Impact of Exotic Tree Plantations on Carbon and Nutrient Dynamics in Abandoned Farmland Soils of Southwestern Ethiopia

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Abstract

The impact of exotic tree plantations in Ethiopia on soil organic carbon (SOC), soil nutrients and decomposition processes was analysed and factors contributing to differences in SOC sequestration identified using the CO2Fix model.

Forest clearing followed by continuous cultivation for 75 years reduced total SOC by 43% (75.4 Mg ha^{-1}), and forest-derived SOC by 73% (128.4 Mg ha^{-1}) within 0-50 cm depth. Total N decreased by 38% (6.8 Mg ha^{-1}) and base cations and available P were significantly decreased within 0-20 cm depth, while δ^{15}N increased within 0-10 cm depth.

Farmland afforestation led to net SOC accretion of 69.6 Mg ha^{-1} (63.3 Mg ha^{-1} tree-derived) under *Cupressus lusitanica* and 29.3 Mg ha^{-1} (24.2 Mg ha^{-1} tree-derived) under *Pinus patula* stands after 20 years. Afforestation with *Eucalyptus grandis* on an area with pasture plus cropping restored SOC stocks to pre-deforestation levels. Total N increased by 5.7 Mg ha^{-1} under *C. lusitanica* and 2.0 Mg ha^{-1} under *P. patula* (0-50 cm soil depth). After afforestation, δ^{15}N decreased more in surface layers with *C. lusitanica* than with *P. patula*. Afforestation had little impact on available P and exchangeable Mg^{++}. Exchangeable K^{+} increased significantly (0-30 cm depth) under *C. lusitanica* and *P. patula*, while Ca^{++} increased significantly (0-5 cm depth) only under *C. lusitanica*. The δ^{15}N pattern in *E. grandis* soil reflected residual effects of C_4 organic matter from pasture.

Leaf litter decomposed in the order *C. lusitanica*> native forest> *E. grandis*~ *P. patula*, whereas fine roots decomposed in order *P. patula*> native forest> *C. lusitanica*~ *E. grandis*. SOC sequestration under the exotic trees might be explained by the higher leaf litter decomposition transferring fresh C to mineral soil and lower fine root decomposition, which occurs directly in mineral soil. The CO2Fix model confirmed that total litter input and proportion of fine woody litter explained differences in SOC. Increasing degree of litter decomposition decreased extractives concentration, holocellulose concentration and *O*-alkyl C spectral regions, and increased spectral regions of alkyl, methoxyl, aromatic and phenolic C and proximate lignin concentration. Fast-growing trees can thus influence SOC and nutrient status within 20 years through their litter input and litter mineralisation rate.

Key words: Afforestation, natural ^{13}C abundance, natural ^{15}N abundance, soil organic carbon, soil nutrients, exotic trees, leaf litter, fine roots, microcosm decomposition, CPMAS ^{13}C NMR, CO2Fix model.

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Dedicated to: my mother, Belaynesh Koyera, and my late father, Lemma Jore

Contents

Appendix

Papers I-IV

This thesis is a summary of the following four papers, which are referred to in the text by their Roman numerals.

I.   Lemma, B., Berggren, D., Nilsson, I. & Olsson, M. 2006. Soil carbon sequestration under different exotic tree species in the southwestern highlands of Ethiopia. Geoderma (*in press*).

II.   Lemma, B. & Olsson, M. 2006. Soil δ^{15}N and nutrients under exotic tree plantations in the southwestern Ethiopian highlands. Forest Ecology and Management (*in press*).

III. Lemma, B., Nilsson, I., Berggren, D., Olsson, M. & Kinker, H. 2006. Decomposition and substrate quality of leaf litter and fine roots from three exotic trees and native forest in the southwestern highlands of Ethiopia. Soil Biology and Biochemistry (submitted).

IV.   Lemma, B., Berggren, D., Olsson, M. & Nilsson, I. 2006. Factors controlling soil organic carbon sequestration under exotic tree plantations: A case study using the CO2Fix model in the southwestern Ethiopia. Forest Ecology and Management (submitted).

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Introduction

Background

Topographical diversity, climatic variability and geological factors have endowed Ethiopia with many different soils, types of vegetation and ecological zones. The topographical features include high mountains, deep gorges, flat-topped plateaus, and rolling plains (Ashagrie, 2004), and elevations vary from 125 m below sea level at Danakil to 4620 m above sea level (a.s.l.) at Ras Dejen (Taddese, 2001). The highlands are part of the vast mountain system in the country with an altitude of above 1500 m, and they constitute the largest massif complex in Africa, which is intersected by the Great Rift Valley (Taddese, 2001; Yirdaw, 2002). Main soil types identified in the country are Lepthosols, Nitisols, Cambisols, Regosols and Vertisols in order of their abundance (MoA, 2000). The natural vegetation ranges from Afro-alpine through dense high canopy montane forest to savannah, scrubland and desert. The agro-ecological zones are traditionally classified mainly based on temperature, rainfall and altitude into: dry hot, dry warm, sub moist warm, sub moist cool, moist cool, cold, moist cold and very cold or alpine (MoA, 2000).

   Large shifts in land use have occurred during the 20th century in the south and southwestern highland regions of Ethiopia. The effect of land use change on soil properties may vary for different soils, vegetation types and ecological zones. The major intensification land use change in the highlands is the deforestation of native forest due to expansion of cropping and grazing land. Afforestation of degraded lands with fast-growing trees species is an increasing method of land rehabilitation (Pohjonen & Pukkala, 1990). The primary goals of the Ethiopian tree-planting programme are to reduce land degradation and provide forest products. An important question is the magnitude of soil improvement after afforestation, the rate of change and the impact of different species, *i.e.* whether fast-growing trees planted on degraded lands can improve the soil after only one rotation.

Deforestation of native forest

Fifty-two percent of the world's forests are concentrated in the tropics, which have the highest rates of deforestation globally (Brown *et al*., 1996). The native forests of Ethiopia have dwindled at an alarming rate (Reusing, 1998). The deforestation of native forest in Ethiopia seems to have occurred relatively early and was extensive in comparison to the situation in other East African countries (Bonnefille & Hamilton, 1986; Siiriäinen, 1996). There is evidence that a major deforestation in the northern highlands took about 2000 years ago (Yirdaw, 2002). However, the disappearance of forests in the south and southwestern highlands has been significant only during the past 100 or more years (Pohjonen & Pukkala, 1990). During this time, the native forest declined from 40% land cover at the beginning of 1900s to approximately 2.7% (Pohjonen & Pukkala, 1990) and is expected to shrink even more due to continued deforestation.

In contrast with most mountain systems outside Africa, the highlands of Ethiopia are very suitable for human habitation. As a result, 88% of the population, 95% of the cropped land and about 75% of the livestock are found on the highlands (Hurni, 1988; EFAP, 1994). With a growing population, the demand for agricultural land has increased. Sedentary agriculture has been accompanied by tree felling to get fuel wood and construction material. Moreover, Ethiopia accommodates the largest number of livestock in Africa. A large proportion is found in the highlands, and has led to grazing pressure. The socio-political instability in the county is another factor which has furthered the heavy deforestation (Yirdaw, 2002). Despite the extensive deforestation that has taken place, there are still forests present in the less accessible and less populated areas of southern and southwestern part of the country. It is also this area that is being deforested the most at present.

