucalypts plantations*

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1. Introduction

The number of human induced activities such as forest loss and degradation are some of the main threats to biodiversity (Fahrig *et al*., 2003; Brook *et al*., 2008). In the tropics, deforestation is one of the main factors in the erosion of biodiversity and is estimated to be at a rate of 13 million hectares per year (Sorley, 1999; FAO, 2007). It is similarly important to give conservation priority for areas rich in biodiversity because of their values as repositories of biological diversity and store house of carbon (Alexander *et al*., 2014). Hence, recovery of biodiversity is a critical concern of ecological restoration (Brudvig, 2011; Maron *et al*., 2012).The schemes of restoration could be through natural recovery (McIver and Starr, 2001) and plantation forests (Bohre and Chaubey, 2014).

Generally the main purpose of tree plantations is for economic purposes such as timber production and fire wood (Sorley, 1999). Nowadays, it is common to see research findings showing tree plantations as an alternative site for the regeneration of native plant species (Feyera Senbeta *et al*., 2002). The tree plantations enhance the regeneration by suppressing competitive grass species and other light demanding species (Parrotta, 1995, 1997; Koonkhunthod *et al*., 2007), and creating a micro-climate that favour colonization and as "stepping stones" between remnant natural forests and agricultural matrix enhancing seed dispersal (Parrotta, 1995; Hartley 2002; Lamb *et al*., 2005; Carnes *et al*., 2006; Ashton *et al*., 2014). Plantations could also contribute to the conservation of biodiversity by providing an environment not only for native woody plants but also for the forest herb species (Boothroyd-Roberts *et al*., 2013).

However, there are some views that consider plantations as sites devoid of valued habitat for other organisms (Brockerhoff *et al*., 2008).Thus, it is important to understand under what circumstances plantations could play a role for restoration purposes. The effectiveness of plantations in restoring biodiversity are expected to be influenced by distance to the native seed source and presence of seed dispersing agents (Eshetu Yirdaw, 2001), plantation species and age (Geldenhuys, 1997; Brockerhoff *et al*., 2003). Furthermore, it could be influenced by local habitat quality variables such as grazing (Mathews *et al*., 2009) and canopy openness (Mulugeta Lemenih *et al*., 2004).

In addition, it is important to take into account which species may benefit from plantation establishment and which may be negatively affected by it (Gonzales and Nakashizuka, 2010). Restoration studies in exotic plantations focused mostly on tree regeneration as a target of forest restoration efforts (Boher *et al*., 2012). In contrast, the restoration of herb species using exotic plantations has been given less attention. In some situations, exotic plantations show higher yields and greater sustainability of native woody species when the colonization of herbs was successful (Parrotta *et al*., 1997). Additionally, herbaceous layer vegetation may be important indicators of site quality in assisting regeneration of woody species and in integrating the whole ecosystem (Hutchinson *et al*., 1999; Bohre *et al*., 2012).

Another major challenge of biodiversity conservation in the tropics is how to balance the need for food security and conserving biodiversity (Ries *et al*., 2010; Harvey *et al*., 1999). The conventional way to protect biodiversity is through establishment of protected forested areas and parks (Zahawi and Augspurger, 2006). Thus, it is vital to find ways that combine the effort to conserve biodiversity while feeding the alarmingly increasing human population (Green *et al*., 2005: Harvey *et al*., 2010). Moreover, biodiversity in the protected areas is connected to the surrounding agricultural matrix (Harvey *et al*., 2006) and this could be useful for fostering conservation efforts. Therefore, clear understanding of the landscape context of agricultural matrix such as the ecological value of retained trees is essential**.** Retained trees are very common and well known features in various human modified landscapes (Manning *et al*., 2006). Retained trees can, therefore, be a source of propagules for future biodiversity conservation efforts (Elmaqvist *et al*., 2002). As such the ecological importance of retained trees in human modified landscape has not been given much attention.

Different species of *Eucalyptus* have been widely planted in many tropical countries, including Ethiopia, outside Australia (Pohjonen and Pukkala, 1990). The reason for the wide-spread early popularity and success of the different *Eucalyptus* species can be attributed to their fast growth, coppicing property, unpalatability of their leaves, and adaptability to a wide range of site conditions (FAO, 1981). However, intensive monocultures of *Eucalyptus* plantations have attracted much criticism because these species have been considered to deplete soil nutrients, decrease available water resources, suppressing ground vegetation by secretion of allopathic chemicals (Evans, 1992; Jagger and Pender, 2000; FAO, 2001) and favouring weeds instead of native plants (Ostertag *et al*., 2007; Rawat *et al*., 2009). For example, the regeneration of native tree species was poor in *Eucalyptus* plantations when compared to other exotic plantation species such as *Cupressus*, *Pinus* and *Grevillea* in southern Kenya (Thijs *et al*., 2014). However, several studies have shown that not all *Eucalyptus* plantations suppress vegetation, but instead natural recolonization by native trees can sometimes take place in the understory of *Eucalyptus* plantations (Parrotta, 1992; de Silva *et al*., 1995). For instance, one study reported 123 native species in plantations of *E. grandis* in Brazil, which was the same number as in the adjacent natural forests (de Silva *et al*., 1995). Another example is that the number of native trees as well as their cover could be higher in plantations of *Eucalyptus* than that of other plantation species, which could have a denser canopy preventing light to reach the understory (Parrotta, 1995). Thus, we need more studies addressing the mechanisms and conditions regulating the roles of *Eucalyptus* plantations for biodiversity colonization and conservation.

Ethiopia has one of the longest forest plantation histories in Africa (Pohjonen and Pukkala, 1990). To relieve the shortage of wood caused by extensive deforestation, *Eucalyptus* was introduced to the country in 1894-1895 (Pohjonen and Pukkala, 1990). Several studies in Ethiopia on *Eucalyptus* plantations indicate that they often promote the recruitment, establishment and succession of native woody species by functioning as foster ecosystems (Feyera Senbeta and Demel Teketay, 2001; Eshetu Yirdaw and Lukkanen, 2003; Mulugeta Lemenih *et al*., 2004; Kitessa Hundera, 2010). Also, it has been found that regeneration is higher in *Eucalyptus* plantations compared to, for example, plantations of *Cupressus lusitanica* and *Pinus* spp., a pattern attributed to more light in the *Eucalyptus* plantations (Eshetu Yirdaw, 2001; Mulugeta Lemenih *et al*., 2004; Hylander and Sileshi Nemomissa, 2009). Some authors have suggested that *Eucalyptus* plantations might be more efficient as a nurse site for native trees in high rainfall compared to low rain fall areas in Ethiopia (Biruk Ketsela, 2012). The regeneration of native plant species in *Eucalyptus* plantations is from dispersed seeds and not from the seed bank (Feyera Senbeta and Demel Teketay, 2001), highlighting the need for studies that take a landscape context into consideration.

In Ethiopia some studies have dealt with the ecological role of dominant and the characteristic forest species in a farmscape in dry Afromontane forest. Forexample, some studies have shown that the density of *Podocarpus falcatus* under *Eucalyptus* patches increased with canopy cover which show that it is shade tolerant (Feyera Senbeta and Demel Teketay, 2001; Mulugeta Lemenih *et al*., 2004).On the other hand *Juniperus procera* regenerates better in somewhat lighter conditions and light is not a limiting factor (Alemayehu Wassie *et al*., 2009; Legesse Negash, 2010). However, this species also regenerates in closed canopy forests indicating that it is shade tolerant (Demel Teketay, 1997; Alemayehu Wassie *et al*., 2009). Thus, the divergent views or results have informed the current species specific study.

This PhD thesis has three main sections. The first section presents the diversity of indigenous woody species in small-scale *Eucalyptus* plantations (patches) scattered across the farmscapes situated at different distances around the remnant Chilimo dry Afromontane forest in Central Ethiopia. Whereas the second part of the thesis focuses on whether or not the isolated *Podocarpus falcatus* and *Juniperus procera* species across the farmscape are the sources of diaspores for the colonization of *Eucalyptus* plantations, the last part deal with the diversity of herbaceous species in these patches. A total of 60 small *Ecucalyptus* plantations were studied.

1.1. Objectives

1.1.1. General objective

The general objective of this study was:

To understand what local, landscape and historical variables constrain or foster native woody and herb species regeneration in *Eucalyptus* plantations and the association of retained *Podocarpus falcatus* and *Juniperus procera* tree species to small patches of *Eucalyptus* in farmscape.

1.1.2. Specific objectives

The specific objectives of the study were as follows:

- To examine what factors determine the regeneration of native woody species in *Eucalyptus* plantations scattered across farmscape surrounding Chilimo remnant forest;
- To investigate the role dispersal syndromes for the colonization of *Eucalyptus* plantations by woody species;

- To examine what factors determine the colonization of *Eucalyptus* plantations by herb species ;
- To investigate the role of remnant natural forest as a source of propagules of herb species for colonizing *Eucalyptus* plantations;
- To examine what factors determine the regeneration of *Podocarpus falcatus* and *Juniperus procera* in *Eucalyptus* plantations; and
- To examine the role of local connectivity of the retained trees of *P. falcatus* and *J. procera* for the colonization of the *Eucalyptus* patches by these species.

1.2. Hypotheses

- Distant *Eucalyptus* patches accumulate less number of woody and herb species than close patches because of dispersal limitations.
- *Eucalyptus* patches with tall grass sward and less percentage cover of bare soil support high woody and herb species than patches with short grass swards and high percentage of bare soil because high grazing level will bring high bare soil cover and less height of grass sward. The height of grass sward and percentage of bare soil cover are indicators of level of disturbance.
- Old *Eucalyptus* patches harbour more herb species than young patches because of more time for colonization of propagules from the source.
- *Eucalyptus* patches with a relatively high canopy cover support less herb species than patches with low canopy cover because light penetration facilitates the growth of herb species.
- Remnant trees with large diameter size and high connectivity are the source of seeds for those regenerating under *Eucalyptus* patches because the tree with large size and high connectivity are expected to produce more seeds that reach more easily than those with small size and isolated from the patches.
- *Eucalyptus* patches with relatively large canopy cover support regeneration of *P. falcatus* whereas patches with low canopy cover facilitates the regeneration of *J. procera* because *P. falcatus* is a shade tolerant species and *J. procera* is light demanding.

2. Literature review

2.1. Ecological restoration

Ecological or ecosystem restoration is defined as the "process of assisting the recovery and management of ecological integrity, including a critical range of variability in biodiversity, ecological processes and structures, regional and historical context, and sustainable cultural practices" (SER, 2004). Even though, there are different schemes in the restoration of biodiversity, passive and active types are common strategies (Mulugeta Lemenih and Demel Teketay, 2004) (Fig.1). In passive restoration, natural succession is a key factor in the restoration process (McIver and Starr *et al.*, 2001). Such strategy might help to increase the number of species and improve species composition which in turn leads to biodiversity recovery (Pain, 2012). The main problem with passive restoration through natural succession is a relative longer time (Lamb, 1998) especially when site degradation is severe (Parrotta, 1997). Hence, these natural factors alone may not be sufficient to allow rapid biodiversity restoration and need to be supported by artificial intervention (i.e. active restoration). In the active restoration strategy restoration management techniques such as tree plantation and exclosure are the common practices (McIver and Starr, 2001; Morrisson and Lindell, 2010). Tree planting has become a common restoration strategy and is often successful in accelerating tropical forest recovery (Parrotta and Knowels, 2001; Lamb *et al.*, 2005).

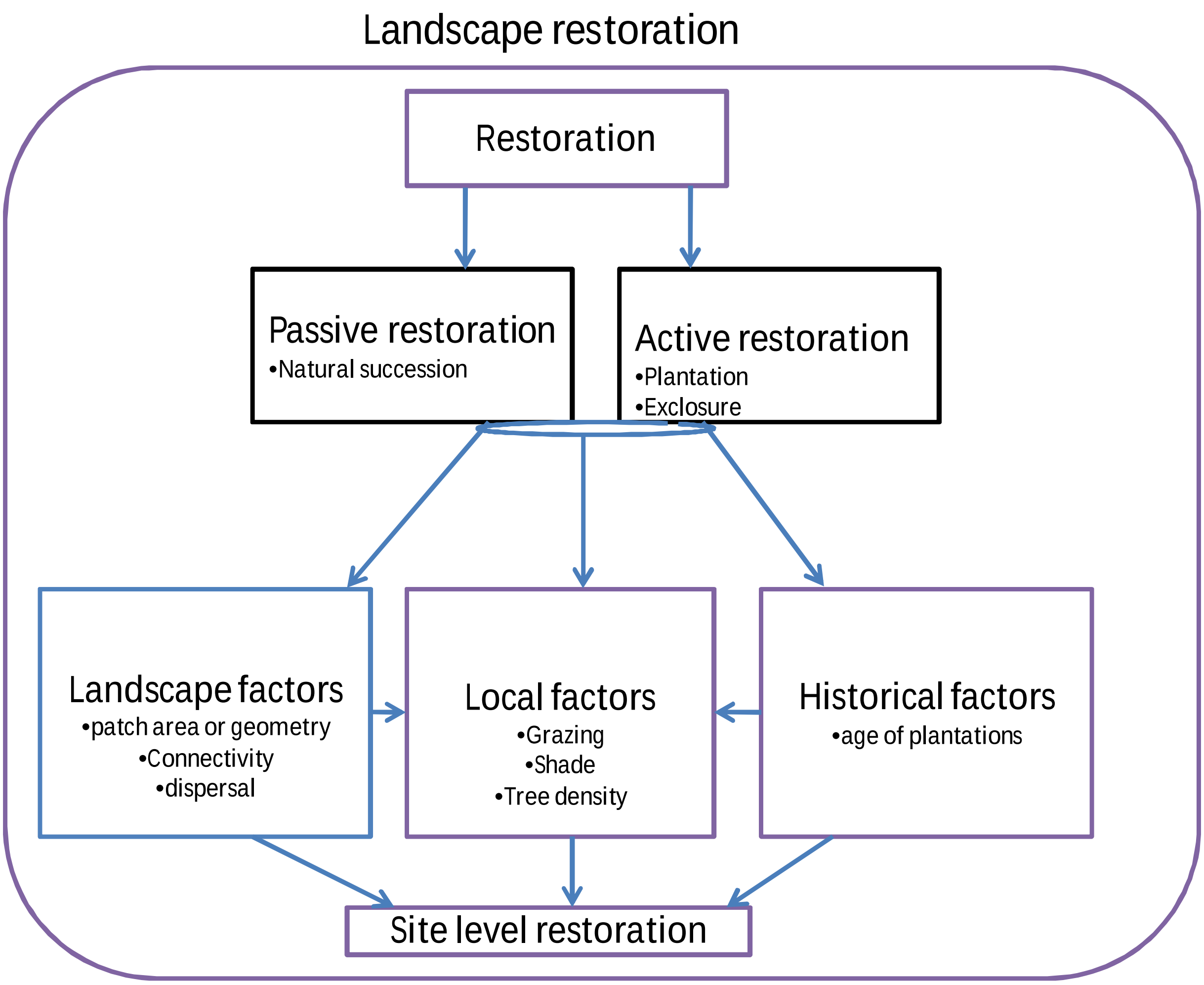

Figure 1. **Conceptual model of biodiversity restoration modified from Brudvig (2011).**

2.2. Forest plantations as restoration tool

The implication for the conservation of biodiversity in plantation forest is highly debated **(**Brockerhoff *et al*., 2008; Kanowiski *et al*., 2000). Forest plantations, especiallyexotic monoculture tree plantations, have been mainly established for wood and timber production (Sorley, 1999) and are thought to offer a less favorable habitat than natural forest (Harley, 2002). There is a common view that plantation forests are "ecological deserts" that are devoid of valuable habitat when compared with natural habitats (Bremer and Farley, 2010; Allen *et al*., 1995). However, several studies have shown that forest plantations can contribute to restoring of the native flora (Brockerhoff *et al*., 2008) (Fig. 2).

Tree plantations can play constructive role in connecting native plant biodiversity to natural forest remnants (Parrotta *et al*., 1997; Lindenmayer *et al*., 1999). For example, tree plantations could facilitate dispersal by attracting birds and mammals that helps regeneration of plants in degraded lands (Wunderele, 1997). Plantation could also help to improve local habitat conditions such as canopy and shading that in turn help to enhance regeneration processes (Lamb *et al*., 2005). For instance, soil fertility is improved since more organic matter is added from the decomposition of tree fall (Luggo, 1992; Parrotta, 1992; Mulugeta Lemenih, 2004).

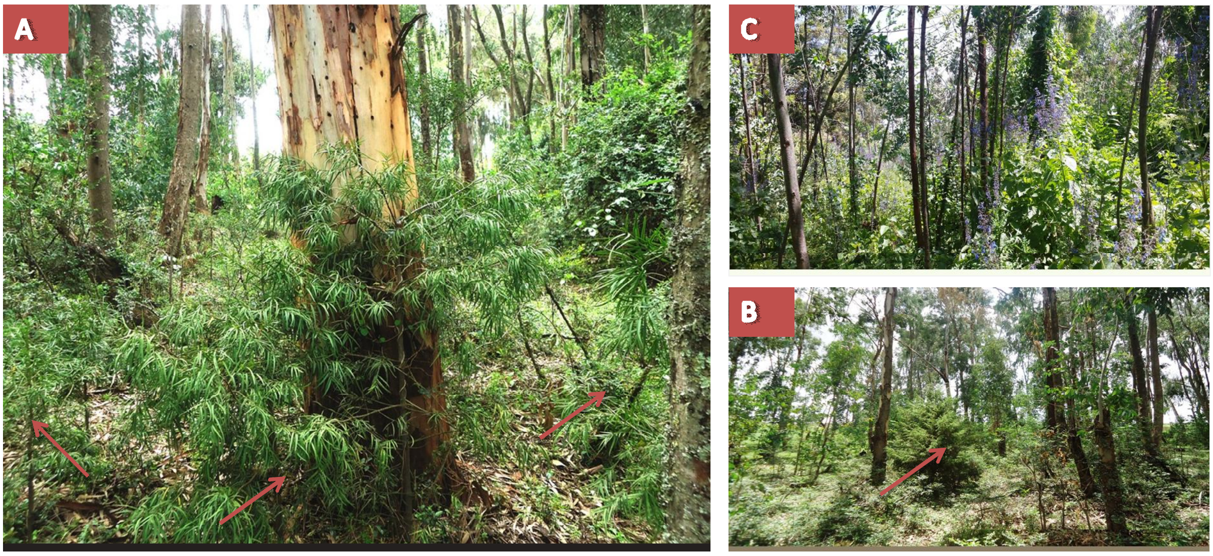

Figure 2. The regeneration of native plant species under *Eucalyptus* plantations in the study area: (A) Saplings of *Podocarpus falcatus*; (B) Sapling of *Juniperus procera*; and (C) herb species. ©Habte Telila

Eucalyptus plantations also attracted many criticisms including the argument that they cause degradation of land since they require plenty of water, that they drain sub-soil water and cause scarcity of water (Evans, 1992), and they are source of allelopathic chemicals that hinders the growth of understorey (Lisanework Nigatu, 2000).

Even though, the pre plantation landuse has an impact on the effect of plantation forests it is commonly accepted that natural forests are better habitat places for the regeneration of native forest species than plantation forests (Lindenmayer and Hobbs, 2004). Howeverwhen compared to farmscapes and barren landuses, plantation forests have higher conservation value (Brockerhoff *et al*., 2001). Moreover, the plantation of trees for the purpose of restoration would be more important if they are planted on the marginal and degraded sites rather than replacing the natural forest (Jagger and Pender, 2000; Hylander and Sileshi Nemomissa, 2009).

2.3. Factors impacting the importance of plantations as restoration tool

Several factors are known to impact the rate and trajectory of the restoration process, these include local habitat quality variables (Mathews *et al*., 2009) and factors related to colonization and availability of propagules (Wang *et al*., 2010) (Fig.1).

2.3.1. Local factors

Local habitat conditions that could be enhanced by plantations include changes in micro-climate that may become more favourable for forest plants, and changes in soil qualities (Feyera Senbeta and Demel Teketay, 2001; Harley, 2002; Carnus *et al*., 2006). The most effective influence of tree plantations on soils are the production of organic matter through litter fall, increased nutrient recycling, increasing rainfall infiltration, decrease of soil bulk density and an increase of soil biological activity (Lugo, 1997). Another factor which might have a great influence on regeneration success in plantations is grazing by livestock, which in fact could eitherfacilitative or hamper regeneration. It has been suggested that a light level of grazing can increase tree regeneration by removing competitive vegetation and fertilizing the soil by animal droppings (Carolina and Javier, 2001; McEvoy *et al*., 2006). On the other hand, a continuous and intensive grazing could prevent forest regeneration by soil compaction, trampling of tree seedlings and browsing of saplings (Opperman and Merenlender, 2000). Overgrazing on aboveground vegetation could mainly bring reduction of herb species richness and biomass in general (Roberts and Gilliam, 2003; Roberts, 2004; Chadioftu *et al*., 2009; Bergmeier *et al*., 2010).

2.3.2. Landscape factors

Landscape may not be always defined as an area having a large extent (1,000-10,000 of hectares). From an ecological point of view it could be considered as spatially heterogeneous composed of an interacting mosaic of ecosystems and encompassing populations of many species regardless of its size (Villard and Metzger, 2014).

The major focus of ecological restorations has been on understanding local factors. However, landscape factors such as dispersal limitation of individuals or propagules across the surrounding matrix can also constrain the restoration (Matthews *et al*., 2009) (Fig.1). Dispersal limitation, the absence of a species from a community due to lack of propagules entering a communitycan be a major constrain for the possibility of native biodiversity to establish in plantations (Stein *et al*., 2008). Seed dispersal through animals is predominant in tropical tree species (Howe and Smallwood, 1982). Hence, a successful colonization depends on, if and to what extent, different dispersal vectors will travel across the matrix between the seed sources and the focal restoration sites. Some studies indicate that seeds drop at short distances from native forests (Holl *et al*., 2000; Cramer *et al*., 2002). On the other hand there are also studies reporting that trees have established in plantations at several kilometres from the seed sources (Zanne andChapman, 2001; de Mendonça-Lima *et al*., 2014). Compared to open habitats, plantations of trees could be more attractive to seed dispersing birds flying across landscapes, since they could find perches and perhaps resources there (Luck and Daily, 2003). The impact of the behaviour of the dispersal agents could be illustrated by an example from de Mendonça-Lima*et al*., (2014), who found a higher frequency of animal-dispersed plants in plantations compared to in native forests.

A large proportion of plant species in tropics are adapted to dispersal by birds and/or mammals (Howe and Smallwood, 1982). Particularly, most Afromontane herb species have seeds that stick tightly to animals and are dispersed for long distances (Demel Teketay, 2005). Furthermore, some seeds are also dispersed by wind (Demel Teketay, 2005). The range of dispersal distance may vary based on the type of dispersal agents (Nathan, 2006). For example, dispersal by ants may travel a distance of less than 1m, whereas dispersal by wind covers a distance of greater than about 10 km (Corlett, 2008). Generally, it is reported that wind and vertebrate dispersed species have longer dispersal distance (Gomez, 2007; Vitoz and Eagler, 2007). Seed or fruit that are attached to different body parts of animals and those transported by humans (intentional and unintentional) are believed to be long distance dispersal modes (Shigesada *et al*., 1995).

2.3.3. Historical factors

The age of plantations could influence the regeneration of native flora (Ito *et al*. 2004; Soo *et al*., 2009). Older plantation stands provide better habitat for forest species than young stands because of increased spatial and vertical heterogeneity (Lugo,1997; Lamb *et al*., 1998), well-developed soil organic layers (Firn *et al*., 2007), increased dead wood on the forest floor which in turn improve litter layer (Bremer and Farley, 2010). For example, the indigenous species richness was significantly higher in older compared to younger stands of pine plantations in Switzerland (Brockerhoff *et al*., 2003).

2.3.4. *Eucalyptus* plantations in Ethiopia

Eucalyptus was introduced to Ethiopia from Australia in 1894/95 to overcome the shortage of fuel wood and construction timber and has been successfully established throughout the country (Von Breitenbach, 1961). Consequently, Ethiopia has one of the longest histories of *Eucalyptus* forest plantations development in Africa (Pohjonen and Pukkala, 1999). The reason for their wide spread occurrence in Ethiopia is mainly due to their fast growth, coppicing ability and less management requirements, the unpalatability of leaves, their adaptability to a wide range of site conditions and good economic returns (FAO, 1981).

Small-scale plantations are established mainly to satisfy household demands for wood and to generate additional household income from sales. For instance, in the Arsi highlands of central Ethiopia (Zenebe Mekonen *et al*., 2007), wood from grown by smallholder farmers contributes to 92% of the poles, 74% of the timber, 85% of the firewood, 40% of the charcoal, 83% of the posts and 91% of the farm implements used by a rural household (FAO, 2011). In some areas, high rate of return from plantations compared to other farm enterprises is leading to the conversion of croplands and grazing fields to *Eucalyptus* woodlots (Jagger and Pender, 2003)..

In Ethiopia, the ecological range of *Eucalyptus* is from a mean annual rainfall of 800mm up to over 2000 mm and altitude between 2000 - 3500 m asl. It can grow in a wide variety of soil and ground conditions from fertile to unfertile and degraded soil, from coarse to fine texture, and from dry to waterlogged conditions (Zenebe Mekonen et al., 2007). At present, forest plantations in Ethiopia are predominantly monocultures of *E. globulus* and they are estimated to cover about 93% of the total plantation area in the country (FAO, 1981) and the area coverage is about 894,240 ha (Million Bekele, 2011).

2.4. Retained trees in a farmscape

Different landuses could be found in an agricultural matrix that might have essential roles in the biodiversity conservation (Chazdon, 2003; Harvey *et al*., 2006). For instance, 59 trees or shrub species have been recorded in the home gardens in the study carried out in an agricultural landscape in south western Ethiopia (Hylander and Sileshi Nemomissa, 2009).

Global biodiversity conservation efforts have focused mainly on the establishment of protected areas (Zahawi and Augspurger, 2006), although majority of biodiversity are present outside of the protected areas (Powell *et al*., 2000). Thus, it is crucial to integrate biodiversity conservation and agriculture (Green *et al*., 2005; Fischer *et al*., 2008; Harvey *et al*., 2010). Therefore, a clear understanding of the landscape context of agricultural matrix such as the ecological value of retained trees is essential**.** Retained trees are very common and well known features in various human modified landscapes (Manning *et al*., 2006). Small-scale farmers retain isolated trees of *Podocarpus falcatus* and *Juniperus procera* in their cultivated fields for various reasons including timber, fodder, firewood and soil fertility (Abebe Yadessa *et al*., 2009). The main ecological function of the retained trees in a disturbed landscape is to preserve biodiversity (Manning *et al*., 2006; Hylander and Nemomissa, 2009), e.g. in the form of seed directly from the trees (Benayas *et al*., 2008) or indirectly from dispersal agents such as birds (Harvey *et al*., 1999). Retained trees can, therefore, be used as main source for future biodiversity restoration activities (Reis *et al*., 2010; Manning *et al*., 2006).

2.5. Use of exclosurs in Ethiopia

The major human related factor that threatens biodiversity in Ethiopia is land degradation (Temesgen Abebe *et al*., 2014). Although the objectives of restoration lacks clarity in Ethiopia (Habtemariam Kasa, 2014), water and soil conservation have been implemented to overcome the disastrous effect of land degradation (Abraham Abiyu *et al.*, 2011; Temesgen Abebe *et al*., 2012). Exclosure is the exclusion of degraded lands from human and animal interference for self sustaining (Aerts *et al*., 2009; Abraham Abiyu *et al*., 2011). Mainly two types of area exclosure management have been identified (Mulugeta Lemenih and Habtemariam Kasa, 2014.) The first one involves protecting enclosed areas against livestock and human interference and no additional management activities.This allows regeneration and ecological succession which inturn rehabilitates degraded forested areas (Wolde Mekuria *et al*., 2007). The second type, which is the most common, involves terracing, exclosure, and tree planting to speed up succession through the modification of microclimatic and soil conditions (Aerts *et al.*, 2009; Mulugeta Lemenih and Habtemariam Kasa, 2014). Conservation measures through area exclosure have been in use in a wide range of forest ecosystems from dry forests and woodlands to the sub-humid Afromontane forests (Mulugeta Lemenih and Habtemariam Kasa, 2014). Practice of exclosure started in 1980s in Ethiopia (Haile Getsesellasie, 2012). However areas under exclosure rapidly increased in different parts of Ethiopia in general and in Northern Ethiopia in particular. For example in 1996, there were only about 143,000 ha of exclosure in Ethiopia (Emiru Berhane *et al.*, 2006). However, in Tigray regional state alone the area under area exclosure reached 895,220 ha in 2011 (Wolde Mekuria *et al*., 2007). By the end of 2013, exclosures covered 1.54 million ha in Tigray (Muluberhan Gebresilassie, 2013) and 1.55 m ha in Amhara Regional State (Shimeles Damene *et al*., 2013). Most of exclosures failed to meet the designed objectives because of lack of clarity, at some places, it was reported that species disappeared in the past have been restored as a result of the exclosures. For instance, in some parts of eastern Tigray, species that had long disappeared from the areas (e.g. *Olea europaea* subsp. cuspidata and *Juniperus procera*) re-appeared (Wolde Mekuria and Ermiyas Ayenekulu, 2011). Moreover, inthe study conducted in the highlands of Tigray in exclosures, about 85 plant species where as only 25 plant species has been recorded in adjacent communal grazing lands. Furthermore it has been shown that the effectiveness of plantations of fast growing exotic tree species to restore degraded lands could be enhanced when assisted by exclosurs (Aerts *et al*., 2007; Wolde Mekuria *et al*., 2007).

2.6. Natural colonization of species

Colonization is fundamental in basic ecological questions such as succession and community dynamics as well as in studies of biological conservation and restoration (Sheffer *et al*., 2014), The successful colonization of a species in a site can have broad implications for the diversity and abundance of resident species, the structure of the ecosystem, and rates of ecosystem processes (Hobbs *et al*., 2009). Colonization can be viewed as the net result of processes starting from propagules production and ending in the survival to reproductive maturity of the colonist (Clark *et al*., 1998)

It is a combined function of landscape structure and species traits. Successful colonization mainly based on the availability of species in source habitats in the surrounding landscape (Matthews *et al*., 2009), the connectivity between the source and target habitat, degree of site degradation and the dispersal capacity of the species (Stein *et al*., 2008). Colonization could be possible through sprouting from root and stems, germination from soil seed bank, regeneration by long-lived shade tolerant seedlings and dispersal (Brunet, 2007). Colonization processes by dispersal across an unsuitable matrix (e.g., agricultural land) to a target habitat is facilitated by dispersal corridors such as plantation trees (Luck and Daily, 2003),

2.7. Traditional knowledge and restoration

For ecological restoration of degraded areas, it is vital to have the knowledge plant biology and ecology (Chazdon, 2003). Similarly the socio-economic and other interests of the local communities should be considered (Alexander *et al*., 2014). People are highly connected and in constant interaction with biodiversity, and through this interaction they developed sets of knowledge and practices called traditional knowledge (Hartman and Cleveland, 2014). Many ecological restoration schemes have failed (Chazdon, 2008), because they fail to address socio-economic factors that are needed to ensure the involvement of local communities (Sayer *et al*., 2004). Information from persons at the local level (local knowledge) is a key factor in designing biodiversity restoration related policies and regulations (Hartman and Cleveland, 2014; Mulugeta Lemenih and Habtemariam Kassa, 2014)

2.8. Descriptions of: *Podocarpus falcatus* and *Juniperus procera*

2.8.1. *Podocarpus falcatus* (Thunb.) Mirb.

2.8.1.1. Description

Podocarpus falcatus is an evergreen tree belonging to family of Podocarpaceae and is one of the two coniferous species in Ethiopia (Demel Teketay *et al*., 2011). It is a tree up to 46 m in nature with a long clean and cylindrical trunk. It is very attractive, evergreen, with dense canopy branches. Likewise, the combination of dark-green mature foliage with light green tips of the new growth gives the tree a distinctive appearance (Leggesse Negash, 2010). It is dioceous plant in which the female cones are solitary and hard ovoid to 2 cm, green with dull purple bloom. The 1-3 male cones (male cones) are axillary. The color is yellow to pinkish-purple. (Azene Bekele *et al*., 1993).

P. falcatus is classified as a "high class of soft wood", this could be the reason for its selective removal from the forest. It has also several ecological advantages, including provision of shade and nourishment for birds and small mammals. It also protects the soil from heavy erosion caused by stormy rainfall (Legesse Negash, 2010).

2.81.2. Distribution

P. falcatus is found in transitional rainforest and dry evergreen forest, Particularly, it is characteristic species of undifferentiated Afromontane forest, where it is one of the dominant species ("Podocarpus forest") or one of the co-dominant (e.g. in "Juniperus-Podocarpus forest"), often persisting in relic forest patches (Friis *et al.*, 2010). Frequent as single tree left in derived grassland or farmland in areas with sufficient rainfall (legesse Negash, 2010).

It prefers rich and dry soils. It is common in the altitudinal range of 1500-3000 m asl., mean annual temperature of 13-20◦C and mean annual rainfall of 1200-1800 mm.. Different species respond differently to different intensity of canopy opening or gap formation. *P. falcatus* is shade tolerant species (Demel Teketay, 2011; Legesse Negash, 2010).

The species is native to Burundi, Democratic Republic of Congo, Eritrea, Ethiopia, Kenya, Lesotho, Malawi, Mozambique, Rwanda, South Africa, Sudan, Swaziland, Tanzania, Ugnda and Zimbabwe. However, it is exotic to India.

2.8.2. *Juniperus procera* Hoechst. ex Endl

2.8.2.1. Description

Juniperus procera is an evergreen timber tree that belongs to family Cupressaceae. It is an evergreen tree about 40 m straight trunk, although often fluted. A pyramidal shape when young. It is a dioecious species and is wind pollinated. The male cones of the tree are small, rounded, solitary, and yellowish. They occur in short brannchlets and are terminal. The female cones are berry-like, rounded and, upon ripening, become fleshy and soft (Legesse Negash, 2002).

The species is moderately shade tolerant when young and moderately intolerant when older. (Legesse Negash, 2010). The wood of *J. procera* mainly used for construction, firewood, poles, fencing and manufacturing pencils. Normally resistant to termites, but very sensitive to fire and frost (Orwa *et al.*, 2009; Legesse Negash 2010).