The consequences of deforestation followed by continuous cultivation are immense. Deforestation has caused a decline in biological diversity to an extent that some plants have faced local extinction (Yirdaw, 2002). It has been estimated that between 1981 and 1990, 2.5% of the higher plant species were lost due to deforestation in the highlands of tropical Africa (FAO, 1993). A decline in forest cover might imply a loss of forest functions, such as regulation of local climate, maintenance of water quality and supply, erosion control and raw material production (Scott, 2001). Moreover, deforestation is one of the major factors contributing to soil degradation, which results in a net loss of soil carbon and is associated with a deterioration in soil structure. In humid regions, it leads to depletion of base cations and other plant nutrients due to leaching and exploitation of soil fertility in conjunction with subsistence agriculture (Lal, 2003).

Forest plantation

Forest plantations are defined as forest stands that have been established artificially with exotic or indigenous species and that have a minimum area requirement of 0.5 ha, have a tree crown cover of at least 10% of the land cover and a total height of mature trees above 5 m (FAO, 2001). Considering the use of plantations for mitigation of greenhouse gas emissions in the Kyoto Protocol, plantations are classified as afforestation and reforestation. Afforestation is defined as planting new forests on land that has not supported forests in the last 50 years, and reforestation on land that has supported forests in the past 50 years but has been converted to non-forested land (Brown *et al.*, 1986).

Tree planting represents a more intensive land use in the tropics than previously practised (Evans, 1999). Establishment of exotic plantations on degraded land has been a slowly growing land use in Ethiopia since tree planting started over a century ago. However, establishment of large-scale forest plantations did not commence until the 1970s (Pohjonen, 1989). Rural afforestation and reforestation were introduced into government policy only in 1974 (Poschen-Eiche, 1987), the objectives being (1) to satisfy the increasing demand for fuel wood, poles and

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construction material; (2) to relieve the pressure on remnants of native forest; and (3) to promote soil and water conservation and rehabilitation of degraded land. Many of the plantations were established at an altitude above 1500 m a.s.l., where the rainfall ($\geq$ 900 mm) is favourable for fast-growing tree species (Pohjonen, 1989; Pohjonen & Pukkala, 1990). Forest plantations in Ethiopia are mainly monocultures of exotic species belonging to the genera *Cupressus*, *Eucalyptus* and *Pinus* or the native species *Juniperus procera*. In fact *Eucalyptus* and *Pinus* spp. alone account for over 50% of all planting that has taken place in the tropics (Evans, 1999). In the year 2000, the estimated forest plantation area in Ethiopia was 216,000 hectares, and about 2000 hectares of new plantations are established each year (FAO, 2001).

Establishment of exotic species plantation can have several advantages. The relatively fast growth rate of exotic species provides wood to be used for various purposes. In addition, recent studies on tropical tree plantations indicate that exotic species may facilitate the regeneration of native species under the canopy and catalyse the subsequent succession processes (Yirdaw, 2002). Generally, trees protect soil via the litter layer and leaf canopy, thereby decreasing runoff and erosion by increasing water infiltration rates and reducing temperature and moisture fluctuations (Sanchez, Buresh & Leakey, 1997). Trees have beneficial effects that are associated with improved soil structure through root action and inputs of organic matter (Fisher, 1995; Olsson, 2001). They may also improve degraded soils by improving soil nutrient status through increased inputs and reduced outputs, while they can increase availability of nutrients through enhanced nutrient cycling (Rhoades, 1997; Jobbágy & Jackson, 2001). Moreover, forests with fast-growing trees are expected to be excellent options for mitigation of CO_2 emissions through biomass and soil C sequestration (Montagnini & Porras, 1998).

Afforestation and soil properties

Soils under forest play an important role in global nutrient and C cycles (Bouwman & Leemans, 1995; Jobbágy & Jackson, 2000, 2001). Land use change, particularly conversion of forests to agricultural ecosystems, causes a disturbance of the ecosystem and affects several soil properties, particularly the soil organic carbon (SOC) content (Lal, 2005). Many authors have tried to quantify the effect of cultivation on SOC. The conversion of forest to arable land mostly results in a depletion of the SOC stock to between 20-50% of the original level (Detwiler, 1986; Post & Mann, 1990; Davidson & Ackerman, 1993). The loss of SOC depends on the agricultural management, antecedent pools, soil type and climate (Lal, 2005). There are few studies in Ethiopia that quantify the loss of SOC and changes in other soil properties when a native forest is converted to arable land. These studies are also limited to the upper 0-10 cm soil layer for SOC (Solomon *et al.*, 2002; Lemenih, 2004) and to the 0-20 cm layer for soil nutrients (Lemenih, 2004). Therefore, more studies are needed to quantify the influence of land use conversion on SOC and other soil properties. Combined with data from cases that include a reversion of the cultivated land to plantation forests, the results can provide information about the specific impact of afforestation process on soil.

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The SOC pools of many cultivated soils are lower than their potential SOC accumulation capacity due to the reduction in SOC associated with the shift in land use from forest to arable. The difference between the actual and potential SOC pools creates the soil's capacity to sequester SOC (Lal, 2005). Thus, the capacity of soil to sequester C depends on the extent to which the SOC pool has been depleted by past land use and management. Therefore afforestation of degraded agricultural land is likely to reverse some of the soil degradation and lead to an enhancement (sequestration) of SOC stock (Silver, Ostertag & Lugo, 2000). Gill & Abrol (1990) assessed the influence of *Casuarina equisetifolia* and *Eucalyptus tereticornis* on SOC following afforestation. They observed an increase in SOC under both species, but the increase in SOC was faster under *C. equisetifolia*. Afforestation with an appropriate choice of tree species is therefore important. The species in question should be suitable for use in reclamation and rehabilitation. There is now much interest in using plantation trees to reclaim degraded and marginal agricultural land (Evans, 1999).

Despite the advantages that result from establishing plantations, the precise effects of different tree species on soil properties remain poorly understood (Binkley, 1995; Davis, 1998). The effects of trees on soil properties are always examined in relation to other land use alternatives and management systems (Olsson, 2001). A number of studies have been made in different parts of the world concerning the effects of forest plantations on soil, but most of these studies focus on soil property changes under plantation forests compared to soils under a nearby native forest, soil under another plantation species (mainly N_2-fixing) or soil under grassland. Few studies in the tropics have reported soil property changes following plantation establishment on sites that were previously used for crop production (Binkley & Resh, 1999; Lemenih, Olsson & Karltun, 2004). Furthermore, Post & Kwon (2000) reported from a literature review that there is considerable variation in SOC accumulation rates following forest plantation or secondary forest succession on former agricultural land. Those authors collated results from different studies with varying accumulation patterns, which either showed a decreasing rate or increasing rate of SOC accumulation after conversion. Due to these inconsistencies and the scarcity of information on Ethiopia, more studies are needed to evaluate the potential for SOC accumulation under productive plantation forests on abandoned farmland. Moreover, few studies have investigated the influence of such plantations on soil nutrients. One example is the work of Lemenih, Olsson & Karltun (2004), who studied the effects of two tree species on soil properties at reforested sites on former arable land in southern Ethiopia. However, studies that quantify the effect on soil properties of specific exotic trees used for afforestation on former abandoned farmland are still scarce.

Factors affecting soil organic carbon

Understanding the mechanisms and factors of SOC dynamics in forest soils is important in identifying and enhancing natural sinks for C sequestration. SOC
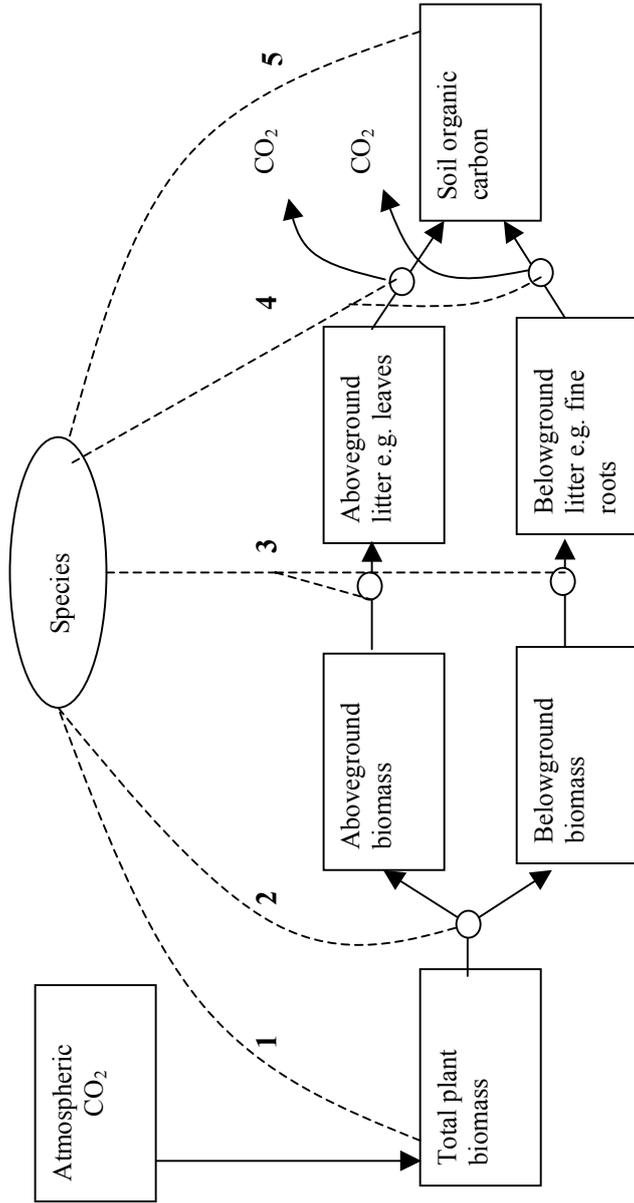
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Fig. 1. Regulation of soil organic carbon dynamics (**5**) by tree species through net primary production (**1**), allocation (**2**), litter input (**3**), and decomposition (**4**). (Adapted with modification from *http://www.nrem.iastate.edu/ECOS/overview.html*)

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plays a major role in soil productivity because it represents the dominant reservoir and source of major plant nutrients (*e.g.* N, P, S). It also influences pH, cation exchange capacity (CEC), anion exchange capacity (AEC), water status and soil structure. In addition, current interest in sequestering SOC in forest soils has increased because of the threat of climate change. The concentration of CO_2 in the atmosphere has increased in the past two centuries, mainly because of fossil fuel combustion, cement production, biomass burning and land use changes (Lal, 2003). A number of factors and processes are believed to determine the direction and rate of change in SOC storage when plantations are established on arable land. Factors that may be important for increasing SOC storage include: (i) litter production (both above- and below-ground); (ii) litter quality; (iii) placing organic matter deeper in the soil either directly by increasing below-ground inputs or indirectly by enhancing surface mixing by soil organisms; (iv) increasing physical protection through either intra-aggregate or organo-mineral complexes; and (v) microclimate change (Lugo & Brown, 1993; Post & Kwon, 2000).

The amount of SOC storage is, however, controlled primarily by two fundamental factors: the quality and quantity of input by net primary production and its decomposition rate (Lutzow *et al.*, 2006). In most terrestrial ecosystems, the majority of Net Primary Production (NPP) is shed in the form of plant litter, which originates from above- and below-ground plant organs (Swift, Heal & Anderson, 1979). Leaves account for most of the above-ground fine litter in forests (Bray & Gorham, 1964). Fine roots also represent a large dynamic part of the below-ground biomass and a significant proportion of NPP in forest ecosystems (Vogt, Grier & Vogt, 1986). Fine roots perform different physiological functions than leaves, which could lead to differences in chemical composition and decomposition. Tree species differ in their allocation of C to above- and below-ground components and in their fine root mortality (Cairns *et al.*, 1997). There is also a considerable interspecific variation in the quality and quantity of litter produced by different tree species (Fig. 1) (Aerts, 1997).

Decomposition of both above-ground and below-ground litter is an important process in most terrestrial ecosystems because of its role in regulating the build-up of SOC and release of nutrients (Schimel, 1995). Through the process of decomposition, plant litter eventually results in SOC (Fig. 1). The rate of litter decomposition depends on local climate, substrate quality, and the activity of decomposer populations (Swift, Heal & Anderson, 1979). However, substrate quality of leaf litter and fine root litter is the most important factor determining the decomposition rate at a local level (Aerts, 1997; Silver & Miya, 2001; Chen *et al.*, 2002). This is most evident in those tropical regions where high temperature is combined with an adequate moisture supply (Aerts, 1997). A number of studies conducted to determine substrate quality factors controlling decomposition rate have reported factors such as concentrations of nutrients, concentrations of proximate carbon components and the relative proportions of these (Schlesinger & Hasey, 1981; Taylor, Parkinson & Parsons, 1989). However there is no particular single factor of substrate quality to predict all litter decomposition. Studies on the impact of trees on soil properties supplemented with leaf litter and fine root decomposition studies are essential in understanding SOC and nutrients in a forest

ecosystem. There are few litter decomposition studies on exotic plantation trees in Ethiopia. One example is the work of Lisanework & Michelsen (1994), who studied decomposition and nutrient release in native forest and tree plantations. There have been many decomposition studies of different *Eucalyptus* and *Pinus* species with different treatments (*e.g.* age, irrigation, fertilisation), most of which have been carried out in sub-tropical and Mediterranean environments (*e.g.* Gholz *et al.*, 1985; O'Connell, 1988; Baker, Will & Oliver, 1989; Ribeiro, Madeira & Araujo, 2002). However, there is little information on the decomposition of *Cupressus* and particular of *Eucalyptus grandis* and *Pinus patula*. Furthermore, few studies have investigated both leaf litter and fine root decomposition simultaneously.

Objectives

The overall objective of this study was to determine the impact of fast-growing exotic tree plantations on the soil carbon and nutrient status of abandoned cropping land and to assess the influence of tree-related factors. Specific objectives were to:

1. Evaluate the rate and magnitude of changes in SOC based on natural ^{13}C abundance following deforestation and conversion into cropping land of native forest and subsequent re-conversion of the cultivated land into exotic tree plantations (Paper I);

2. Evaluate the rate and magnitude of changes in soil nutrients and natural ^{15}N abundance following deforestation and conversion into cropping land and subsequent re-conversion into exotic plantations (Paper II);

3. Determine differences in leaf litter and fine root mineralisation rates between the exotic tree species, characterise their chemistry and relate the relative mineralisation rates to their chemistry (Paper III);

4. Assess the factors controlling differences in SOC sequestration of the exotic tree plantations using the CO2Fix model (Paper IV).