Natural regeneration is good where there is a good mix of reproductively mature male and female trees, and where the moisture and light requirements are favorable; the species appears to regenerate well in *Eucalyptus* plantations (Legesse Negash, 2010).

2.8.2.2. Distribution

J. procera is a species of dry Afro-montane forests often reaching 30-50 m high. Biophysically it prefers an altitudinal range between 1750 and 2500m asl. and mean annual rainfall of 400-1200mm. It prefers dry soil. The secies usually grows in mountainous areas and rocky sites (Tamirat Bekele, 1994: Friis *et al*, 2010; Legesse Negash, 2002).

It is native to Democratic Republic of Congo, Djibouti, Eretria, Ethiopia, Kenya, Lesotho, Malawi, Mozambique, Rwanda, Saudi Arabia, Somalia, Sudan, Swaziland, Tanzania, Uganda, Yemen and Zimbabwe. However, it is exotic to Australia, India and South Africa. (Friis *et al*, 2010; Legesse Negash, 2002).

3. Materials and Methods

3.1. Study area

The study was conducted in the farmscape of central Ethiopia surrounding the remnant Chilimo forest, which is located about 90 km west of Addis Ababa (Fig. 3; 9°04' 37" N and 38°09' 50" E). Chilimo forest represents the undifferentiated Afromontane forest of the dry evergreen montane forest and grassland complex type typically dominated by *Juniperus procera* and *Podocarpus falcatus* (Tamirat Bekele, 1994; Friis *et. al*., 2010). Some other frequent plant species in the forest are: *Prunus africana*, *Olea europaea* subsp.*cuspidata*, *Hagenia abyssinica*, *Satureja paradoxa*, *S. nilotica*, *Hypoestes forsskaolii* and *Geranium arabicum*. Out of 193 vascular plant species recorded in this area, 63% (122 species) of them were herbs including grasses and 37% (71 species) were woody plants (Teshome Soromessa and Ensermu Kelebessa, 2014). There used to be a saw mill in the forest around 1937-1968 (Farm Africa 1996), but the forested area has decreased in size even after protection in 1968. The area of the forest in early 1980s was said to be 22,000 hectares (MOA, 1982) which shrunk to 6,000 hectares in the 1990s because of agricultural expansion (Farm Africa, 1996). The altitudinal range of the forested area is from 2170-3050 m asl.

Chilimo forest is surrounded by vast areas of agricultural land (Fig. 3). The altitudinal range of this area is 2300-3200 m.asl. The inhabitants are mainly Oromo, but there are also a small number of other ethnic groups, who immigrated to work in the saw mill and settled permanently around the forest (Farm Africa 1996). The study area receives rain for five months (May – September) but the peak is in July. The average yearly rainfall is approximately 1093 mm in the study area (NMSA, 2014).

A)

B)

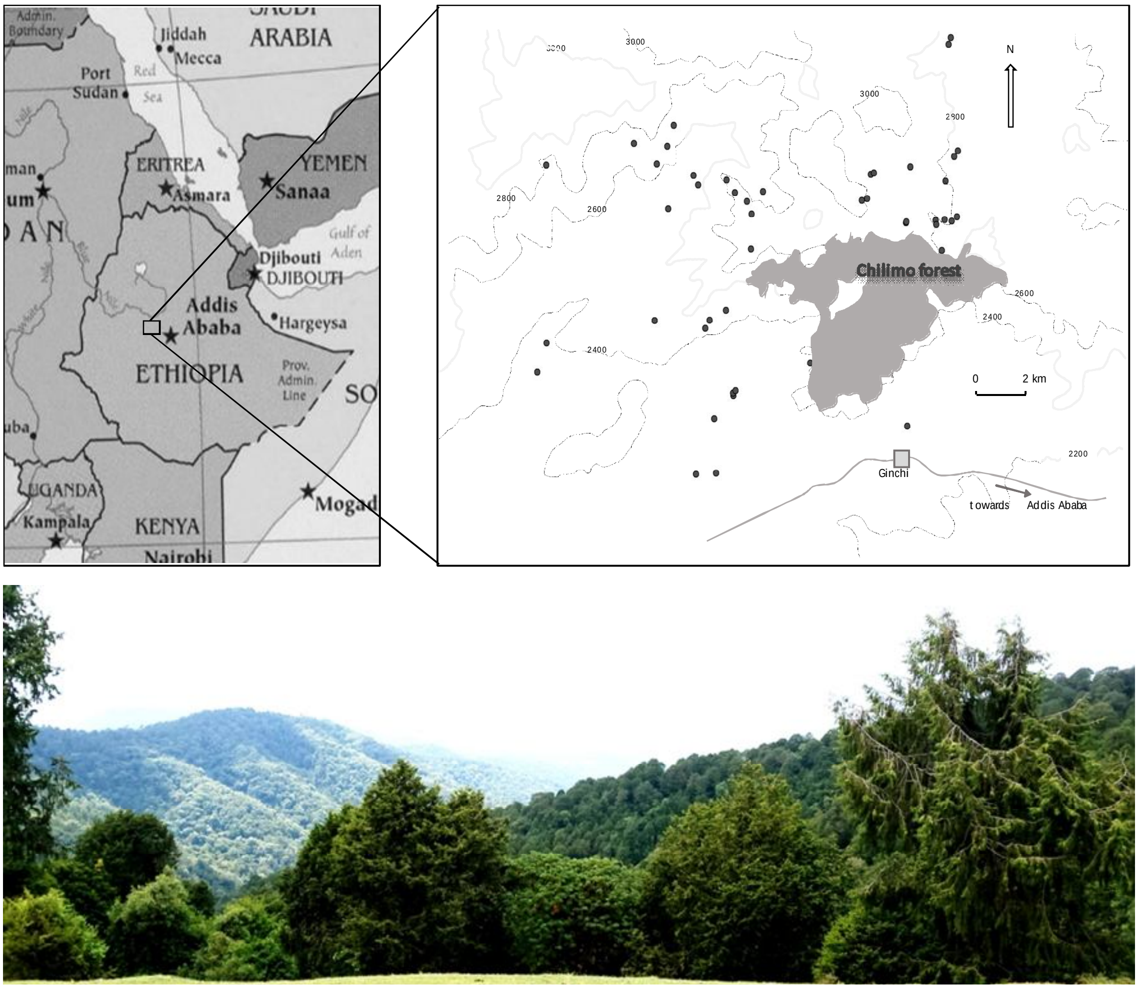

Figure 3.The study area; (A) Distribution of investigated *Eucalyptus* plantations patches (black dots) surrounding Chilimo forest in central Ethiopia; (B) Photo of part of Chilimo forest. ©Habte Telila

The farmscape is characterized by an open landscape, a very low tree cover, a mosaic of small fields, home gardens, grazing lands, wetlands and interspersed small plantations of exotic trees (Fig. 4). The trees in the plantations are used for fire wood, as live fences, local construction activities and as a source of income (Farm Africa, 1996).

Figure 4. The farmscape variegated with small *Eucalyptus* patches around the Chilimo forest. ©Habte Telila

3.2. Study design and data collection

From a Google Earth picture (from 2012) a large number of plantation patches in the landscape around Chilimo forest were selected. Reconnaissance surveys were made in the selected sites to record the trees and shrubs and understand the types of plantations. The vegetation consisted of *Eucalyptus camaldulensis, E. globulus* and *E. saligna* and a few of *Cupressus lusitanica*. I have All *Cupressus* sites were omitted inorder to get a more homogenous dataset and also because they were few in number in the farmscape. Instead 60 patches of *Eucalyptus* plantations ranging in size from 0.5-1.75 ha were selected with varying in distance to the continuous forests from 100 m to 12 km. Before the final selection it was checked that there were no strong correlations between size of the patch, distance from the forest and altitude (size vs. distance r=-0.03, size vs. altitude r=0.13, distance vs. altitude r=0.12).

3.2.1. Data of woody plant species

Data on native trees and shrubs in these stands were collected between May and June 2013 from all sites by carefully walking up and down each patch, thus covering the whole area. All tree and shrub species were identified and counted. All individuals were classified assaplings (height <2 m), juveniles (height 2-3 m) or mature trees (height >3 m).

3.2.2. Presence/absence of non-grass herbs

Data on the presence/absence of non-grass herbs in these stands were collected between September and October 2015 from all sites by carefully walking up and down, thus covering the whole area. Grasses were not included since their height were used as an indicator of grazing level and to avoid introducing errors into the dataset through potential misidentification in such a highly grazed area. When identification proved difficult in the field, specimens were collected for later identification at the National Herbarium of Ethiopia, Addis Ababa University.

A plot size of 100 m x 50 m in four different places and altitudes in the Chilimo forest were used to record presence/absence of the herbs. This was to investigate which species are out in the landscape from the natural forest.

3.2.3. *Eucalyptus* trees growth variables

In each site, *Eucalyptus* trees growth variables related to the size and density of the *Eucalyptus* trees and the disturbance level on the forest floor were collected. The height and diameter at breast height (dbh) of five representative *Eucalyptus* were measured; the height with a hypsometer and the dbh with a caliper. Canopy covers of the *Eucalyptus* trees were estimated in percentages. The level of grazing was recorded by measuring the height of the grass sward and by estimating the percentage cover of bare soil, i.e. high grazing level may bring high bare soil cover and less height of the grass sward. For the calculation of the area of each patch as well as the distance in metre to the continuous forest edge Arc GIS (ArcGIS v.10.2, Esri, Redlands, California) was used. Although the plantations show some variations in the height of the trees and size of the patch they differed more in grazing, distance from the source and elevation. For example, the elevation varied from 2300 to over 3000 m asl and the height of grass sward from zero to 60 cm (Table 1).

3.2.4. Dispersal Agents

For each tree and shrub species found data about its likely dispersal agent were extracted from the Flora of Ethiopia and Eritrea and other literature sources (See Appendix 1). Each species was assigned to one of five main dispersal syndromes: (1) birds, (2) both birds and mammals, (3) mammals, (4) wind and (5) water. Species found in Chilimo forest was also assigned but not in the plantations to one of the dispersal syndrome including one new syndrome: unspecified dispersal (Appendix 1).

3.2.5. Age of *Eucalyptus* patches

The age of *Eucalyptus* patches was estimated by using Aerial photos between 1988-2014 (EMA, 2014) stratifying then in to three periods (1988-1997; 1998-2007; 2008-2014). The period before 1988 was not included because no *Eucalyptus* patches were proved present. This variable was not used in analysis of the first part (woody species) of the study.

3.2.6. Biological attributes of herbs

For each herb species collected, data about its likely dispersal syndrome was assessed by carefully examining published information in the Flora of Ethiopia and Eritrea, morphology of seeds from the Herbarium specimens, own collections and other literature sources (Appendix 4). For instance, criteria such as lightweight pappi or wings as an indicator of dispersal by wind (anemochory), barbs and hooks indicating attachment to animals (exozoochory) and fleshy pulp pointing to endozoochory were used. Hence, each species was assigned to one of four main dispersal syndromes recognized in this study: (1) exozoochory (2) endozoochory (3) autochory, (4) anemochory. Appendix 5 gives the list of the herb species collected from Chilimo forest.

3.2.7. Data of retained trees

In addition to the data recorded in the first part of this thesis (woody species), the absolute locations of P. *falcatus* and *J. procera* across the farmscape were recorded by using a hand-held GPS in radii of 500 m in all directions of the *Ecucalyptus* patches. The distances of these retained trees to the *Eucalyptus* patches of interest were measured by using ArcGIS software. A circular plot area of 0.78 km^2 (radius=500 m) that was placed surrounding the *Eucalyptus* patches in the landscape (Fig.5) was used. The stem density of *P. falcatus* and *J. procera* was calculated and recorded within the sample plots. All individuals of *P. falcatus* and *J. procera* were classified as: saplings (height <2 m), juveniles (height 2-3 m) or mature trees (height >3 m). Again height was measured by a hypsometer and the dbh by a caliper.

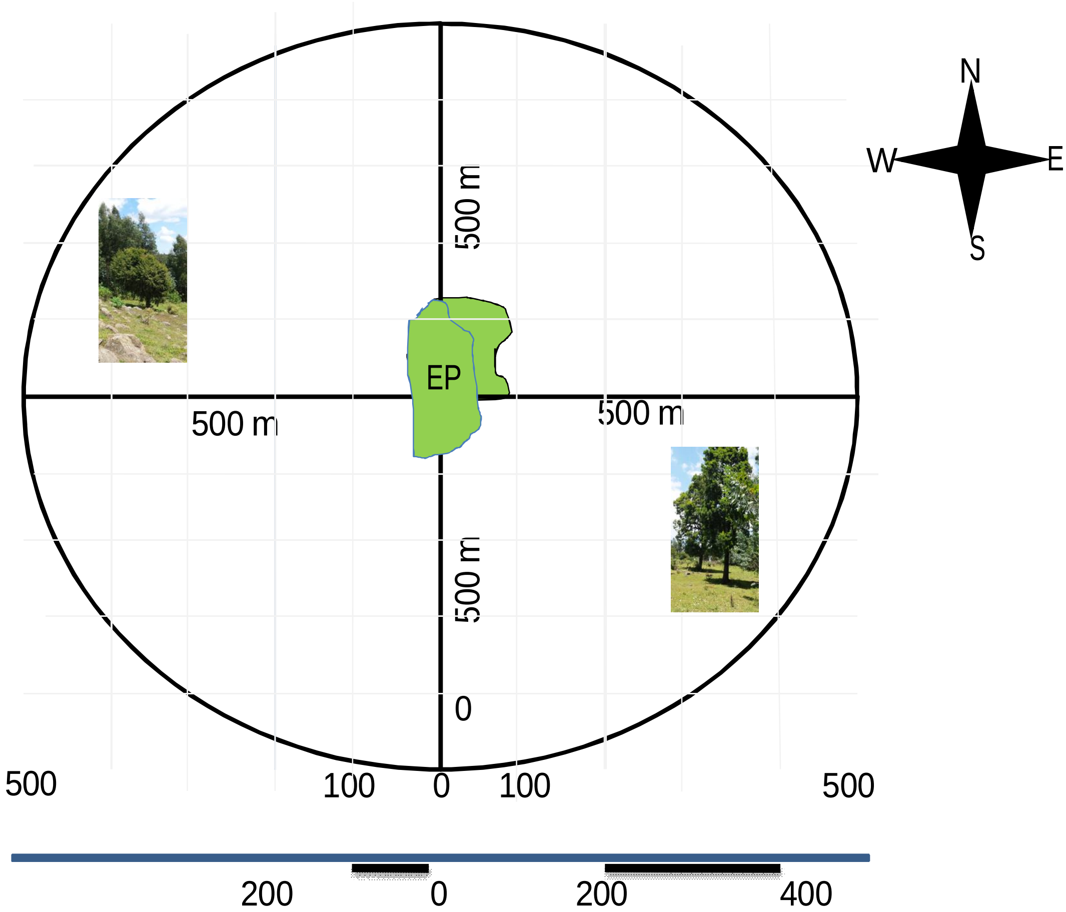

Figure 5.The sampling design used to sample the remnant trees of *P. falcatus* and *J. procera* around *Eucalyptus* plantations. The presence of the two trees was assessed in the radii of 500 m in all directions.

3.3. Statistical analyses

3.3.1. Woody species

Generally the following variables were chosen as explanatory variables for all the major analysis: (1) distance from the continuous forest (log-transformed), (2) elevation, (3) patch size, (4) height of the grass sward, (5) height of the *Eucalyptus*, (6) canopy cover (7) mean stem size of *Eucalyptus* and for the analysis including the retained trees of *P. falcatus* and *J. procera* (8) the age of the *Eucalyptus* patches and (9) local connectivity index (see Table 1).

3.3.1.1. Density

The number of individuals of each size class was divided by the area of the patch to get a measure of density (individuals per hectare) of tree species The influence of both local and landscape factors on the density of all size classes separately as well as the total density was analyzed.

Density was analysed with linear models and model selection was performed by stepwise selection using AIC (Akaike Information Criterion) and retaining the model with the lowest AIC-value (Oksanen *et al*., 2013). Moreover, the residual plots of the final model were inspected to validate the model assumptions (normality and homoscedasticity). Moreover; the densities of three selected characteristic forest species of the study area were also analyzed.

3.3.1.2. Species richness

The relationships between these explanatory variables and the woody species richness of the patches were analyzed with a General Linear Model (GLM) with a Poisson distribution. A backward model selection was performed until only significant variables remained (at $p<0.05$). Eventually, the residual plots of the final model were inspected to validate the model assumptions (normality and homoscedasticity).

3.3.1.3. Species composition

A two-dimensional non metric multidimensional scaling (NMDS) ordination was conducted to look at the overall pattern of species composition and used the *envfit* function to explore the relationship between the measured explanatory variables (Oksanen, 2015). The fitting of the explanatory variables to the NMDS result was performed with 1000 permutations.

3.3.1.4. Number of species with distance

To evaluate how far from the natural forest that different woody species had dispersed and established, presence/absence data of each species in each patch were used in addition to separate logistic regression models with distance as explanatory variable. The slopes of the logistic regression indicate how strongly dispersal limited a species is. A value around 0 shows that the species is equally common far out from the forest edge as it is close to the forest, while a strong negative value denotes dispersal limitation. GLM-models with binomial error structure for each species were run separately, but included only species that occurred in six or more patches, which represent 10% of all sites. Slope scores for groups of species with different dispersal syndromes were compared with an ANOVA. All data were analyzed using the statistical R software (R Core Team 2012, v. R 2.15.2, Vienna, Austria). For the ordination the package *vegan* in R was used (Oksanen, 2015).

Since there is no obvious grouping in the NMDS ordination and due to a gradual decrease in number of species with distance. It was notfound meaningful to do a cluster analysis. However, in order to further describe how the pattern of species composition varied along the gradient of distance from the main forest the 18 closest sites (<3 km from the main forest) with the 20 furthest sites (>7 km from the forest) were compared with an indicator analysis. This analysis thus complements the logistic regression analysis of single species responses to distances. For this analysis I used the package *labdsv* and did an analysis for presence/absence data. The mean altitude for the group close to the forest was 2724 m asl (range 2336-3050) and for the group far from the forest 2813 m asl (range 2376-3092).

To illustrate the distribution pattern along the distance gradient, the species-by-site matrix by distance to the main forest and frequency of the species were sorted further. If there would have been a perfect nestedness all the occurrences would be in the left corner of the table.

3.3.2. Retained trees of *P. falcatus* and *J. procera*

To investigate to what extent the retained trees of *P. falcaus* and *J. procera* surrounding the *Eucalyptus* plantations are important source of diaspore for those regenerating inside the *Eucalyptus* patches, the connectivity measures for each patch were calculated using the following formula:

$$\boldsymbol{S} = \sum_{i=1}^{n} \exp\left(-\propto \mathrm{di}\right) Ai$$

Where S= connectivity measure

$d_{i=}$ the distance (m) between *Eucalyptus* patch and the remnant trees i

A_i= diameter at breast height (cm) of the remnant trees i

α= describes the dispersal ability of the species, set to -1

S is summed over all trees (n)

For each patch the values of the connectivity measure were extracted and used them as one of the predictor variables listed above.

The relationships between the explanatory variables and the presence/absence of *P. falcaus* and *J. procera* inside the *Eucalyptus* patches were analyzed with a General Linear Model (GLM) with binomial error. Stepwise forward selection of explanatory variables was performed according to Akakie Information criterion (AIC). Eventually, the residual plots of the final model were inspected to validate the model assumptions (normality and homoscedasticity).

3.3.3. Herb species

To investigate which species colonized the *Eucalyptus* patches from the natural forest and from the farmland or grazing land, the herb species recorded under *Eucalyptus* were sorted as forest and non forest from the values of an Indicator Species Analysis. Indicator species analysis was ran from the presence/absence data and using the difference between the indicator values to decide the species in either of the two groups (i.e. Forest and non-forest). Those that had negative values were grouped as forest herb species and positive values to non-forest herb species (Table 2).

The relationships between the explanatory variables and the forest and non forest herb species richness were analyzed, respectively, with a General Linear Model (GLM) with a Poisson distribution. A backward model selection was performed until only significant variables remained (at $p<0.05$). Eventually, the residual plots of the final model were inspected to validate the model assumptions (normality and homoscedasticity).

A two-dimensional non metric multidimensional scaling (NMDS) ordination was conducted to look at the overall pattern of species composition of herbs recorded in *Eucalyptus* plantations and used the *envfit* function to explore the relationship between the measured explanatory variables (Oksanen, 2015). The fitting of the environmental variables to the NMDS result was performed with 1000 permutations.

4. Results

4.1. Colonization of *Eucalyptus* plantations by woody species

4.1.1. Explanatory variables

The general pattern was that both local and landscape factors affected the density and species richness of the woody species plants, while the species composition was mostly affected by landscape factors. The environmental variables are given in Table 1.

Table 1.Explanatory variables (environmental and growth variables) in the 60 small *Eucalyptus* plantations embedded in an open farmscape in Central Ethiopia.

Variables	Average	Range
Elevation (m asl)	2739	2322-3092
Size (ha)	0.8	0.51-1.3
Distance from forest (km)	5	0.1-12.2
Height of the *Eucalyptus* (m)	11	6-19
Height of grass sward (cm)	19	0-60
dbh, diameter at breast height (cm)	24	18-33
Canopy cover (%)	58	50-75
Bare soil cover (%)	44	10-75

4.1.2. Density and size classes

Within the 60 investigated *Eucalyptus* patches, 1571 individuals of native woody plants were recorded. Out of this 44% were adults (>3 meters), 17% juveniles (2-3m), and 39% saplings (<2 m). The average density of woody species was 11 per hectare (range 0-225 individuals per ha). The density of individuals of all size classes increased with increasing height of the grass sward (indicating less grazing) ($p< 0.001$), whereas density of saplings decreased with increasing distance from the continuous forest ($p=0.005$) (Table 2). Total density was also lower in more shaded plantations with high canopy cover ($p=0.008$) (Table 2a).

The total density of *Juniperus procera* decreased with distance from the forest ($p=0.004$). The total density of *Podocarpus falcatus* decreased with elevation ($p=0.003$), but increased with canopy cover (in contrast to the total density) ($p=0.013$) (Table 2b).

Table 2. The final linear models (lowest AIC, see Methods section) for (A) woody plant density of each size class (Adults >3m, Juvenile 2-3 m and Saplings <2 m) in the 60 plantations and(B) the total density versus the measured explanatory variables for three known characteristic forest species of the study area.

A					
Models	Variables	Estimates	Adjusted R^2	AIC	*p*-values
Total density			0.47	390	
	Height of the grass sward (cm)	1.76			<0.001
	Canopy cover (%)	-1.24			0.008
Density of Adults			0.24	270	
	Distance to forest (km)	-2.71			0.033
	Height of the grass sward (cm)	0.31			0.004
Density of Juvenile			0.28	180	
	Height of the grass sward (cm)	0.05			<0.001
Density of Saplings			0.36	297	
	Distance to forest (km)	-4.72			0.005
	Height of the grass sward (cm)	0.40			0.006
B					
Juniperus procera	Distance (km)	-4.46	0.17	251	0.004
Olea europaea subsp. *Cuspidate*	Basal area (m^2/ha)	0.43	0.37	65	0.068
Podocarpus falcatus	Elevation(m asl)	-0.01	0.18	210	0.003
	Canopy (%)	0.23			0.013

4.1.3. Species richness

A total of 55 species of woody plants (trees and shrubs) were recorded; 77 % of this is woody found in the remnant Chilimo natural forest (Appendix 1 and 2). The number of native woody plant species in the *Eucalyptus* patches decreased with distance from the forest from around 10 species at the closest distances to less than 5 species at ten kilometre out into the landscape (GLM, Z=-4.32, p<0.001). The number of species also decreased with elevation (GLM, Z=-2.55, p=0.01). However, with increased height of the grass sward the number of native woody plant species increased from less than 5 species in a patch with grass swards less than 10 cm up to more than 10 species in patches with grass swards higher than 40 cm (GLM, Z=3.65, p< 0.001) (Table 3; Fig. 6).

Table 3. Results from the final model of a General Linear Model with Poisson error structure on the number of native woody plant species in *Eucalyptus* plantation patches as a function of explanatory variables. (Distance was log-transformed)

Variables	Estimate	SE	Z	*P*
Intercept	3.25	0.53	6.07	<0.001
Distance to forest (km)	-0.06	0.01	-4.32	<0.001
Elevation (m asl)	-0.01	0.01	-2.55	0.01
Height of grass sward (cm)	0.01	0.01	3.65	0.001

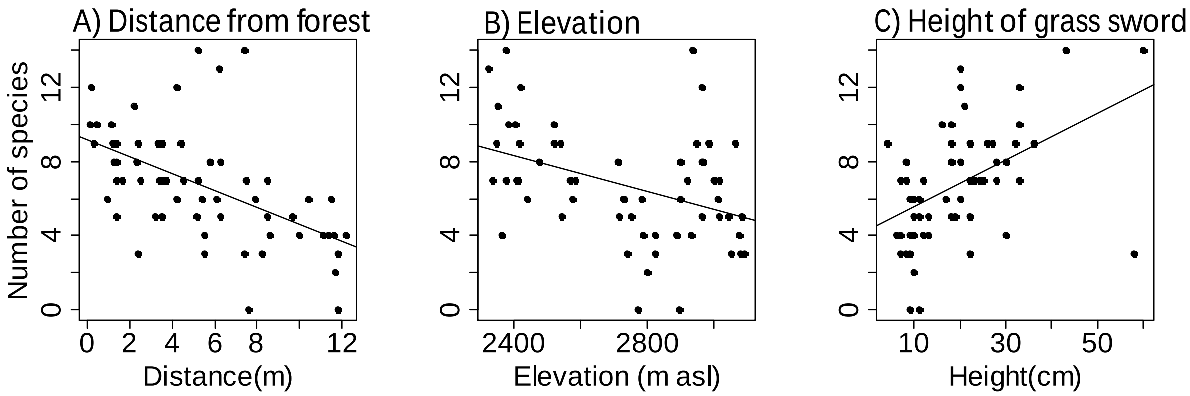

Figure 6. Number of native tree and shrub species as a function of (A) distance to a continuous forest, (B) elevation and (C) height of the grass sward (that indicate the density of grazing) in *Eucalyptus* plantation patches surrounding Chilimo forest, Central Ethiopia. I added linear trend lines to guide the reader. For statistical analysis, see Table 2

4.1.4. Species composition

Species composition of native woody plants was significantly related to elevation (r^2 =0.56, p<0.001) and distance (r^2 =0.22, p<0.001) (Fig. 7) from the forest. However, local factors such as height of the grass sward did not correlate with the gradients of turnover in species composition (p=0.064). An indicator species analysis showed that more species had a higher indication for close sites compared to far sites from the forest (Table 4). The species with highest difference in indicator value, and thus with highest affinity to the border of the forest, were *Acacia abyssinica, Olea europaea* subsp. *cuspidata,* and *Osyris quadripartita.* Among the few with a slightly higher indicator value at the far sites was *Discopodium penninervum.* The pattern was similar for the analysis with abundances and presence/absences. However, sorting of the species-by-site matrix by distance showed that the matrix had a poor fill and not any clear pattern of nestedness (Appendix 3).

Table 4. The species occurring in the 60 sites included in the indicator species analysis (0-2 km and 8-12 km). Difference in indicator value denotes the difference between the groups "Close" and "Far sites. The p-values are from a Monte Carlo test. The table is sorted from the largest positive to the largest negative difference in indicator values. Only species which occurred on ≥ 6 sites are included in this table (See Appendix 1 for a full species list).

Species	Occurrences	Difference in indicator value	*P* value	Slope
Olea europaea subsp. *cuspidate*	26	0.44	0.02	-0.18
Osyris quadripartite	6	0.28	0.05	-0.59
Acacia abyssinica	17	0.22	0.21	-0.18
Maytenus undata	23	0.22	0.31	-0.04
Podocarpus falcatus	15	0.17	0.48	-0.15
Maytenus gracilipes	7	0.17	0.34	-0.32
Myrica salicifolia	10	0.17	0.34	-0.13
Premna schimperi	8	0.17	0.22	-0.21
Solanecio gigas	13	0.17	0.5	-0.12
Croton macrostachyus	13	0.11	0.65	-0.12
Vernonia auriculifera	6	0.11	0.6	-0.15
Clausena anisata	7	0.06	1	-0.04
Dombeya torrida	9	0.06	1	-0.03
Dovyalis abyssinica	13	0.06	1	-0.06
Galiniera saxifraga	6	0.06	1	-0.06
Launea taraxacifolia	6	0.06	1	-0.1
Prunus Africana	11	0.06	1	-0.06
Rhus glutinosa	7	0.06	1	-0.07
Rumex nervosus	7	0.06	1	
Carissa spinarum	10	0	1	-0.04
Hagenia abyssinica	12	0	1	0.03
Maesa lanceolata	8	0	1	
Myrsine melanophloeos	23	0	1	-0.04
Apodytes dimidiata	10	-0.06	1	0.01
Buddleja polystachya	13	-0.06	1	0.01
Juniperus procera	23	-0.06	1	-0.01
Rosa abyssinica	10	-0.06	1	-0.03
Hypericum quartinianum	8	-0.11	0.6	-0.01
Maytenus arbutifolia	23	-0.11	0.61	0.02
Discopodium penninervum	6	-0.22	0.2	0.54

4.1.5. Dispersal syndromes and dispersal distances

Among the 55 species, dispersal by birds was the most frequent dispersal syndrome followed by mammal dispersal (Appendix 1). There was a large variation in how far different species dispersed as judged by the differences in logistic regression slopes (see Table 4; Fig. 7 for a couple of examples). There were no significant difference in how far different groups dispersed seeds as judged by their differences in slope from the logistic regressions (ANOVA, F=1.2, p=0.29; Fig. 8).

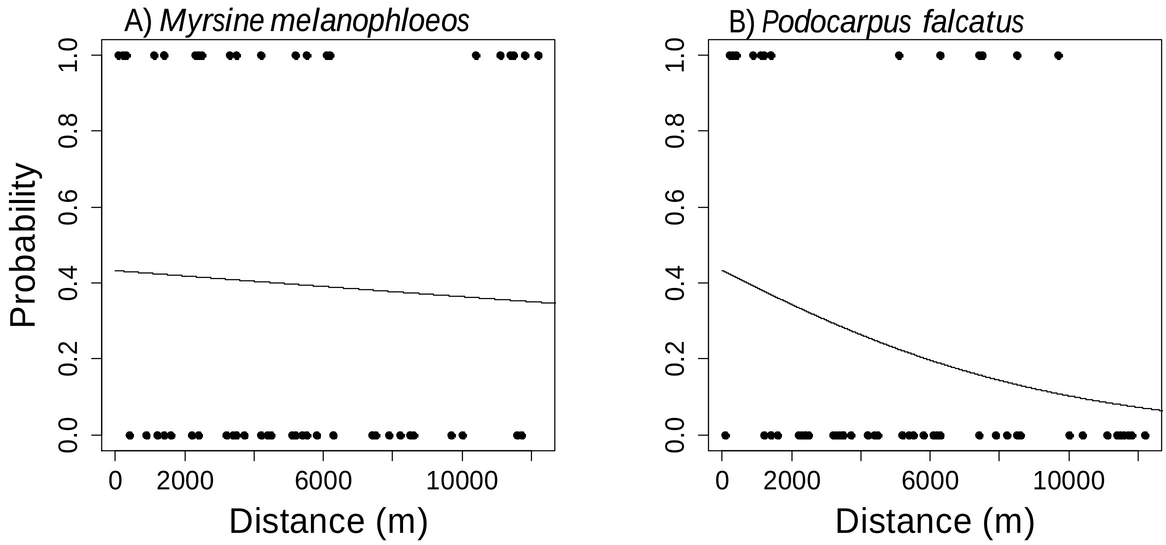

Figure 7. Examples of native plant species with (A) good long dispersal capacity (shallow slope of the logistic regression) and (B) poor dispersal capacity (negative slope in the logistic regression).

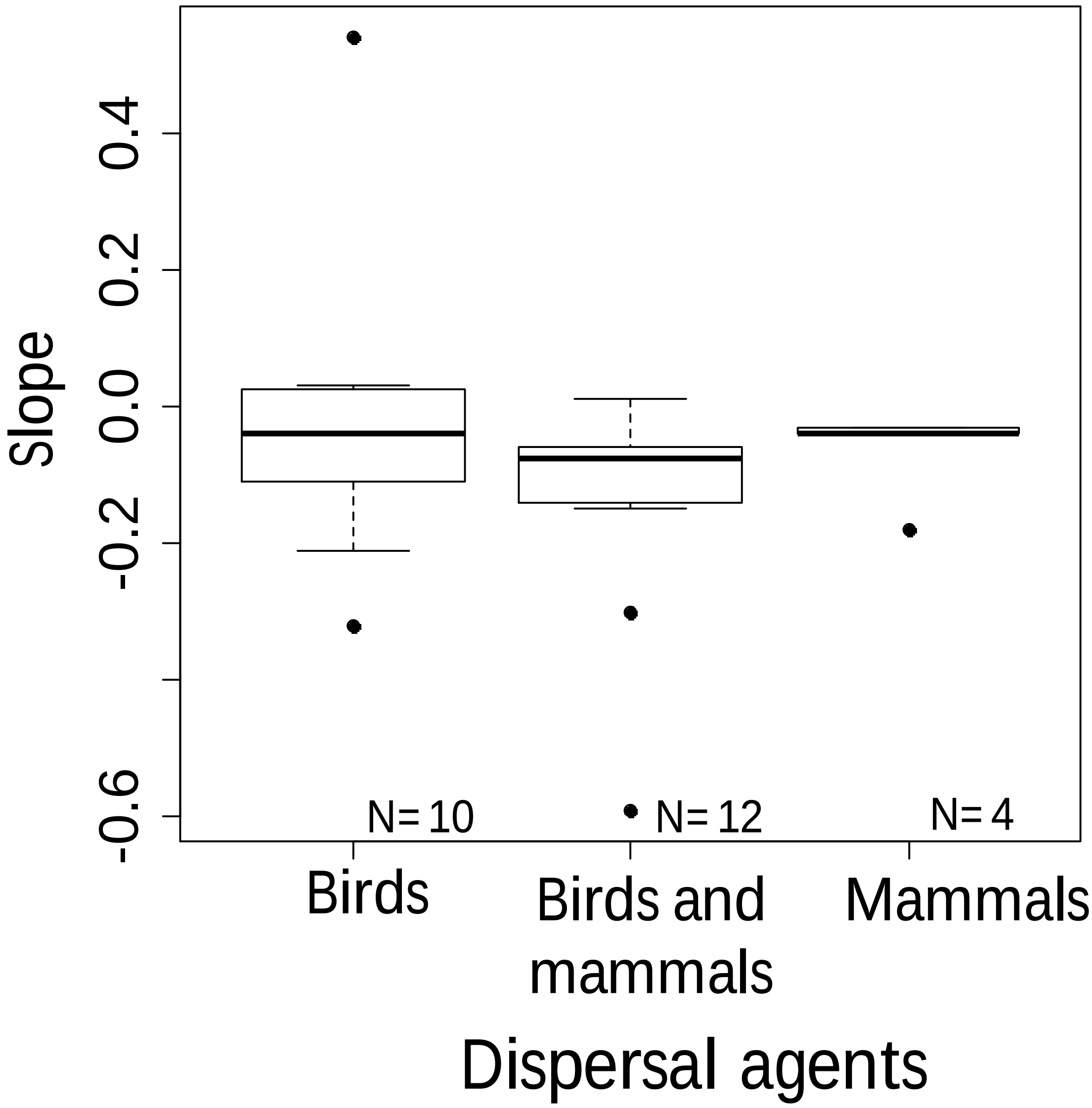

Figure 8. Dispersal capacity of species dispersed by different dispersal agents. Dispersal capacity is expressed as the slope of a logistic regression. N= number of plant species in the different dispersal agent groups.