Materials and Methods

General descriptions of the site

The study area, the Belete Forest and its adjoining farmland (7°33'N, 36°35'E) is situated in Seka Chekorsa district of Jimma zone in Ethiopia, about 417 km from the capital city, Addis Ababa, in the south-western highlands (Fig. 2). Climatic data for the period 1983-2003 were obtained from the National Metrological Station, Addis Ababa, for the closest meteorological station at Jimma, 38 km from the study site. During this period the mean annual precipitation was 1517 mm, with high variation from year to year giving a range from about 1100 to 2000 mm

year^{-1}. The rainfall pattern in this area is distinctly uni-modal, with low rainfall between January and February, and gradually increased precipitation to the peak period between May and September, and then decreasing values in November and December. The mean annual temperature for the period covered by the data was 19.4 $^{\circ}$C, and the mean annual minimum and maximum air temperature were 11.4 and 27.4 $^{\circ}$C, respectively. The air temperature fluctuated widely between day and night. The sampling sites were located within the same agro-climatic zone, and situated between 2100 m and 2340 m a.s.l.
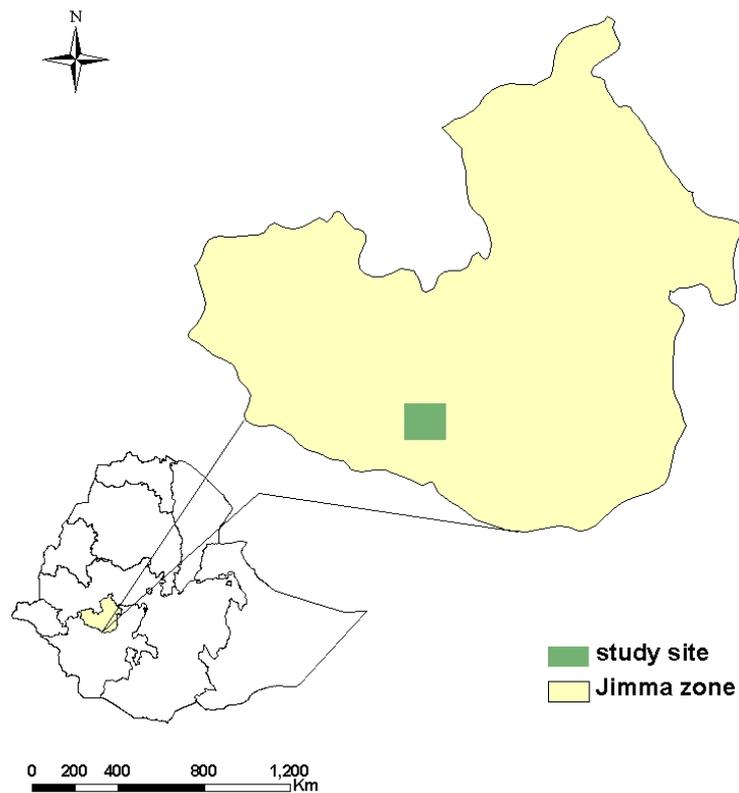
*Fig. 2*. A map of Ethiopia showing the location of the study site.

Ethiopia falls within the geological structural unit of the Horn of Africa. The underlying basement rock consists of intensively folded and faulted Precambrian material, and is overlain by Mesozoic marine sediments and Tertiary volcanic rocks (Tefera, Chernet & Haro, 1999). The study area is associated with Jimma Volcanics, which is abundant in rhyolites and trachybasalts (Tefera, Chernet & Haro, 1999). The soils in the parts of the forest and farmland under this study were suggested to be Humic Nitisols (FAO, 1998; JICA, 1998) with a clay loam texture and dark reddish-brown colour.

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Afforestation at Belete

Belete Forest is one of the remaining mountain rainforests in southwestern Ethiopia. The Ethiopian government designated the area part of 'Belete-Gera Forest Priority Area' in 1994. Belete Forest is managed with multiple objectives, including contributing to soil, water and biodiversity conservation and protecting and developing sustainable use of the forest (Scott, 2001). It consists of a mixed deciduous native forest and recently established plantation forests, and covers an area of 18,324 ha (JICA, 1998). A number of tree, shrub and climber species dominate the mixed deciduous forest. The dominant tree species include: *Ilex mitis* (L.) Radelk, *Syzygium guineense* (Willd.)DC., *Verpis daniellii* (Pichi.Serm.) Kokowaro, and *Allophylus abyssinicus* (Hochst.) Radlkofer. There is also a considerable amount of trees of *Macaranga capensis* (Ball.) Sim and *Croton macrostachyus* Del. Dominant climbers include *Hippocratea getzei* Loes. and *Rubus apetalus* Poir, and dominant shrubs are *Maytenus arbutifolia* (Hochst. Ex A.Rich.) Wilczek and *Dracaena afromontana* Mildbr. The native forest consists of plants that have a C_3 photosynthetic pathway or Calvin cycle. Such plants incorporate CO_2 first into a 3-C compound in the Calvin cycle and have a low carbon isotope ratio ($^{13}C/^{12}C$) because they discriminate against the ^{13}C isotope (Farquhar, Ehleringer & Hubick, 1989). The plantation forest has an area of 918.7 ha, and is mainly composed of *Eucalyptus saligna*, *E. grandis* (Hill ex Maiden), *E. citriodora*, *Cupressus lusitanica* (Miller), and the pine *Pinus patula* (Schiede and Deppe) (JICA, 1998).

The plantations have been established as even-aged monocultures. The area of the current plantation forests under *C. lusitanica* and *P. patula* was deforested around 1928, at the same time as the area under the farmland, and was used for crop production, mainly maize, until it was set aside in 1981 and afforested in 1983 as part of the government afforestation programme. The area under *E. grandis* was also deforested in about the same period, but it was first used for pasture before being converted to cropping in 1963, and was finally afforested in 1983 under the national programme. Trees were planted from containerised seedlings which involves pit planting (manual digging of a planting hole), a common practice in Ethiopia. The trees were planted at 2.5 m spacing within and between rows (initial density 1600 trees ha^{-1}). The aim of this afforestation programme was to create a buffer zone around the native forest in order to divert pressure away from it. Aerial photographs of most of the forest area taken in 1975 and 1996 were used to locate plantation areas that were formerly used as arable land (Fig. 3). Informal interviews with local elders living close by the study site were used to support this information and to obtain information about the conditions after abandonment.

As in most parts of Ethiopia, the farming system practised on the cleared land was characterised by small households and traditional, rain-fed cropping, usually neither limed nor fertilised. The present and former agricultural crop was mainly
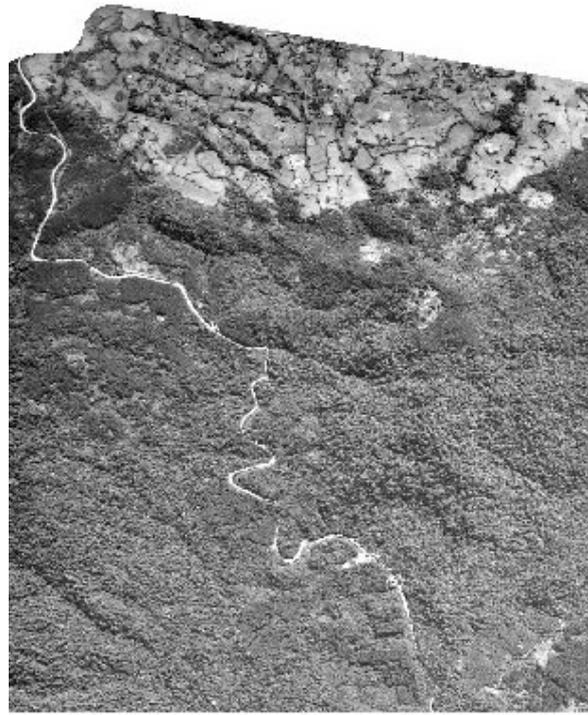
Land cover 1975
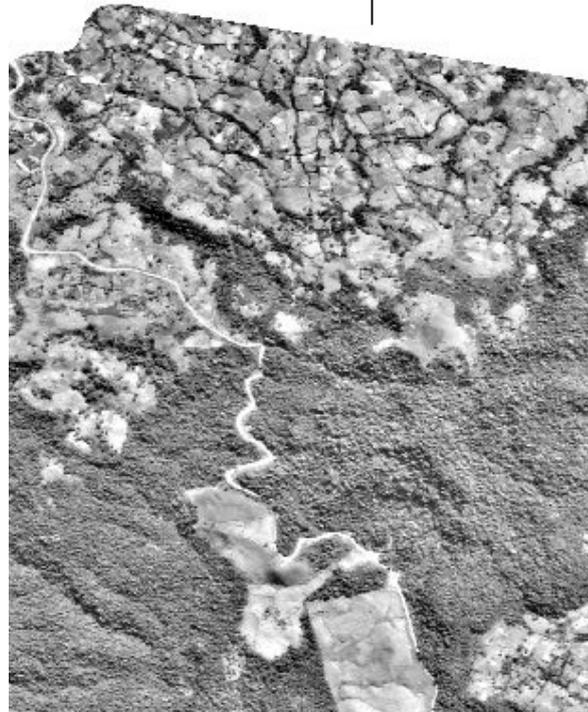
Land cover 1996