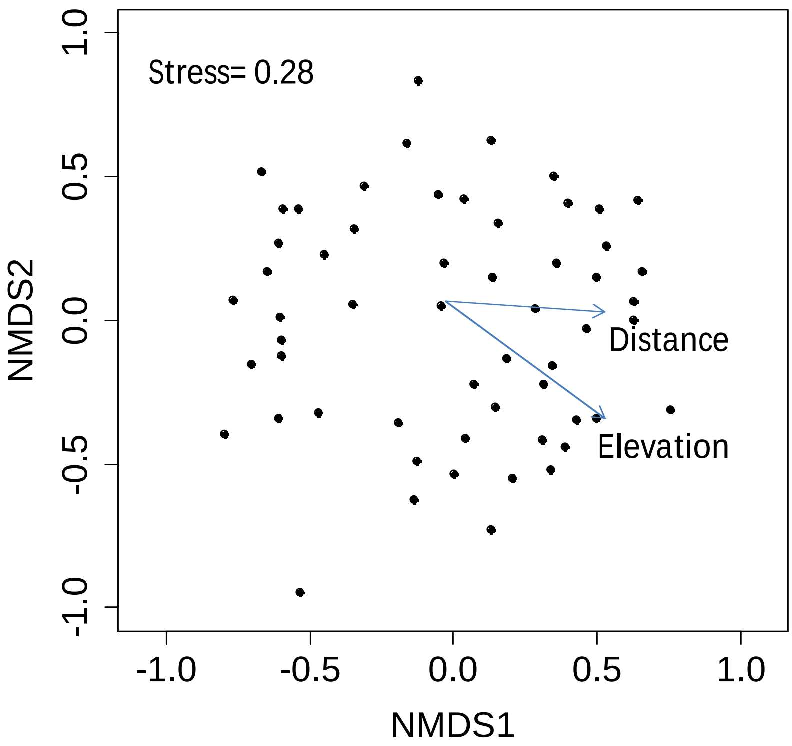

Figure 9. Non metric multi-dimensional scaling (NMDS) ordination performed on the species composition of 55 native tree and shrub species (based on the presence/absence data) in 60 patches of *Eucalyptus* plantations. Significant relationships with environmental variables are shown as vectors. Stress=0.28.

4.2. The roles of isolated retained trees for regeneration in *Eucalyptus* plantations

Generally the regeneration of the two characteristic forest tree species *P. falcatus* and *J. procera* in *Eucalyptus* plantations is affected by local, historical and local landscape connectivity.

The presence of *P. falcatus* in *Eucalyptus* plantations increased with connectivity to the retained trees to surrounding mature trees (GLM, Z= 3.450, p=0.001) and age of plantations (GLM, Z= 2.803, p=0.001) (Table 5a, Fig.10).

Table 5. Results from the final model of a General Linear Model with binomial error structure on the presence/absence: (A) *Podocarpus falcatus*, (B) *Juniperus procera* in *Eucalyptus* plantationsas a function of explanatory variables. (Distance was log-transformed).

A) *Podocarpus falcatus*				
Variables	Estimate	SE	Z	P
Intercept	-3.95721	0.97758	-4.048	*P* <0001
Local connectivity index	0.03862	0.01119	3.450	*P* = 0.001
Age of plantations (yr)	0.14518	0.05180	2.803	*P* = 0.001
B) *Juniperus procera*				
Intercept	-2.117	1.011	-2.094	*P* = 0.036
Size (ha)	2.073	1.171	1.770	*P* = 0.076

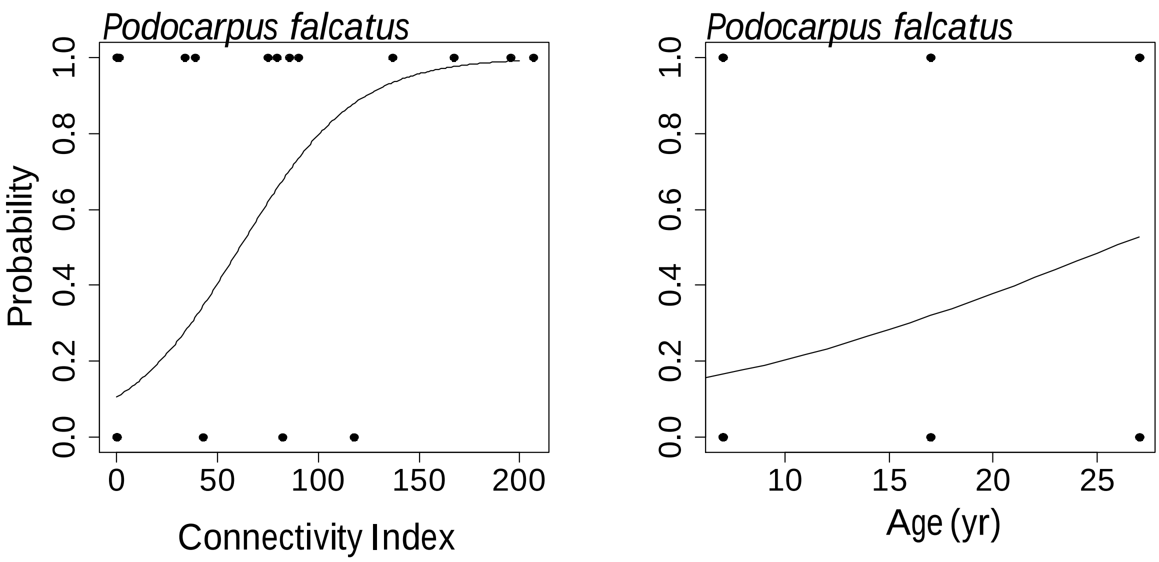

Figure 10. The presence of *P. falcatus* as a function of (A) local connectivity to *Eucalyptus* plantations (B) stand age of *Eucalyptus* plantation surrounding Chilimo forest, Central Ethiopia. Fitted lines added to guide the reader. For statistical analysis, see Table 5b

Colonization of *Eucalyptus* plantations by *J. procera* showed a slight increment with the size of the plantations (GLM, Z= 3.450, p=0.076) (Table 5b, Fig.11). However, local connectivity did not explain the colonization pattern**.**

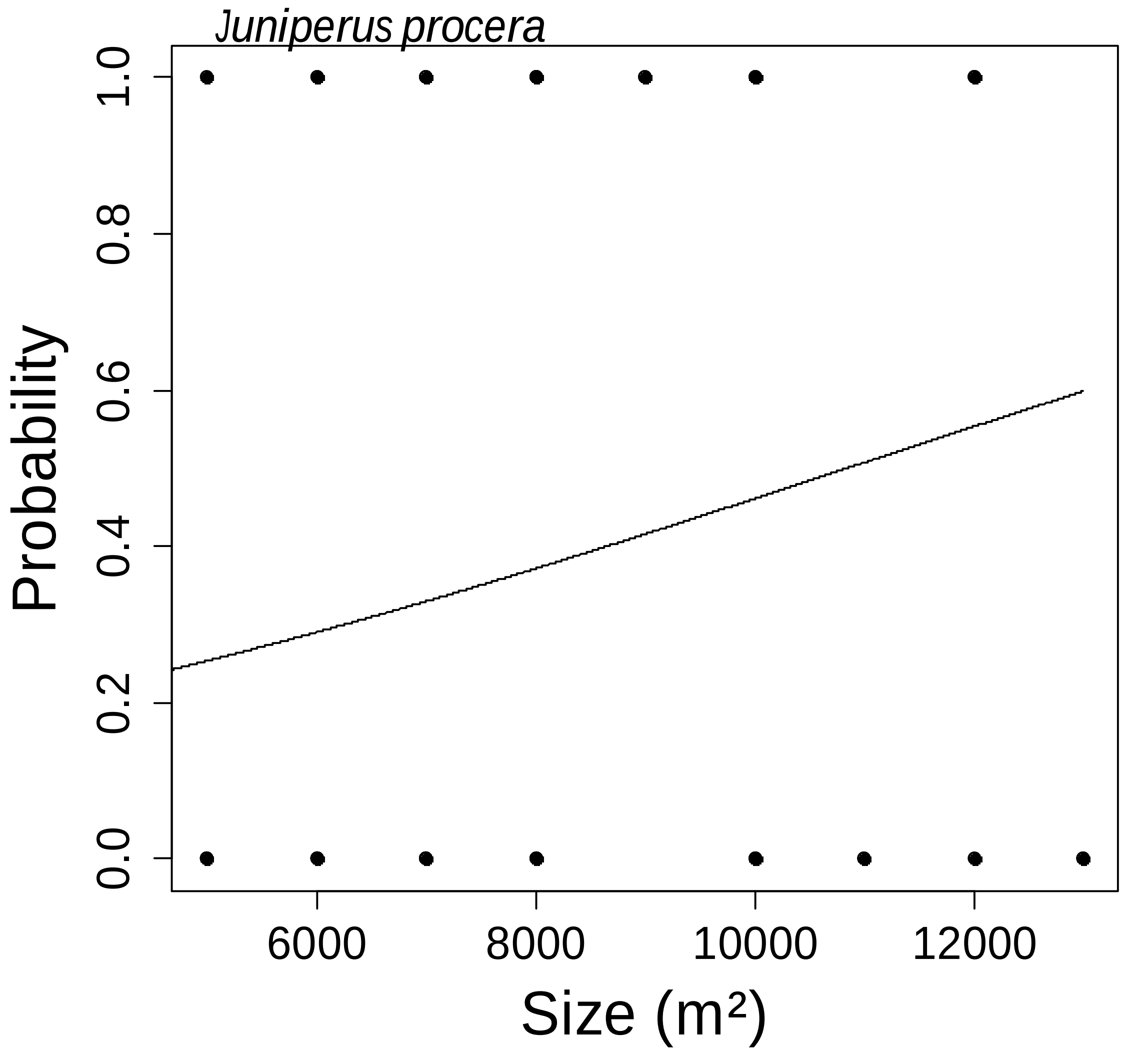

Figure 11. The presence of *J. procera* as a function of plantation size of *Eucalyptus* plantation surrounding Chilimo forest, Central Ethiopia. I added the fitted lines to guide the reader. For statistical analysis, see Table 6b.

4.3. Colonization of *Eucalyptus* plantations by herbs

4.3.1. Species richness

A total 92 herb species were recorded in the *Eucalyptus* plantations and 68 in the Chilimo natural forest (Appendix 4 and 5). The number of forest herb species (i.e. herb species usually prefer shade areas) in the patches decreased with canopy cover (GLM, Z= -2.786, p=0.005) and distance from the forest from around 12 species at the closest distances to less than 5 species at 10 kilometres out into the landscape (GLM, Z=-5.449, p<0.001). However, with increased height of the grass sward the number of forest herb species increased from less than 5 species in a patch with grass swards less than 10 cm up to more than 15 species in patches with grass swards higher than 40 cm (GLM, Z=4.065, p<0.001) (Table 6a, Fig. 12).

Table 6. Results from the final model of a General Linear Model with Poisson error structure on the number of: (A) forest herb species, (B) non-forest species in *Eucalyptus* plantation patches as a function of explanatory variables. (Distance was log-transformed).

A) The number of forest herb plant species				
Variables	Estimate	SE	Z	*P*
Intercept	2.70	10.62	0.25	< 0.001
Distance (km)	-0.08	0.02	-5.45	< 0.001
Canopy cover (%)	-0.02	0.01	-2.79	=0.005
Height of grass sward (cm)	0.02	0.00	3.39	< 0.001
B) The number of non- forest herb plant species				
Intercept	0.31	0.56	0.56	=0.5778
Age (yr)	0.03	0.01	4.78	< 0.001
Elevation (m asl)	0.00	0.00	2.16	=0.0308
Height of grass sward (cm)	0.01	0.00	2.41	=0.0158

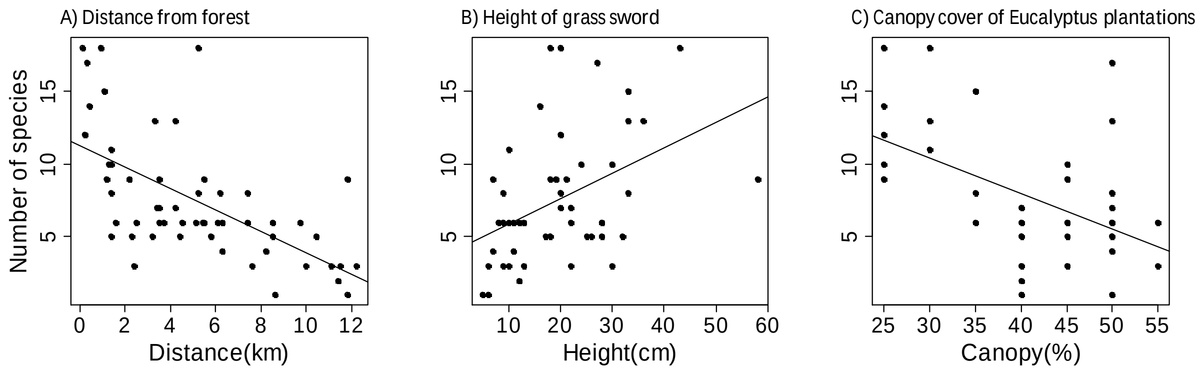

Figure 12.The number of forest herb species as a function of (A) Distance from forest (B) Height of grass sward, and (C) canopy cover of the *Eucalyptus* plantations.

The number of non- forest herb species (i.e. herb species mostly prefer open habitats) increased with age of plantations from 5 species at stand age of 7 years to 20 species at stand age of 25 years (GLM, Z = 4.782, p< 0.001), height of grass sward (GLM, Z = 2.413, p= 0.015) and elevation (GLM, Z = 2.159, P= 0.0158). However, the number of non forest herb species did not show significant difference with distance from the natural forest (GLM, Z =-0.071, p =0.943) (Table 6b, Fig. 13).

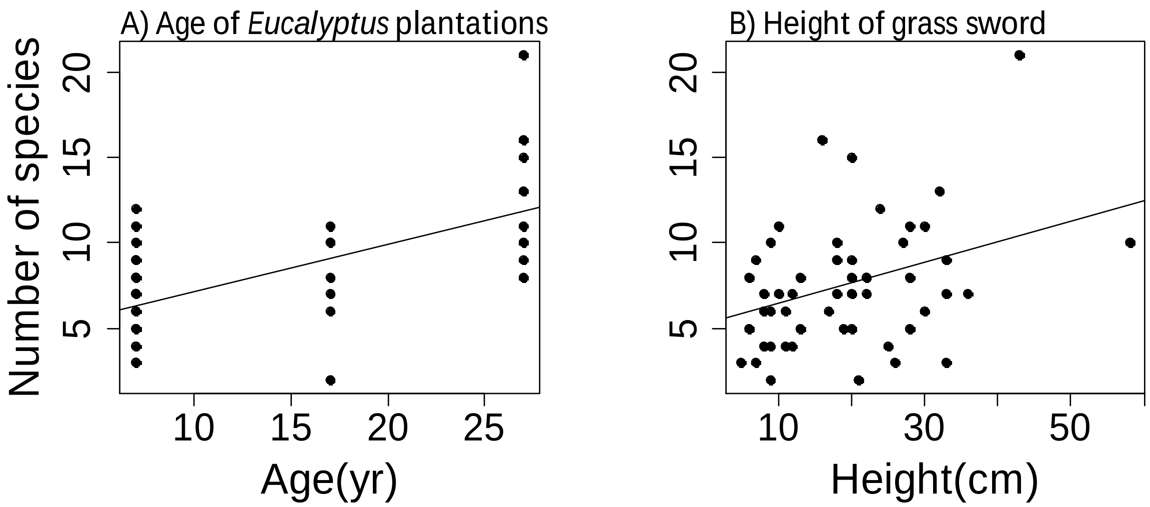

Figure 13.The number of non- forest herb species as a function of (A) Age of the *Eucalyptus* plantations (B) Height of grass sward.