Fig. 3. Clips of aerial photographs that show changes due to afforestation of exotic trees in the study area between 1975 and 1996.

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maize (*Zea mays* L.) and to a smaller extent teff (*Eragrostis tef*) and sorghum (*Sorghum bicolor* (L.) Moench), and harvest was once or twice a year. All of these crops are C_4 species with a photosynthetic pathway or Hatch-Slack cycle, which incorporate CO_2 into a 4-C compound. They have higher carbon isotope ratio than C_3 plants, and discriminate less against ^{13}C than C_3 plants during photosynthesis (Farquhar, Ehleringer & Hubick, 1989). Tillage involved a simple oxen-plough with ploughing depth varying between 10 and 12 cm (Solomon *et al.*, 2002), which is typical of traditional soil cultivation in Ethiopia.

The study approach

Studies that evaluate changes following land use conversion require decades to provide conclusive results. On the other hand, studies that use comparisons between existing land use types provide more timely results, but are subject to the risk of confounding effects due to natural spatial variation. In Papers I and II, we used the latter approach, comparing replicates of existing land use patches. We tried to minimise the risk of confounding effects through careful site selection using the criteria slope form, soil depth, soil texture, soil mineralogy, minimum disturbance (*e.g.* roads, footpaths), size and accessibility. A comparative approach has been frequently used when data on SOC and soil nutrients prior to land use conversion are not available. For example, most of the data reviewed by Post & Kwon (2000) on potential rates of SOC accumulation following the conversion of arable land to perennial vegetation were derived using a comparative method. A necessary assumption in Papers I and II was that SOC and other soil properties were identical at each of the present plantation sites and at the reference farmland prior to plantation establishment. This assumption is reasonable given the proximity of the reference farmland to its companion plantations and the selection criteria. Soil nutrients and SOC under *Eucalyptus grandis* stands were assumed to be lower than under native forest prior to plantation in the pasture followed by a period of cropping (Papers I and II). Since *Eucalyptus grandis* was established on mixed-use land (pasture followed by arable) and a reference area was not available at the site, the influence of *Eucalyptus grandis* was evaluated by comparing it with the native forest as a reference, assuming that soil properties in the native forest have not changed significantly. The influence of deforestation on soil nutrients and SOC was evaluated by comparing arable land with native forest as a reference.

A less frequently used method in decomposition studies involves determination of changes in chemical structure of organic components, starting from the surface litter layer and proceeding through more decomposed layers down to the mineral soil (Zech *et al.*, 1997). In Paper III we used this approach to characterise changes in elements and organic components by comparing fresh leaf litter and decomposed litter (from the Oe layer) on the forest floor. Decomposition of litter can be studied in microcosms in the laboratory (*e.g.* Gillon, Joffre & Ibrahima, 1994; Li *et al.*, 2001). In such microcosm studies, CO_2 production resulting from litter decomposition (litter mass loss) is determined under standardised moisture and temperature conditions. This method was used in Paper III since it provides

information about potential decomposition rates and correlates with field litterbag studies (Aerts & de Caluwe, 1997).

To assess factors contributing to the differences in SOC under exotic trees, a modelling approach using the CO2Fix model was employed (Paper IV). CO2Fix is a C accounting model that has been described in detail in some studies and the model manual (Nabuurs & Schelhaas, 2002; Masera *et al*., 2003; Schelhaas *et al*., 2004). It consists of three modules - biomass, soil and wood products. The biomass module estimates the annual litter input (foliage, branch and root) to the soil module using turnover parameters. The so-called Yasso soil module takes into account the amount of litter, initial litter quality and the effect of climate on decomposition to estimate soil carbon. Litter data enters the soil module in terms of size of the litter supply as non-woody litter (foliage and fine root), fine woody litter (branches and coarse root) and coarse woody litter (stem) and is distributed into the contents of different classes of organic compounds. The soil model has been validated for its soil carbon estimates, mass loss estimate and ability to appropriately describe the effects of climate on decomposition rates within a wide range of environments, including the tropics (Liski *et al*., 2003, 2005; Palosuo *et al*., 2005). The CO2Fix model has been used to estimate biomass and soil carbon in a range of environments (*e.g*. Nabuurs & Mohren, 1993; Nabuurs & Schelhaas, 2002; Masera *et al*., 2003).

Sampling and analysis

Sampling and characterisation of soil properties and plant materials (Papers I and II)

Soil, litter and root samples were collected from four random square plots (100 m^2 each) in each type of land use system, i.e. farmland, native forest and plantations of *C. lusitanica, P. patula* and *E. grandis*. In each plot, two soil pits were dug at randomly chosen points, and five soil samples were taken from the 0-5, 5-10, 10-20, 20-30 and 30-50 cm soil layer in each pit using a core sampler (diameter = 7.2 cm). The upper mineral soil sample included thin Oe and Oa layers. The samples from the two soil pits in each plot were bulked to give one composite sample for each soil depth. All soil samples were air-dried (about 25 ^{o}C), and sieved (mesh size 2 mm) to remove coarse living roots and gravel, while decaying coarse organic material, such as dead roots, was crushed and then mixed into the sieved fine earth (< 2 mm).

Dry bulk density was calculated by dividing the oven dry mass at (105 ^{o}C) of the < 2 mm fraction by the volume of the core, taking volumes of roots and gravel into account. Soil pH was determined in a 1:2.5 (v/v) soil:water suspension. Exchangeable base cations (Ca^{2+}, Mg^{2+} and K^+), cation exchange capacity and base saturation were determined after extraction with 1 M NH_4OAc (pH =7) and cation analysis using atomic absorption spectroscopy. Exchangeable acidity of the extract was analysed by titration with 0.02 M NaOH after extraction of 10 g mineral soil using 1.0 M KCl. Available P was analysed according to the method of Olsen et al. (1954).

Soil, litter and root samples were oven-dried at 70 $^{\circ}$C for 24 h and ground before being analysed for natural abundance of ^{13}C and ^{15}N and C and N content. It was assumed that the analysis of soil carbon closely estimated the true organic C content, since tests with HCl in all soil pits did not show any signs of carbonates. Furthermore, carbonates are unlikely to occur due to the climatic and geological setting. Analyses were performed on an automatic, on-line carbon and nitrogen analyser coupled to an isotope ratio mass spectrometer (ANCA-NT solid/liquid preparation module coupled to a model 20-20 IRMS, Europa Scientific Ltd., Crewe, UK) using the calibration method suggested by Ohlsson & Wallmark (1999). Variations in the ^{13}C/^{12}C and ^{15}N/^{14}N ratios are reported in relation to the Vienna Pee Dee Belemnite and atmospheric N$_2$, respectively, and expressed in δ units, which represent parts per thousand (‰) deviation of the ratios from the standards.