Table 7. The herb species recorded in 56 *Eucalyptus* plantations occurring in the sites included in the indicator species analysis (i.e. ≥ 6 sites of the *Eucalyptus* plantations). Difference in indicator value denotes the difference between the two groups (F) "forest herbs" and (NF) "non-forest herbs". The *p*-values are from a Monte Carlo test. The table is sorted from the largest positive to the largest negative difference in indicator values. See Appendix 4 for a full species list).

Scientific name	Occurrences	Difference in Indicator values	*P*-values	Habit
Urtica simensis	7	0.65	0.02	NF
Echinops giganteus	32	0.53	0.11	NF
Anthemis tigreensis	20	0.35	0.27	NF
Rumex nepalensis	26	0.35	0.33	NF
Cirsium schimperi	13	0.22	0.57	NF
Tagetes minuta	13	0.22	0.58	NF
Torilis arevensis	12	0.22	0.54	NF
Bothriocline schimperi	11	0.2	0.6	NF
Salvia merjamie Forssk	11	0.18	0.64	NF
Bidens macroptera	11	0.18	0.6	NF
Arisaema schimperianum	9	0.15	1	NF
Conyza steudelii	7	0.13	1	NF
Hypoestes forskaolii	8	0.13	1	NF
Launaea intybacea	8	0.13	1	NF
Satureja paradoxa.	18	0.08	1	NF
Commelina africana	19	0.07	0.98	NF
Plantago palmate	21	0.03	1	NF
Conyza tigreensis	8	0.02	1	NF
Medicago polymorpha	28	0.02	1	NF
Rumex abyssinicus	20	0.02	1	NF
Lactuca inermis	8	0	1	NF
Cirsium englerianum	11	-0.02	1	F
Thalictrum rhynchocarpum	13	-0.03	1	F
Bidens prestinaria	26	-0.03	1	F
Argyrolobium ramosissimum	12	-0.05	1	F
Trifolium arvense	12	-0.05	1	F
Crepis rueppellii	9	-0.07	1	F
Agrocharis melanantha	9	-0.07	1	F
Anthriscus sylvestris	11	-0.07	1	F
Ageratum conyzoides	9	-0.08	1	F
Heracleum abyssinicum	9	-0.08	1	F
Cerastium octandrum	10	-0.08	1	F
Alchemilla pedata	7	-0.12	0.47	F

Plectranthus punctatus	6	-0.15	0.64	F
Achyranthes aspera	18	-0.17	0.59	F
Cyanotis barbata	26	-0.18	0.3	F
Pimpinella hirtella	13	-0.2	0.24	F
Senecio tamoides	8	-0.2	0.24	F
Cynoglossum amplifolium	11	-0.22	0.34	F
Oxalis corniculata	23	-0.23	0.13	F
Alchemilla abyssinica	14	-0.23	0.32	F
Hypoestes triflora	33	-0.25	0.06	F
Plantago lanceolata	9	-0.27	0.28	F
Stellaria sennii	14	-0.27	0.26	F
Laggera crispata	8	-0.37	0.12	F
Kalanchoe densiflora	14	-0.4	0.17	F
Trifolium rueppellianum	37	-0.5	0.01	F
Geranium arabicum	12	-0.53	0.04	F
Galium spurium	7	-0.87	0	F
Cynoglossum geometricum	16	-0.95	0	F

4.3.2. Species composition

Species composition of herb plants was significantly related to local factors such as canopy cover (r^2= 0.11, *p*=0.03) and distance (r^2 = 0.11, *p*=0.03) (Fig. 14).

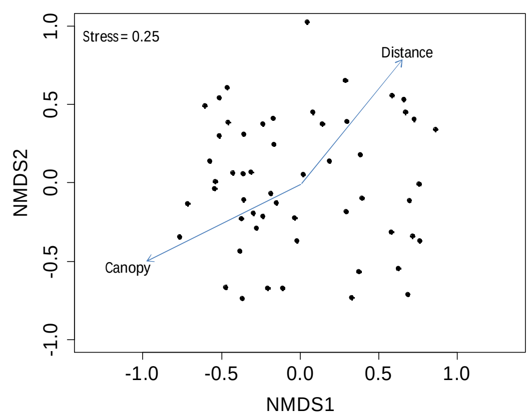

Figure 14. Non metric multi-dimensional scaling (NMDS) ordination performed on the species composition of 92 herb species (based on the presence/absence data) in 56 patches of *Eucalyptus*. Significant relationships with explanatory variables are shown as vectors.Stress=0.25.

5. Discussion

There could be many both local and historical landscape level constraints for a plantation to become an efficient site for regeneration of native plant biodiversity (Ostertag *et al*., 2007; Brudvig, 2011). In this study, one of the major variables regulating the density and species richness of native woody plants in small *Eucalyptus* plantations in Ethiopia was the grazing pressure. Furthermore, another observed general pattern is that higher species richness was recorded at distances of a few hundred meters from a remnant natural forest compared to plantations located at kilometres away from the forest. This clearly suggests that, in addition to local factors, the availability of diaspores is limiting the regeneration of native species under *Eucalyptus* plantation in the study area. With regard to isolated retained trees, these isolated individuals of *P. falcatus* and *J. procera* are important sources of diaspores colonizing the scattered *Eucalyptus* plantations across the farmscape surrounding the remnant Chilimo forest. In the same way, herbs species richness species was also affected by grazing pressure. Noteworthy is the general pattern where by higher species richness of forest herbs only at a distance of a few hundred meters from a remnant natural forest compared to plantations located at kilometers away from the forest was recorded in this study. The non-forest herbs have not been influenced by distance from the natural forest.

5.1. Local factors

Although, some studies suggested the role of livestock grazing as a facilitative process (McEvoy *et al*., 2006) this study clearly indicated that livestock induced disturbances might be among the major factors constraining the regeneration of woody and herb species in *Eucalyptus* plantations. Studies in forest patches with native trees in Northern Ethiopia have similarly found strong negative effects by goats and cows on recruitment of trees (Bongers *et al*., 2006; Alemayehu Wassie *et al*., 2009). Likewise, study on the herb layer in overgrazed Oak forest in Greece showed that overgrazing reduced species richness of typical forest herbs (Chaideftou *et al*., 2011). A similar study on herbaceous vegetation potential in mountain grazing land in Eastern Tigray reported very few herb species in open grazing land when compared to ten and five years area enclosure (Gebrewahd Amaha, 2014). This shows that the results in this study were not due to the sites being *Eucalyptus* plantations. The negative effect of grazing by domestic animals on regeneration was also confirmed by observations of many native woody and herb species inside a few places that were fenced. Moreover, another study on transplant experiment of forest herbs in plantations with conservation status in Canada suggested that understorey herbaceous plantations establishment could be successful if there is protection from large herbivors (Boothroyd-Roberts *et al*., 2013). Thus, protection from herbivore is critical, as a large number of livestock grazing can seriously impair restoration efforts (Abraham Abiyu *et al*., 2011; Boothroyd-Roberts *et al*., 2013; Waswa *et al*., 2013). Again, the construction of fences to keep grazing livestock away from seedlings and saplings will be necessary if the plantations should be used as restoration sites or a foster ecosystem. On the other hand it is understood the construction of fences around *Eucalyptus* plantations may not be easy to convince the farmers as fencing around crop fields.

However, sometimes there is still poor regeneration even after excluding grazing even in forests with seed sources indicating that many factors may interact for a successful regeneration to take place (Ermias Aynekulu *et al*., 2009, Alemayehu Wassie *et al*., 2009). Excluding livestock is one of the main strategies in many types of restorations (Abraham Abiyu *et al*., 2011; Waswa *et al*., 2013). For example, in Northern Ethiopia it has become a widespread practice to use exclosures to restore native vegetation (Abraham Abiyu *et al*., 2011). However, an important question is"is excluding livestock from tree plantations may be a more efficient method than exclosures of open areas, since many native tree species have poor growth in sunny conditions?" (Alemayehu Wassie *et al*., 2009, Demel Teketay, 2011). It is also reported (Green and Boone, 1995) that livestock grazing sometimes favour establishment of pioneer species. Sequential restoration with more shade tolerant species coming in at a later stage is one option when restoring open areas and likewise it could be valuable to investigate if species that has an efficient regeneration under *Eucalyptus* could be facilitating other species establishiment.

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The current study has demonstrated a clear evidence of a positive effect of light penetration into the *Eucalyptus* plantations in enhancing plant colonization, since the total abundance of trees and shrubs increased with decreased canopy cover. This is inline with results of, several previous studies reported (Feyera Senbeta and Demel Teketay, 2001; Mulugeta Lemenih *et al*., 2004). However, the density of *Podocarpus falcatus* was instead positively correlated to canopy cover. This is in fact in line with several other studies showing that *P. falcatus* is a typical shade tolerant species regenerating better in shaded than open conditions (e.g. Demel Teketay, 2011). Two other climax species in dry Afromontane forests, *Juniperus procera* and *Olea europaea* subsp. *cuspidata*, have been found to regenerate better in somewhat more light or less dense canopy conditions (Alemayehu Wassie *et al*.,2009). However, both of these species also regenerate in closed canopy forests indicating that they are to some extent shade tolerant (Demel Teketay, 1997; Alemayehu Wassie *et al*., 2009). However, the contrasting results between how light affects the general species richness and the density of *P. falcatus* highlight the importance of species specific studies and indicate that perhaps light is an important factor regulating which species that are favoured or disfavoured in plantations. However, even if these results suggest that light would be an important factor regulating species composition in plantations, canopy cover did not correlate significantly with the composition in our plantations (p=0.61), perhaps because other gradients dominated (e.g. distance and elevation) (see Fig.9).

Light is an important factor for species richness of herb species in plantations (Mulugeta Lemenih *et al*., 2004; Ostertag *et al*., 2007). In some earlier studies it was reported that canopy cover inhibit open habitat herb species (Parrotta *et al*., 1997). The current study has shown a less expected finding, i.e. the strong negative pattern between canopy cover and species richness of forest species.

In contrary to the result reported in (Habte Telila *et al*., 2015), light did not show significant difference in the species specific studies of the regeneration of *P. falcatus* and *J. procera*. This could indicate that canopy cover may not be very important as an explanatory variable within the range observed at the study site, or, light conditions may be of some importance that canopy cover is unsuitable estimation of light flux. Further experimental studies including light measurements, are needed to elucidate the role of canopy cover on the two characteristic forest species colonization in *Eucalyptus* plantations.

5.2. Landscape factors

The results demonstrated a decrease in species richness of native woody species with distance from the major remnant forest in the area (Fig.6; Table 3), a pattern also confirmed by the indicator species analysis with many species associated with sites close to the forest. This shows that many seeds come from the Chilimo forest even if other sources such as small remnant patches and isolated trees might be important. The results of the study is in line with several previous studies showing that the proximity to seed sources is crucial for a successful colonization in plantations (Holl *et al.*, 2000; Jörg-Goddert- *et al.*, 2000; Cramer *et al.*, 2002; Zamora *et al.*, 2010). The current study has shown a steady decline of distance over approximately 10 kilometres. That some native trees could be dispersed such long distances is also shown by de Mendonça-Lima *et al.*, (2014), who showed much regeneration of animal dispersed species in plantations around 8 kilometres from native forests in Brazil. Similar to our study, a clear decline in regeneration of trees in plantations over a few kilometres from the source was also found in Uganda (Zanne and Chapman, 2001). However, the decline can be quite abrupt at short distance from the source, with a more gradual decline at further distances, as shown in a study in Puerto Rico for distances only up to 640 m (Parrotta, 1993).

Even if there was a decline in species richness of native woody species with distance from the natural forest, there were woody species that did not show decrease with distance (Table 2, Fig. 7 and 8). One possible explanation is that those species occur across the whole landscape or in pockets here and there, including in the soil seed bank, and thus that the seed source for these species may not be solely from the remnant forest. For example, even if *Acacia abyssinica* was more common in the plantations close to the forest, it is known to occur across the highlands of the study area even outside the remnant forest patches. However, as no inventory of the whole landscape was done information on mechanism is lacking. Another explanation for a lack of effect of distance to the remnant forest could be that some species are dispersing effectively over even longer distances than currently studied. Species mostly regenerating under the canopy of their own mother are sometimes denoted fine-grained species (Lawes *et al*., 2007). To what extent the species recorded from Chilimo forest but not in our plantations (Table 2) are poorer dispersers than the ones that regenerated in the plantations is not unknown, but there is no clear difference in the distribution of dispersal syndromes (Table 2). The chance that some of the woody species in the plantations are resprouting from individuals present in the pre-plantation period is small given that most *Eucalyptus* plantations are established at open areas with little competition from other plants (personal observation).

The results from the study of herbs show a decrease in species richness of forest herb species with distance from the major remnant forest in the area (Fig.12; Table 6). This shows that many seeds come from the Chilimo forest even if other sources such as small remnant patches also might be important. In this study, most of the forest herb species have the adhesive fruits are transported until they fall off naturally or groomed off (Agnew and Flux, 1970), consequently, they have the potential to travel over longer distances (Brunet, 2007; Matlack, 2005). This was not seen in the current study, this might be due to lack of wild vertebrates in the study landscape that possibly take propagules longer distance (personal observation). Cattle grazers might not be efficient dispersal agent of forest herb species through adhesion because of local range limitation of these animals due to habitat alteration (Chaideftou *et al*., 2009). Consistent with earlier studies in Japan a clear decline in colonization of plantation by forest herbs around 17.4 km from native forests was also found (Takahashi and Kamitani, 2004), some typical forest species in broadleaved plantations on former arable land showed lowest number to about 2 km from ancient woodland edge (Brunet, 2007).

Though, the forest herb species number decrease with distance from the natural forest the number of non-forest species did not show significant pattern in this study. The possible reason is that the diaspore source of those open habitat herb species might be farm lands, grazing land or seed bank in the farmscape. A study in crop fields in the Eastern Ethiopia reported about 102 weed species (Tamado Tana and Milberg, 2000) an indication for the presence of plenty of non forest herbs in the farm lands. Further experimental studies may be important on the dispersal of herbs. The other probable reason is that most of these species are common in farm and grazing lands in which the *Eucalyptus* plantations are nearer. In contrarst, cattle might be efficient dispersal agents for non-forest herbs since they move regularly between farm and grazing lands and *Eucalyptus* plantations (Personal observation).

Birds have been reported to generally be better dispersers than mammals (Carlo *et al*., 2013) even if no evidence for this was found in our data. Among several species that did not reach far out in the landscape were mammal dispersed even many bird dispersed (or bird and mammal dispersed) species. These species might be dispersed by forest dependent birds of which some are fruit eating (cf. Gove *et al*., 2013). Main dispersal agents besides birds in our study landscape are probably primates; in our case olive baboon, vervet monkey and Colobus monkey. At least for olive baboon it has been shown in other Ethiopian agricultural landscapes that it has difficulties in crossing the agricultural matrix and that its occurrence decline sharply at distances of 1-2 km from the forest edge (Debissa Lemessa *et al*., 2013; Samnegård *et al*., 2014). There is a need of explicit studies of which animals are the main dispersal vectors, since less obvious nocturnal animals could also play a role. An interesting pattern appearing as a consequence of animal dispersal is that the regeneration pattern could display a strong clustering at the base of canopy trees that have been chosen as roosting and perching trees (Fig. 15).

Figure 15.Clustered saplings of (A) *Myrsine melanophloeos* and (B) *Podocarpus falcatus* tree species in the two *Eucalyptus* patches of plantations at the study site. © Habte Telila.

5.3. Local connectivity

In this study the probability of colonization of *P. falcatus* in *Eucalyptus* plantation *was* high with connectivity to the remnant trees. This shows that remnant trees in an agricultural landscape may act as the source of seed. The absence of significant pattern of connectivity between remnant trees of *J. procera* and *Eucalyptus* plantations may be explained partly by the small number of studied sites that did not provide enough variation in the data. It could also be argued that this result reflects the absence of a real pattern, but it is believed the presence of retained trees of *J. procera* surrounding the plantations may be an indication that they could be the source for colonizing the *Eucalyptus* plantations (Fig. 16). Nonetheless, the problem with the local connectivity measure is that it was not easy to asertain if the sampled trees of the two species were female or male.