Litter sampling, litter quality analysis and microcosm decomposition (Paper III)

Newly shed leaf litter (leaves or needles) and decomposed litter (from the Oe layer of the forest floor) were collected under native forest and plantations. In each forest stand, leaf litter (fresh and decomposed) was collected from four randomly selected 10 x 10 m^2 plots, bulked to composite samples and air-dried. Live fine roots (< 2 mm) were dug up from the upper 50 cm of the soil from three randomly sited 2 x 2 x 0.5 m^3 soil pits under each stand. After collection, samples were kept in a refrigerator overnight. Thereafter they were gently washed and dried at 60 $^{\circ}$C for 8 h, bulked to one composite sample per stand and stored in polyethylene bags at room temperature until analysis.

Ground samples (< 1 mm) of fresh leaf litter, decomposed litter and fine roots were used for element, organic fraction and ^{13}C NMR analysis. Organic C and total N (mostly organic N) were analysed with a LECO CNS-1000 analyser. The total concentrations of P, S, Ca, Mg and K were determined in aqueous extracts after wet digestion with concentrated HNO$_3$ by Inductively Coupled Plasma Emission Spectroscopy (Spectro Analytical Instruments). Water-soluble extractives (WSE) were extracted with distilled water (21 $^{\circ}$C) for 24 h. Non-polar extractives (NPE) were removed using dichloromethane in a Soxtec® apparatus. After sub-samples were hydrolysed with H$_2$SO$_4$ (Bethge, Rädeström & Theander, 1971), the acid-insoluble residue (Klason lignin, hereafter simply referred to as lignin) was determined. The holocellulose concentration was obtained by difference. In the acid-soluble fraction, sugar monomers (arabinose, xylose, mannose, galactose and glucose) were determined using methods described by Theander & Westerlund (1986). Ash content was determined as loss on ignition at 525 $^{\circ}$C according to ISO 1762, 2001. The amount of each fraction was reported on an ash-free dry organic matter basis.

The solid-state ^{13}C NMR spectra were obtained on a Bruker DSX 200 operating at a frequency of 50.3 MHz. The cross polarisation magic angle spinning (CPMAS) technique was applied with a frequency at 6.8 kHz. A contact time of 1

ms and a 90° ^{1}H-pulse width of 5.3 µs were used for all spectra. The ^{13}C chemical shifts were calibrated to tetramethylsilane (= 0 ppm) and were calibrated with glycine (176.04 ppm). The NMR spectra of litter samples were divided into the following chemical shift regions: alkyl C (0-45 ppm); methoxyl C (45-60 ppm); O-alkyl C (60-90 ppm); di-O-alkyl C (90-110 ppm); aromatic C (110-140 ppm); phenolic C (140-160 ppm); and carbonyl C (160-185 ppm). Areas of the chemical shift regions were integrated to determine the relative contribution of the respective C groups and are expressed as relative intensity.

Fresh leaf litter and fine root decomposition was studied in laboratory microcosms, consisting of glass jars equipped with a hole in the lid to permit free gas exchange. Leaf litter (6 g) or fine roots (10 g) were placed in the jars and moistened to 70% of their water holding capacity with distilled water. Three laboratory replicates were used for each combination of organic substrate and forest stand. Soil suspensions were prepared by shaking 1 g of soil with 1 L of distilled water for 5 minutes. Portions (5 mL) were extracted from the suspensions and added to each jar in order to inoculate microorganisms to the leaf litter and fine root samples. The jars were incubated at 15 ^{o}C for 120 days (leaves) or 121 days (roots). Distilled water was added weekly to keep the moisture content at a constant value. Readings of CO_2 production during incubation were made by gas chromatography on nine occasions at increasing intervals of days, as described in detail in Paper III.

Stand parameter, litterfall and fine root biomass measurements and parameterisation (Paper IV)

Stand parameters (height, diameter at breast height (DBH), tree stocking) were measured in four random circular plots of 10 m radius in each plantation. Dominant height was mainly used to select the appropriate yield tables of trees from Ethiopia and Eastern Africa. Branch and leaf biomass equations were also found in the literature for Ethiopian or other East African conditions, but root biomass data were scarce and collected from reviews of the tropics (Paper IV). Twenty years of local climate data for the site were collected from the nearest metrological station. Litter quality characteristics of the plantation were determined in Paper III. Using these and other parameters for the tree species or for trees in the same genera from the literature, the model was parameterised to estimate the soil carbon sequestered at stand level.

Leaf litter data for a year and fine root biomass data were used to compare the litter production of the biomass module of CO2Fix. For litterfall measurements, eight wooden frames with a wire mesh were placed randomly under each plantation forest type. The litter falling in each wooden frame was collected at approximately monthly intervals for a year and reported on an air-dry weight basis. For root biomass, the soil was sampled to 50 cm depth with a core sampler in three randomly selected soil profiles. Roots were first separated by hand from the soil core samples and then the remaining soil was soaked and agitated, after which floating roots were removed with a sieve. The hand-picked roots and roots from the sieve were washed with care and dried at 65 ^{o}C for 24 h before weighing.

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As a measure of general microclimatic differences between plantations, we measured leaf area index (LAI) with the LAI-2000 Plant Canopy Analyser (Li-Cor Inc., Lincoln, Nebraska, USA) under each plantation.

Data analysis

Total SOC stock, total soil N and available P within a soil layer were converted to an area basis (mass ha^{-1}) according to the following equations:

$$SOC \ (Mg \ ha^{-1}) = SOC \ (g \ kg^{-1}). \ z. \ \rho_b . \ 10 \qquad (1)$$

$$Total \ N \ (Mg \ ha^{-1}) = total \ N \ (g \ kg^{-1}). \ z. \ \rho_b. \ 10 \qquad (2)$$

$$Avail. \ P \ (kg \ ha^{-1}) = avail. \ P \ (mg \ kg^{-1}) \ . \ z. \ \rho_b. \ 10 \qquad (3)$$

where z = sampled soil layer thickness (m), and ρ_b = bulk density (Mg m^{-3}).

Sampling was based on depth, and the depth of the soil layer had to be adjusted to avoid errors due to differences in bulk density (Veldkamp, 1994) in calculation of SOC stock, total nitrogen (TN) and available P. Calculations for equivalent soil masses and correcting thickness of soil layers were carried out as shown in Ellert, Janzen & McConkey (2001).

The SOC pool was divided into carbon of C_3 or C_4 vegetation origin. The percentage of the SOC derived from C_4 plants (% SOC_4) and from C_3 plants (% SOC_3) in each layer was calculated according to the following equations (Bashkin & Binkley, 1998; Rhoades, Eckert & Coleman, 2000):

$$\% \ SOC_4 = [(\delta - \delta o)/(\delta c - \delta o)].100 \qquad (4)$$

$$\% \ SOC_3 = 100 - \% \ SOC_4$$

where δ is the $\delta^{13}C$ value of the soil sample, δo the $\delta^{13}C$ value of the soil from the natural forest, and δc is the $\delta^{13}C$ value of litter and roots from the agricultural site.

SOC stock was multiplied by % SOC_3 and % SOC_4 to obtain SOC derived from C_3 and C_4 vegetation. Net C change (Mg ha^{-1}) in SOC_3, SOC_4 and total carbon due to 20 years of *C. lusitanica* and *P. patula* growth was calculated as the difference between the pools in these plantations and the farmland. Assuming a linear increase, the rates of SOC and total N accretion (Mg ha^{-1}y^{-1}) were computed by dividing the SOC pools and total N respectively by 20, *i.e.* the age of the plantation trees.

Cumulative CO_2-C loss from litter and fine roots over time was described by fitting the following first order kinetics equation to data (Olson, 1963):

$$Ln \ (X/Xo) = - \ kt \qquad (5)$$

where Xo is the original carbon of litter, X is the amount of carbon remaining after time t, and k is the decomposition constant (d^{-1}).

Data were analysed using analysis of variance (ANOVA) with the Minitab statistical software, release 14 (Minitab, 2003). The significance (p<0.05) of the differences between land uses and tree species at each depth in $\delta^{13}C$, total SOC, SOC_3, SOC_4, $\delta^{15}N$, total N, C:N, available P and base cations was tested using one-way ANOVA. One-way ANOVA was also used to detect the statistical significance (P<0.05) of differences in cumulative CO_2-C evolution among litter types during decomposition in the measurement period. Tukey's method was used for pair-wise comparisons of means. The two-sample T test was used to test the significance of differences in cumulative CO_2-C released from leaf litter and the corresponding fine roots. Pearson's correlation was used to estimate the strength of association between variables (Paper II). The Spearman rank correlation coefficient was used to examine the relationship between rate constants and cumulative C mineralised with litter chemistry. The wet chemical analyses and elemental analyses were carried out on composite samples and were interpreted qualitatively.

Results and discussion

The impact of exotic trees on soil organic carbon (Paper I)

SOC pools in native forest, farmland and plantations of *C. lusitanica*, *P. patula* and *E. grandis* are shown in Fig 4. Deforestation of native forest followed by continuous cultivation depleted the SOC by 43% (75.4 Mg ha^{-1}). The average proportion of the native forest-derived SOC that was lost amounted to 73% (128.4 Mg ha^{-1}), and the SOC from C_4 crops that was added during continuous cultivation was 53.0 Mg ha^{-1}. Reviews estimate that forest clearing and subsequent cultivation result in SOC losses in the range 30-50% over several decades (Detwiler, 1986; Johnson, 1992). Our findings fell in the upper part of this range. Tillage breaks up aggregates and exposes organo-mineral surfaces to decomposers and SOC losses are likely to be higher if soil is ploughed to greater depth than in traditional farming (Lemenih, Olsson & Karltun, 2004). Cropping with sugar cane has been shown to decrease the total SOC by 24% and the original forest-derived SOC by 70% in the upper 30 cm after 50 years (Rhoades, Eckert & Coleman, 2000). Residue inputs from the cropping system used could also influence the total SOC loss. For instance, the net loss of SOC reported in Rhoades, Eckert & Coleman (2000) was much lower than the loss of the original forest-derived SOC compared with our site, which could be mainly due to higher inputs of crop residues. The observed net loss of SOC in farmland was most likely due to accelerated decomposition of SOC following cultivation, together with reduced litter input from the farming system (Solomon *et al.*, 2002).

Afforestation of former farmland at Belete led to total SOC accretion of 69.6 Mg ha^{-1} under *C. lusitanica* and 29.3 Mg ha^{-1} under *P. patula* (Fig. 4). Assuming linearity, SOC increased at a rate of 3.5 Mg ha^{-1} y^{-1} under *C. lusitanica* and 1.5 Mg ha^{-1} y^{-1} under *P. patula*. Similarly, Zou & Bashkin (1997) found a 2.8 Mg ha^{-1} y^{-1}

SOC increase in a Hawaiian *Eucalyptus* plantation growing on an abandoned sugar cane farm for 10 years. Garten (2002) estimated an accumulation rate of 0.4
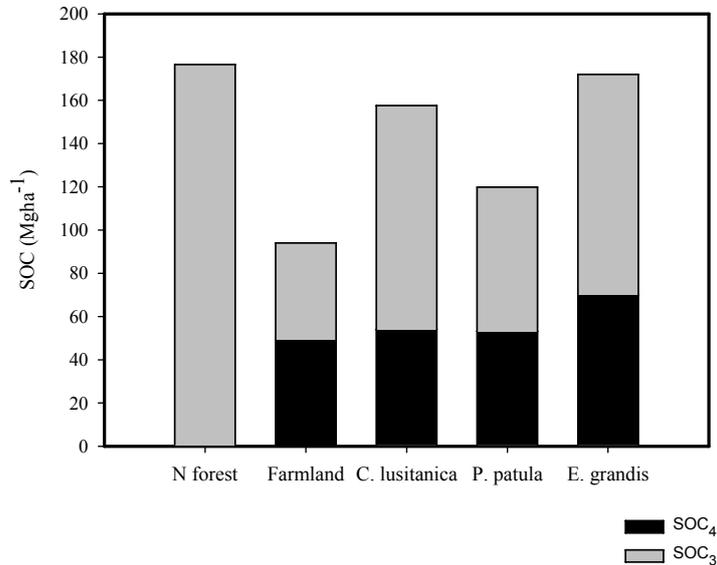

Fig 4. SOC pools within 0-50 cm depth under different land uses and trees species.

to 1.7 Mg ha^{-1} y^{-1} in the first decade following plantation of farmland. Lemenih, Lemma & Demel (2005) found a 1.6 Mg ha^{-1} y^{-1} increase under *C. lusitanica* following 15 years of reforestation of farmland, while Bouwman & Leemans (1995) estimated a SOC accretion rate of 1.7 Mg ha^{-1} y^{-1}.

The SOC derived from plantation origin (SOC$_3$) was 63.3 Mg ha^{-1} for *C. lusitanica* and 24.2 Mg ha^{-1} for *P. patula*. Total SOC derived from C$_4$ crop (mainly maize) origin (SOC$_4$) was 6.2 Mg ha^{-1} in the *C. lusitanica* stand and 5.1 Mg ha^{-1} in the *P. patula* stand (Paper I). The net increase in SOC$_3$ was equivalent to 3.2 and 1.2 Mg ha^{-1} y^{-1} under *C. lusitanica* and *P. patula*, respectively, while the net increase in SOC$_4$ was equivalent to 0.3 Mg ha^{-1} y^{-1} under these trees. Bashkin & Binkley (1998) investigated the influence of *Eucalyptus* plantation on SOC on a former cane farm and found that SOC$_3$ increased at a rate of 1.2 Mg ha^{-1} y^{-1} in the upper soil layers, while SOC$_4$ decreased at a rate of 1.0 Mg ha^{-1} y^{-1} in the lower soil layers. Rhoades, Eckert & Coleman (2000) found that secondary forest re-colonisation on an abandoned sugar cane farm increased SOC$_3$ by 3 Mg ha^{-1} y^{-1} and decreased SOC$_4$ at a rate of 1.1 Mg ha^{-1} y^{-1} compared with the sugar cane farm. In contrast to these studies, which showed a decline in SOC$_4$, the present study showed an increase. This could be an effect of the pasture grass species that colonised the land for 2 years after abandonment and continued to grow until decreased and annihilated under *P. patula* and *C. lusitanica* respectively.

The total SOC under *E. grandis* was not significantly different from that under native forest (Paper I). Since it had been planted on mixed-use land (pasture followed by cropping) and there was no appropriate reference, we hypothesised that *E. grandis* reversed the loss in the previous land uses. Consistent with the assumed increase in SOC under *E. grandis* in our study, a significant increase in SOC by a factor of 2.1 was found under *E. grandis* in Cameroon (Tchienkoua & Zech, 2004). Furthermore, SOC_4 in the 10-50 cm layer under the *E. grandis* stand in the present study was significantly higher than under farmland and plantation, which indicates a higher input from pasture.