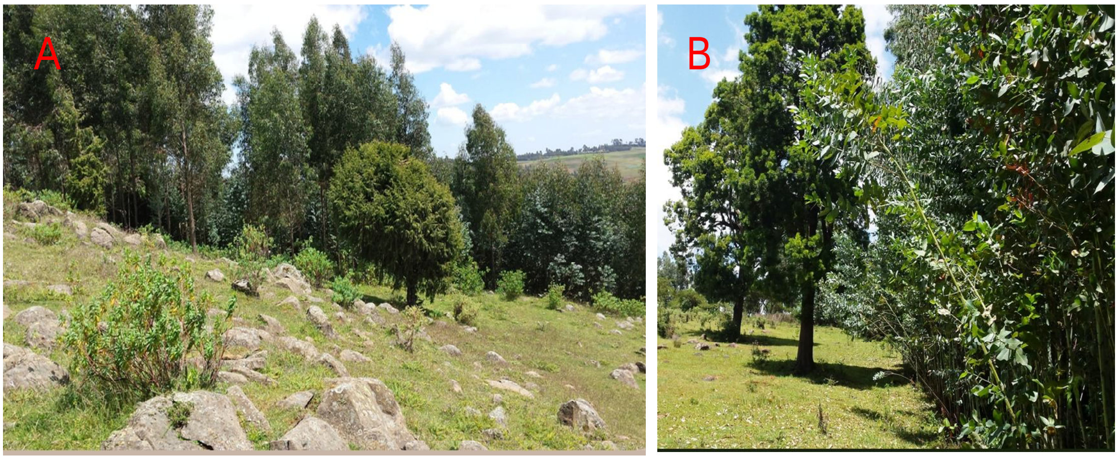

Figure 16. Retained trees of (A) *J. procera*, and (B) *P. falcatus* proximate to *Eucalyptus* plantations of the study area. © Habte Telila

5.4. Plantation size

The study shows that the size of the *Eucalyptus* plantations did not show a significant pattern with number of species and density neither in native woody species nor herb species. The smaller influence of plantation size could be brought about by my selection of more or less homogeneous stand size (see the method part). However, in the species specific study of the two characteristic forest species, the plantation size was the best predictor of presence of *J. procera i.e.* it showed slight increment with the size of the plantations**.** This might be because of high competition for the use of a resource in small sizes that hamper establishment and recruitment. For example, in a large size patches more light reaches the plants that in turn could increase the chance of presence in greater numbers of plant species (Chen *et al*., 2014). However, some studies, for example Honnay *et al*., (2002) revealed that large size of the patch *per se* may not be necessarily the reason for the presence of a given plant species (Honnay *et al*., 2002).

5.5. Plantation age

The conclusion that species richness increases with stand age of plantations (e.g.Brunet, 2007) is confirmed in our study of herbs, particularly in non-forest herbs and remnant tree of *P. falcatus* surrounding the *Eucalyptus* plantations. This might be due to the continuous management of *Eucalyptus* through coppicing that increase the canopy openness and may favour the growth of herb species. The other reason may be litter depths that are very low under *Eucalyptus* because people usually collect the fallen leaves and branches of *Eucalyptus* for fire wood and this may lead to low regeneration of herb species. Earlier studies also show that pioneer species tend to dominate younger plantations of *Eucalyptus* (Feyera Senbeta *et al*., 2002).

Similarly, the probability of colonization by P. *falcatus* trees of *Eucalyptus* plantations increased with age of the plantations (Fig.10; Table 5). This might be because the old stands of *Eucalyptus* plantations that are are opened after cutting could favour the mature trees of *P*. falcatus, since the species is moderately shade tolerant when young and moderately intolerant when older. (Legesse Negash, 2010). Consistent to this finding an earlier study (see Feyera Senbeta *et al*., 2002) also reported *P. falcatus* as one of the most commonly found in the *Eucalyptus* stands of age of 27 years.

The stand age did not influence the number of forest herb species and the colonization by *J. Procera* of *Eucalyptus* plantations. This might be because of impacts from other variables such as size. However, older plantations in Europe support populations of some typical forest species (Endels *et al*., 2004), but young plantations less than 10 years are often devoid of most herb species typical of natural forest (Boothroyd_Roberts, *et al*., 2013).

5.6. Plantations as a restoration tool in an Ethiopian socio-ecological context

This study highlights the finding that woody and herb species could colonize and thrive in plantations, especially if grazing could be controlled and if the problem with seed limitation could be handled. It was found that not only the natural forest, but also the remnant trees in farmscape, play an important role s a source of seed for the regeneration of trees, if the retained trees in an agricultural matrix are protected.

Farmers prefer planting *Eucalyptus* because of its fast growing, coppicing ability and incentives from selling its timber. Moreover, farmers are not prohibited by law to cut *Eucalyptus* (Tola Gemechu *et al*., 2014). A study in Northern Ethiopia (Michael and Holm, 2006) also reported that the factors which influence farmers' to grow woody plants in their home garden is determined by means of how woody plants contribute to home consumption and immediate cash regeneration whereas farmers are prohibited by law from cutting and using a single tree of *P. falcatus* and *J. Procera* this is not the case with *Eucalyptus*. The law indirectly discourage farmers from letting native trees regenerate in *Eucalyptus* plantations need to be changed to help foster growth of native tree species in *Eucalyptus* plantations. In principle restoring native biodiversity in *Eucalyptus* plantations seem to hold a great deal of promise. Moreover, *Eucalyptus* plantations are common and the cover is increasing in Ethiopia (Million Bekele, 2011). However, if farmers do not get any incentives of letting native trees and shrubs regenerate it is not likely that *Eucalyptus* plantations will play an important role in restoration of native biodiversity across Ethiopian landscapes. In principle there might be a high economic value of trees such as *Podocarpus* and *Juniperus*, but Ethiopian regulations intended to protect native biodiversity prohibits cutting and selling these species (FDRE, 2007). Thus, farmers might not be confident that they could earn money on these species if they let them regenerate on their land. As a consequence, the rules aiming at protecting biodiversity hinder restoration efforts (Mulugeta Lemenih and Habtemariam Kassa, 2014).

It is suggest that laws and policies be revised and the possibility of introducing different types of certification and control mechanisms evaluated to enable farmers sell timber of native trees from plantations, while at the same time minimizing the risk of over-exploitation of native forests. The Ethiopian government has launched a “re-greening” campaign aimed at increasing the availability of woody products as well as restoring the country’s forest associated biodiversity (FDRE, 2011). The Ethiopian government is also giving licences to small woodwork enterprises to create job opportunities. Thus, in the Ethiopian context there is a great need of finding positive synergies between goals for conservation and economic development that also are supported by the farming community (cf. Mulugeta Lemenih and Habtemariam Kassa, 2014). The future role of *Eucalyptus* plantations for fostering native biodiversity will thus not only depend on grazing pressure and diaspore availability but also on external factors such as land tenure, markets and regulations.

6. Conclusions and recommendations

The study discovered that if small *Eucalyptus* plantations should be used as a restoration sites for native vegetation it is important to control livestock grazing. Despite the improvement of habitat quality (e. g. regulating grazing) is crucial for recovery of plant biodiversity under exotic plantations landscape factors such as connectivity could also play prominent role. Since conservation measures of plant biodiversity close to continuous natural forest could solve the problem of dispersal limitation by using as source propagules (Habte Telila *et al.*, 2015). In landscapes with small amounts of existing adjacent forests, and a poor soil seed bank (which is the case in the dry Afromontane forest type we have studied) seeds from native trees and shrubs need to be applied if *Eucalyptus* plantations should also be used to restore biodiversity. Moreover, restoration studies in exotic plantations focused on mostly tree regeneration as a target of forest restoration efforts (Boher *et al*. 2012) should also include herbaceous plant species since herbaceous layer vegetation may provide favorable habitat for regeneration of woody species (Hutchinson *et al*., 1999; Bohre *et al*., 2012). However there is a critical need to under take experimental studies on the dispersal of herbs. Beside protection of destruction of remnant natural forest, retained trees can be used as important sources for future restoration activities if the quality of agricultural matrix will be improved. Futhermore since light is an important factor regulating which species that are favoured or disfavoured in plantations further experimental studies including light measurements, are needed to elucidate the role of canopy cover on the two characteristic forest species colonization in *Eucalyptus* plantations.

Eucalyptus plantations embedded in farmscape mostly owned by farmers might contribute to the conservation of biodiversity by providing an environment for the cultivation of forest herb species as an alternative to their destructive harvest from the natural population (Boothroyd-Roberts *et al*., 2013).Thus, the knowledge from this study might initiate an approach of restoration from which people gets benefits (Bullock *et al*., 2011). Mainly in countries like Ethiopia where food, firewood and other key resources to sustain human populations are limiting. Finally, the existing law that prohibits the use of native trees for timber extraction needs to be amended to allow farmers to nurse seedlings of these trees in their *Eucalyptus* plantations.

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8. Appendices

8.1. Appendix 1

Native tree and shrub species in *Eucalyptus* plantations with their total abundances as well as the abundances in respective size class (Ad= Adult, Juv= Juvenile, Sap= Sapling). For each species is also given the dispersal agents (DA) (where, M= Mammals, B= Birds, BM= Birds and Mammals, W=Wind and WA= Water) and source of literature for this information. Note that Acacia spp. could also be dispersed by autochory over shorter distances.

Species	Ad	Juv	Sap	Total	DA	Source
Acacia abyssinica Hochst. ex Benth.	49	11	7	67	M	Feyera Senbeta *et al.*, 2002
Acacia seyal Del.	2	0	0	2	B	Demel Teketay, 2005
Acanthus sennii Chiov.	4	0	0	4	M	Feyera Senbeta *et al.*, 2002
Allophylus abyssinicus (Hochst.) Radlk.	3	2	1	6	M	Feyera Senbeta *et al.*, 2002; Demel Teketay, 2005
Apodytes dimidiata E. Mey. ex Arn.	11	6	1	18	BM	Feyera Senbeta *et al.*, .2002
Asparagus africanus Lam.	9	3	0	12	B	Demel Teketay, 2005
Bersama abyssinica Fresen.	7	4	0	11	BM	Feyera Senbeta *et al.*, 2002
Borassus aethiopum Mart.	4	0	0	4	M	Feyera Senbeta *et al.*, 2002
Buddleja polystachya Fresen.	22	1	7	30	B	Demel Teketay, 2005
Calpurnia aurea (Ait.) Benth.	9	0	0	9	BM	Feyera Senbeta *et al.*, 2002; Demel Teketay, 2005
Canthium oligocarpum Hiern.	5	3	2	10	B	Demel Teketay, 2005
Capparis tomentosa L.	5	0	0	5	BM	Demel Teketay, 2005

Carissa spinarum L.	50	10	5	65	BM	Feyera Senbeta 2002
Clausena anisata (Willd.) Benth.	2	3	5	10	M	Feyera Senbeta and Demel Teketay, 2001
Croton macrostachyus Del.	15	20	16	52	BM	Feyera Senbeta *et al.*,2002
Discopodium penninervum Hochst.	14	2	0	14	B	Feyera Senbeta *et al.*,2002
Dombeya torrida (G.F.Gmel.) P. Bamps	11	3	1	15	M	Demel Teketay, 2005
Dovyalis abyssinica (A.Rich.) Warp.	16	6	6	28	BM	Feyera Senbeta *et al.*,2002; Teketay 2005
Ekebergia capensis Sparrm.	5	5	2	12	BM	Feyera Senbeta *et al.*,2002; Demel Teketay, 2005
Erythrina brucei Schweinf.	1	0	0	1	B	Eshetu Yirdaw 2002
Erythrococca trichogyne (Muell.Arg.) Prain	4	3	0	7	B	Demel Teketay, 2005
Euphorbia tirucalli L.	7	3	1	11	B	Demel Teketay, 2005
Ficus sur Forssk.	3	0	0	3	BM	Demel Teketay, 2005
Galiniera saxifraga (Hochst.) Bridson	5	6	2	13	BM	Feyera Senbeta *et al.*, 2002
Hagenia abyssinica (Bruce) G.F. Gmel.	16	7	0	23	B	Eshetu Yirdaw, 2002
Hypericum quartinianum A.Rich.	7	6	5	18	W	Demel Teketay, 2005
Ilex mitis (L.) Radlk.	0	8	10	18	B	Feyera Senbeta *et al.*, 2002
Juniperus procera L.	46	21	59	126	BM	Eshetu Yirdaw 2002; Demel Teketay, 2005
Launaea taraxacifolia (Wild.)Amin ex C.Jeffrey	11	3	2	16	B	Demel Teketay, 2005

Maesa lanceolata Forssk.	7	4	4	15	M	Feyera Senbeta *et al.*, 2002
Maytenus arbutifolia (A. Rich.) Wilczek	38	28	16	82	B	Feyera Senbeta *et al.*, 2002
Maytenus gracilipes (Welw. Ex Oliv.) Exell	17	3	0	20	B	Feyera Senbeta *et.al.*, 2002
Maytenus undata (Thnub.)Blakelok	21	7	6	34	B	Feyera Senbeta *et.al.*, 2002
Myrica salicifolia Hochst. ex A.Rich.	12	1	2	15	BM	Feyera Senbeta et al., 2002
Myrsine africana L.	5	2	4	11	BM	Demel Teketay, 2005
Myrsine melanophloeos (L.) R.Br.	30	5	201	236	B	Demel Teketay, 2005
Nuxia congesta R.Br. ex Fresen.	2	2	0	4	BW	Demel Teketay, 2005
Olea capensis L. subsp. macrocarpa (C.A.Wright) Verdc.	7	4	3	16	BM	Feyera Senbeta *et al.*, 2002
Olea europea L. subsp. cuspidata (Wall. ex G.Don) Cif.	32	10	133	175	BM	Feyera Senbeta *et al.*,2002 ; Demel Demel Teketay,2005
Olinia rochetiana A.Juss.	1	0	0	1	B	Feyera Senbeta *et.al.*, 2002
Osyris quadripartita Decne	7	1	1	9	BM	Feyera Senbeta *et.al.*, 2002
Podocarpus falcatus (Thunb.) Mirb.	25	15	70	110	BM	Feyera Senbeta *et.al.*, 2002; Eshetu Yirdaw, 2002; Demel Teketay, 2005
Premna schimperi Engl.	4	5	112	21	B	Feyera Senbeta *et.al.*,2002
Prunus africana (Hook.f.) Kalkm.	15	4	1	20	BM	Feyera Senbeta *et.al.*,2002; Eshetu Yirdaw, 2002

Rhamnus staddo A.Rich.	2	4	1	7	BM	Feyera Senbeta *et.al.*, 2002
Rhus glutinosa A.Rich. subsp. neoglutinosa(M.Gilbert) M.Gilbert	8	3	6	17	BM	Feyera Senbeta *et.al.*, 2002
Rosa abyssinica Lindley	22	5	1	28	M	Demel Teketay, 2005
Rubus steudneri Schweinf.	14	0	1	15	BM	Demel Teketay, 2005
Rumex nervosus Vahl	16	2	0	18	B	Feyera Senbeta *et.al.*, 2002
Salix mucronataThunb.	1	0	0	1	WA	Feyera Senbeta *et.al.*, 2002
Scolopia theifolia Gilg	1	2	0	3	B	Feyera Senbeta *et.al.*, 2002
Sideroxylon oxyacanthum Baill.	4	2	4	10	BM	Feyera Senbeta *et.al.*, 2002
Solanecio gigas (Vatke) C.Jeffrey	54	15	1	1	B	Demel Teketay, 2005
Spiniluma oxycantha Baill.	7	0	3	10	BM	Feyera Senbeta *et al.*, 2002
Vernonia auriculifera A. Rich.	4	4	5	13	W	Feyera Senbeta *et.al.*, 2002

*Beside the sources cited in reference part Flora of Ethiopia and Eritrea (The Ethiopian Flora Project) was used as a source for identifying dispersal agents

8.2. Appendix 2

Lists of native tree and shrub species found in Chilimo forest but absent in the investigated *Eucalyptus* plantations. * Unspecialized dispersal agent.

Species	Dispersal agents	Source
Acacia lahai Steud. and Hochst. ex Benth.	Uns*	Or and Ward, 2003
Cassipourea malossana (Baker) Alston	M	Feyera Senbeta *et al.,* 2002
Embelia schimperi Vatke	BM	Sun, *et al*.,1997; Gross-Camp, 2005
Erica arborea L.	W	Demel Teketay, 2005
Gnidia glauca (Fresen.) Gilg.	Uns*	Puyravaud *et al*., 2003
Hypericum revolutum Vahl	B	Feyera Senbeta and Demel Teketay, 2001
Maytenus addat (Loes.) Sebsebe	B,M	Feyera Senbeta and Teketay, 2001
Olea welwitschii (Knobl.) Gilg and Schellenb.	BM	Demel Teketay, 2005
Pittosporum viridiflorum Sims	BM	Feyera Senbeta and Teketay, 2001
Rhus vulgaris Meikle	BM	Demel Teketay, 2005
Schefflera abyssinica (Hochst. ex A.Rich.) Harms	B	Demel Teketay,2005
Schefflera volkensii (Engl.) Harms	B	Demel Teketay, 2005
Schrebera alata (Hochst.) Welw.	Uns*	Voigt *et al*., 2004
Solanecio mannii (Hook.f.) C. Jeffrey	BM	Demel Teketay, 2005
Teclea nobilis Del.	BM	Feyera Senbeta *et al*.,2002; Demel Teketay, 2005
Vernonia amygdalina Del..	W	Feyera Senbeta *et al*., 2002

8.3. Appendix 3

The species-by-site matrix sorted by distance showed neither a pattern of nestedness nor turnover of the species.

Species	P 33	P 34	P 48	P 25	P 31	P 3	P 17	P 64	P 14	P 15	P 16	P 18	P 30	P 32	P 20	P 22	P 6	P 21	P 24	P 23	P 13	P 11	P 19	P 26	P 35	P 29	P 65	P 55	P 50	P 8	P 42	P 43	P 7	P 44	P 9	P 10	P 51	P 46	P 45	P 52	P 12	P 47	P 49	P 36	P 37	P 39	P 27	P 53	P 40	P 54	P 38	P 66	P 60	P 61	P 56	P 57	P 59	P 58	P 62	P 63	Frequency
Acacia seyal	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Acanthus sennii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Erythrina brucei	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Ficus sur	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Olinia rochetiana	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Salix subserrata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Capparis tomentosa	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
Scolopia theifolia	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	2
Asparagus africanus	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	3
Borassus aethiopum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Canthium oligocarpum	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Erythrococca trichogyne	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Ilex mitis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Maytenus undata	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Olea capensis subsp.macroc	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Allophylus abyssinicus	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
Ekebergia capensis	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	4
Euphorbia tirucalli	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	4
Nuxia congesta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
Oxycanthum stenocarpus	1	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
Rhamnus staddo	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	4
Rubus steudneri	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
Sideroxylon oxycanthum	0	0	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
Bersama abyssinica	0	0	0	1	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5
Calpurnia aurea	0	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5
Discopodium penninervum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	0	1	1	6
Galiniera saxifraga	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6
Launea taraxacifolia	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6
Myrsine africana	0	0	0	1	0	0	1	0	1	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	6
Osyris quadripartita	1	0	0	0	0	0	1	1	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6
Vernonia auriculifera	0	0	0	0	1	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6
Clausena anisatna	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	7
Maytenus gracilipes	0	1	0	0	0	0	1	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	7
Rhus glutinosa	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	1	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7
Rumex nervosus	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	1	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	7
Hypericum quartinianum	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	8
Maesa lanceolata	0	1	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	8
Premna schimperi	1	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8
Dombeya torrida	0	1	0	0	0	0	1	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	9
Apodytes dimidiata	0	0	0	1	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	1	10
Carissa spinarum	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	1	0	0	0	0	1	1	0	0	1	0	0	0	0	0	0	1	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10
Myrica salicifolia	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	1	0	0	1	0	0	0	0	1	0	0	0	0	0	1	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	10
Rosa abyssinica	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	1	1	1	0	1	0	0	0	0	0	0	1	1	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10
Prunus africana	1	1	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	11
Hagenia abyssinica	0	0	0	0	0	0	1	0	1	1	0	0	0	0	0	0	0	1	0	1	0	0	1	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	1	0	1	0	0	0	12
Maytenus arbutifolia	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0	1	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	1	0	1	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	12
Buddelja polystachya	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	1	1	1	0	0	1	0	1	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	1	0	0	0	0	0	0	1	13
Croton macrostachyus	0	0	0	1	1	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	1	0	0	0	0	1	1	0	0	1	0	0	0	0	0	0	1	1	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	13
Dovyalis abyssinica	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	1	0	0	0	0	1	0	1	1	0	0	0	0	0	1	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	13
Solanecio gigas	0	1	0	0	0	0	1	1	1	0	1	1	0	0	1	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	1	0	0	0	1	0	0	0	0	0	0	13
Afrocarpus falcatus	0	1	1	1	1	1	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	1	0	0	0	1	0	0	1	1	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	15
Acacia abyssinica	0	1	1	1	0	1	0	0	0	0	0	0	1	0	0	0	1	0	0	0	1	1	0	0	0	0	1	1	0	0	1	1	0	1	0	0	0	0	1	1	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	17
Juniperus procera	0	1	0	0	0	0	1	1	1	1	1	1	0	0	0	0	0	0	1	0	0	1	0	1	1	1	1	0	0	0	0	0	0	1	0	0	0	0	0	1	1	0	0	0	0	0	0	1	1	0	1	1	0	1	1	1	0	0	0	0	23
Myrsine melanophloeos	1	1	1	0	0	1	0	0	0	1	1	1	0	0	1	1	0	1	0	1	0	0	1	0	0	0	1	1	0	0	1	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1	0	0	0	1	1	23
Olea europaea subsp.cuspid	1	1	1	1	1	1	1	0	0	0	0	1	0	1	1	0	0	1	0	1	1	0	1	0	1	1	1	0	1	0	1	1	0	0	0	1	1	0	1	0	0	1	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	26
Richness	10	12	9	10	6	10	9	8	5	8	9	7	7	11	8	3	9	7	5	9	7	9	7	5	7	6	12	9	7	5	14	7	6	4	3	8	6	13	8	5	3	14	7	0	6	3	7	5	4	5	4	6	4	4	6	4	2	0	3	4	
distance	0	0	0	0	1	1	1	1	1	1	1	1	2	2	2	2	2	3	3	3	3	4	4	4	4	4	4	4	5	5	5	5	5	6	6	6	6	6	6	6	7	7	8	8	8	8	9	9	9	10	10	10	11	11	12	12	12	12	12	12	

8.4. Appendix 4

Herb plant species in *Eucalyptus* plantations with their dispersal syndromes. Dispersal syndromes are estimated based on seed/fruit morphology using Flora of Ethiopia and Eritrea, Herbarium specimens and other literature sources as well.

Scientific name	Dispersal syndrome	Family	Occurrence
Achyranthes aspera L.	exozoochory	Amaranthaceae	18
Ageratum conyzoides L.	anemochory	Asteraceae	9
Agrocharis incognita (Norman) Heyw. And Jury	exozoochory	Apiaceae	8
Agrocharis melanantha Hochst.	exozoochory	Apiaceae	9
Alchemilla abyssinica Fresen.	endozoochory	Rosaceae	14
Alchemilla pedata A.Rich	endozoochory	Rosaceae	7
Aloe sp.	endozoochory	Aloaceae	1
Anthemis tigreensis J.Gay ex A.Rich.	*anemochory*	Asteraceae	20
Anthriscus sylvestris (L.) Hoffm.	*exozoochory*	Apiaceae	11
Argyrolobium ramosissimum Bak.	*endozoochory*	Fabaceae	12
Arisaema enneaphyllum Hochst. Ex A.Rich.	*endozoochory*	Areaceae	2
Arisaema schimperianum Schott	*endozoochory*	Areaceae	9
Bidens macroptera (Sch.Bip.ex.Chiov.) Mesfin	*exozoochory*	Asteraceae	11
Bidens prestinaria (Sch.Bip) Cufod.	*exozoochory*	Asteraceae	26
Bothriocline schimperi Oliv. and Hiern ex Benth	*exozoochory*	Asteraceae	11
Centella asiatica (L.) Urban	*exozoochory*	Apiaceae	5
Cerastium octandrum A.Rich.	*anemochory*	Caryophyllaceae	10
Cirsium englerianum O. Hoffm.	*anemochory*	Asteraceae	11
Cirsium schimperi (Vatke) C.Jeffrey ex Cufod.	*anemochory*	Asteraceae	13
Clematis hirusta Perr. and Guill.	*endozoochory*	Ranunculaceae	2
Commelina africana L.	*endozoochory*	Commelinaceae	19
Commelina imberbis Ehrneb.Hassk.	*endozoochory*	Commelinaceae	1
Conyza hochstetteri Sch.Bip. ex A.Rich.	*anemochory*	Asteraceae	4
Conyza nana Sch.Bip. ex Oliv. and Hiern	*anemochory*	Asteraceae	2
Conyza steudelii Sch.Bip.ex A.Rich.	*anemochory*	Asteraceae	7
Conyza tigreensis Oliv. and Hiern	*anemochory*	Asteraceae	8
Cordia sp.	*endozoochory*	Verbenaceae	1
Corrigiola capensisWilld.	*endozoochory*	Molluginaceae	2
Crepis rueppellii Sch. Bip	*anemochory*	Asteraceae	9
Cyanotis barbata D. Don	*exozoochory*	Commelinaceae	26
Cyathula cylindrical L.	*exozoochory*	Amaranthaceae	4
Cyathula polycephala Bak.	*exozoochory*	Amaranthaceae	1
Cynoglossum amplifolium Hochst. ex A.Rich.	*exozoochory*	Boraginaceae	11
Cynoglossum geometricum Bak. and Wright	*exozoochory*	Boraginaceae	16

Cynoglosum lanceolatum Forssk.	exozoochory	Boraginaceae	3
Desmodium repandum (Vahl) DC.	exozoochory	Fabaceae	1
Dichrocephala integrifolia (L.f.) Kuntze	exozoochory	Asteraceae	1
Dicliptera laxata C.B.Clarke.	autochory	Acanthaceae	5
Echinops giganteus A. Rich	anemochory	Asteraceae	32
Echinops kebericho Mesfin.	anemochory	Asteraceae	2
Galinsoga parviflora Cav.	anemochory	Asteraceae	4
Galium spurium L.	endozoochory	Rubiaceae	7
Geranium arabicum Forssk.	anemochory	Geraniaceae	12
Glycine wightii (Wight and Arn.) Verdc.	endozoochory	Fabaceae	2
Guizotia schimperi Sch. Bip.ex Walp	anemochory	Asteraceae	0
Haplocarpha schimperi (Sch.Bip.) Beauv.	anemochory	Asteraceae	4
Heracleum abyssinicum (Boiss.) Norman	*anemochory*	Apiaceae	9
Hypoestes forskaolii (Vahl) R.Br.	*anemochory*	Acanthaceae	8
Hypoestes triflora (Forssk.) Roem. and Schult.	*anemochory*	Acanthaceae	33
Kalanchoe densiflora Rolfe.	endozoochory	Crassulaceae	14
Lactuca inermis Forssk.	anemochory	Asteraceae	8
Laggera crispata (Vahl) Hepper and Wood	exozoochory	Asteraceae	8
Launaea intybacea (Jacq.) Beauv.	exozoochory	Asteraceae	8
Malva verticillata L.	*anemochory*	Malvaceae	1
Medicago polymorpha L.	*exozoochory*	Fabaceae	28
Nepeta azurea R. Br.ex Benth.	anemochory	Lamiaceae	2
Ocimum forskolei Benthi.	anemochory	Lamiaceae	5
Oenanthe palustris (Chiov.) Norman	anemochory	Apiaceae	1
Orobanche minor Smith	anemochory	Orobanchaceae	1
Oxalis corniculata L.	endozoochory	Oxalidaceae	23
Pavonia urens Cav.	*anemochory*	Malvaceae	1
Physalis peruviana L.	endozoochory	Solanaceae	3
Pilea rivularis Wedd.	*anemochory*	Urticaceae	1
Pilea tetraphylla (Steudel) Blume	*anemochory*	Urticaceae	3
Pimpinella hirtella (Hochst.) A Rich.	anemochory	Apiaceae	13
Plantago lanceolata L.	endozoochory	Plantaginaceae	9
Plantago major L.	endozoochory	Plantaginaceae	2
Plantago palmata Hook.f	endozoochory	Plantaginaceae	21
*Plectranthus punctatus L'**Hérit**.*	anemochory	Lamiaceae	6
Pouzolzia parasitica (Forssk.) Schweinf.	*anemochory*	Urticaceae	1
Rumex abyssinicus Jacq.	exozoochory	Polygonaceae	20
Rumex nepalensis Spreng.	exozoochory	Polygonaceae	26
Salvia merjamie Forssk.	anemochory	Lamiaceae	11
Salvia nilotica Juss. ex Jacq.	anemochory	Lamiaceae	5
Satureja paradoxa (Vatke) Engl.	endozoochory	Lamiaceae	18

Satureja punctata (Benth.)Briq.	endozoochory	Lamiaceae	5
Scadoxus multiflorus (Martyn) Raf.	anemochory	Amarylidaceae	1
Senecio mannii (Hook.f.) C.Jeffrey	endozoochory	Asteraceae	3
Senecio tamoides DC.	anemochory	Asteraceae	8
Sida teranta L.f.	anemochory	Malvaceae	2
Sium simense Gay ex A.Rich	anemochory	Apiaceae	5
Sonchus bipontini Asch.	anemochory	Asteraceae	2
Stellaria sennii Chiov.	*anemochory*	Caryophyllaceae	14
Stephania abyssinica (Dillon andA.Rich)Walp	endozoochory	Menispermaceae	4
Tagetes minuta L.	exozoochory	Asteraceae	13
Thalictrum rhynchocarpum Dill. and A.Rich.	exozoochory	Ranunculaceae	13
Torilis arevensis (Hudson) Link	anemochory	Apiaceae	12
Trifolium arvense L.	endozoochory	Fabaceae	12
Trifolium rueppellianum Fresen.	endozoochory	Fabaceae	37
Urtica simensis Steudel.	anemochory	Urticaceae	7
Vernonia urticifolia A. Rich.	anemochory	Asteraceae	1
Veronica abyssinica Fresen.	endozoochory	Scrophulariaceae	1

8.5. Appendix 5

Lists of herb species recorded from natural forest (i.e. Chilimo forest) with their occurrence out of four sites

Scientific name	Family	occurrences
Achyranthes aspera L.	Amaranthaceae	2
Acmella caulirhiza Del.	Asteraceae	1
Ageratum conyzoides L.	Asteraceae	1
Agrocharis melanantha Hochst.	Apiaceae	3
Agrocharis melanantha Hochst.	Apiaceae	1
Alchemilla abyssinica Fresen.	Rosaceae	2
Alchemilla pedata A.Rich	Rosaceae	1
Anthospermum herbaceum L.f.	Rubiaceae	2
Argyrolobium ramosissimum Bak.	Fabaceae	1
Arisaema enneaphyllum Hochst. Ex A.Rich.	Areceae	1
Bidens pilosa L.	Asteraceae	1
Bidens prestinaria (Sch.Bip) Cufod.	Asteraceae	2
Cerastium octandrum A.Rich.	Caryophyllaceae	1
Cineraria abyssinica Sch.Bip.ex A.Rich	Asteraceae	1
Cirsium englerianum O. Hoffm.	Asteraceae	1
Commelina imberbis Ehrneb.Hassk.	Commelinaceae	1
Commelina latifolia Hochst.ex A.Rich	Commelinaceae	1
Conyza steudelii Sch.Bip.ex A.Rich	Asteraceae	1
Crepis rueppellii Sch. Bip	Asteraceae	1
Cyanotis barbata D. Don	Commelinaceae	2
Cyathula cylindrical L.	Amaranthaceae	1
Cyathula polycephala Bak.	Amaranthaceae	1
*Cynoglossum geometricum*Bak. and Wright	Boraginaceae	2
Cynoglosum lanceolatum Forssk.	Boraginaceae	4
Dichrocephala integrifolia (L.f.) Kuntze	Asteraceae	2
Dicliptera laxata C.B.Clarke.	Acanthaceae	3
Echinops kebericho Mesfin.	Asteraceae	1
Echinops macrochaetus Fresen.	Asteraceae	1
Galinsoga parviflora Cav.	Asteraceae	1
Gallium simense Fresen.	Rubiaceae	4
Geranium arabicum Forssk.	Geraniaceae	3
Glycine wightii (Wight and Arn.) Verdc.	Fabaceae	4
Gnaphalium tweedieae Hilliard	Asteraceae	1
Haplocarpha schimperi (Sch.Bip.) Beauv.	Asteraceae	1
Heracleum abyssinicum (Boiss.) Norman	Apiaceae	1
Hypoestes forskaolii (Vahl) R.Br.	Acanthaceae	1

Hypoestes triflora (Forssk.) Roem. and Schult.	Acanthaceae	4
Impatiens hochstetteri Warp.	Balsaminaceae	2
Ipomoea purpurea (L.) Roth.	*Convolvulaceae*	2
Kalanchoe densiflora Rolfe	Crassulaceae	1
Lactuca inermis Forssk.	Asteraceae	2
Orobanche minor Smith	Orobanchaceae	1
Persicaria nepalensis (Meisn) Miyabe	Polygonaceae	1
Peucedanum mattirolii Chiov.	Apiaceae	1
Pilea rivularis Wedd.	Urticaceae	2
Pilea tetraphylla (Steudel) Blume	Urticaceae	1
Plantago palmata Hook.f.	Plantaginaceae	2
Plectranthus garckeanus Vatke	Lamiaceae	1
Polygonum nepalense Meisn.	Polygonaceae	2
Rumex nepalensis Spreng.	Polygonaceae	1
Rumex nervosus Val.	Polygonaceae	1
Salvia merjamie Forssk.	Lamiaceae	1
Satureja paradoxa (Vatke) Engl.	Lamiaceae	2
Satureja punctata (Benth.)Briq.	Lamiaceae	1
Senecio mannii (Hook.f.) C.Jeffrey	Asteraceae	1
Senecio tamoides DC.	Asteraceae	1
Sida schimperiana A.Rich.	Malvaceae	1
Solanum nigrum L.	Solanaceae	1
Stellaria sennii Chiov.	Caryophyllaceae	2
Thalictrum rhynchocarpum Dill. and A.Rich.	Ranunculaceae	1
Trifolium arvense L.	Fabaceae	1
Trifolium calocephallum Fresen.	Fabaceae	1
Trifolium multinerve A. Rich.	Fabaceae	1
Trifolium rueppellianum Fresen.	Fabaceae	2
Trifolium semipllosumFresen.	Fabaceae	1
Trifolium Simense Fresen.	Fabaceae	2
Urtica simensis Steudel.	Urticaceae	2
Veronica abyssinica Fresen.	Scrophulariaceae	1

8.6. Appendix 6.

My published paper entiled as *The potential of small Eucalyptus plantations in farmscapes to foster native woody plant diversity: local and landscape constraints* was included as part of my PhD thesis. *Restoration Ecology Vol.23, No.6, pp.918-926 November 2015*

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