The difference in SOC sequestration under exotic trees could be attributed to the differences in litter inputs and fractions of litter types (Paper IV) and to the differences in decomposition of leaf litter and fine roots (Paper III). Faster decomposition of leaf litter enhances transfer of fresh carbon to mineral soil (Polglase et al., 2000). During decomposition parts are leached from the litter layer and organic layer in a form of dissolved organic carbon (DOC) into the mineral soil (Kalbitz et al., 2000; Fröberg et al., 2005). Microbial activity and other factors such as temperature, wet-dry cycles enhance mineralization and DOC release, and precipitation and water fluxes increase DOC leaching (Kalbitz et al., 2000). In mineral soils leached DOC is strongly adsorbed to Al and Fe oxides and hydroxides and clay minerals such as kaolinite that result in a stabilisation of C in soils (Kalbitz et al., 2000). Fine roots directly place carbon in mineral soil (Post & Kwon, 2000), and therefore slower decomposition of fine roots may favour C sequestration. Higher SOC sequestration in the *C. lusitanica* stand was consistent with faster decomposition of leaf litter and slower decomposition of fine roots (Paper III). However, the general increase in SOC under the tree species could also be influenced by the mineralogy (kaolinite, Fe oxides and amorphous Fe minerals) and the clay content of the soil (Paper I). The minerals stabilise soil carbon through organo-mineral complexes and protection through aggregate formation. Osher, Matson & Amundson (2003) showed that Fe and Al oxide form complexes with SOC and help sequester carbon in soils. Clay mineralogy, particularly with high amounts of kaolinite and oxides, increases the formation of micro-aggregates (Denef *et al.*, 2004), which protects soil carbon.

The impact of exotic trees on $\delta^{15}N$ and nutrients (Paper II)

The $\delta^{15}N$ values under farmland were higher than under native forest (Fig. 5). This is consistent with the findings of Rega *et al*. (1970 *cit*. Yoneyama, 1996). Lemenih, Karltun & Olsson (2005) also observed increasing $\delta^{15}N$ values in surface layers with increasing cultivation age in southern Ethiopia. In the present study, the depletion in ^{15}N was higher at the surface soil layers than in deeper soil layers (Fig. 5). In farmland, ^{15}N depleted litter input is low and cultivation mixes litter with deeper soil layers, which have higher $\delta^{15}N$, contributing to the increase in $\delta^{15}N$ of surface soil layers (Eshetu & Högberg, 2000). The increased mineralisation and loss of ^{15}N-depleted nitrate beyond the reach of crops could

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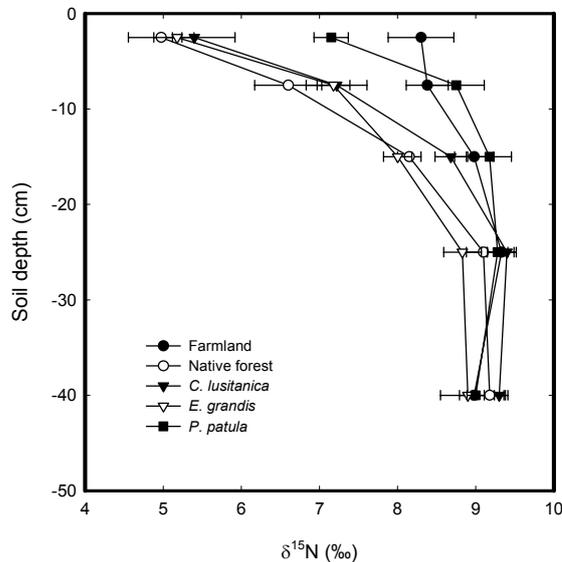also explain the ^{15}N enrichment in farmland (Högberg, 1997; Mekonnen, Buresh & Jama, 1997).

Fig. 5. Mean (± S.E.) of the δ^{15}N in the soil profile under different land uses and tree species (n=4).

Native forest clearing and subsequent cultivation led to a reduction in soil nutrients. There was a loss of 38% (6.8 Mg ha^{-1}) of the initial total soil N within the 0-50 cm soil layer over 75 years (Fig. 6). This loss on a linear basis was equivalent to a rate of 90.6 kg ha^{-1} y^{-1}. Total soil N losses of 52% (2.8 Mg ha^{-1}) after 30 years and of 61% (3.1 Mg ha^{-1}) after 25 years have been observed following deforestation and subsequent cultivation to 0-10 cm at other sites in Ethiopia (Solomon *et al.*, 2002). In our study the loss amounted to 60% (3.2 Mg ha^{-1}) for the same depth. In a chronosequence study, Lemenih, Karltun & Olsson (2005) showed that the loss in soil N in the 0-10 cm soil layer decreased to a equilibrium already in the first 15-20 years. Therefore, the loss in the surface layer at our site might have occurred long before the present. The decline in total soil N could be attributed to lower crop residue inputs to the soil (Paper I) and to the loss in crops. Accelerated mineralisation and nitrification rates on farmland and enhanced leaching losses of excess soil N from the topsoil contribute to the decline (Mekonnen, Buresh & Jama, 1997). The concentrations of exchangeable K$^+$ and Ca^{++} in the 0-10 cm soil layer under farmland and those of Mg^{++} and available P within the 0-20 cm layer were significantly lower than under the native forest (Paper II). Lemenih (2004) found lower concentration of exchangeable Ca^{++} in arable land than native forest while available P and K$^+$ were higher in the arable land. The loss of the most limiting plant nutrients such as N, available P and K has a great impact on the yield and productivity of subsistence farming in Ethiopia.

Significant reductions in soil nutrients were confined to the 0-20 cm soil layer, indicating that they were influenced by plough depth.
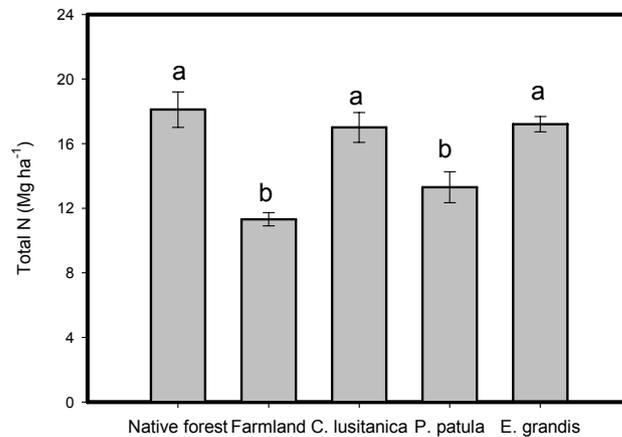

Fig. 6. Mean (± S.E.) of total N (Mg ha^{-1}) within 0-50 cm depth under different land uses and tree species (n=4). Different lowercase letters denote significant differences at *p*< 0.05

After afforestation with *C. lusitanica* and *P. patula*, the $\delta^{15}N$ value decreased in the surface soil layers under the trees (Fig. 5). The differences in fractionation of organic matter during microbial transformation and differential N sources might explain the differences in $\delta^{15}N$ distribution under the trees (Piccolo *et al.*, 1996). The $\delta^{15}N$ of *C. lusitanica* needle litter was 1.4‰, while that of *P. patula* was 3.0‰ (Paper II). The decrease in $\delta^{15}N$ was greater for the surface layer of *C. lusitanica* than *P. patula*, which was consistent with the deposition of more ^{15}N-depleted needle litter and faster mineralisation of *C. lusitanica* needles (Paper III). The $\delta^{15}N$ in soils under *E. grandis* and native forest did not differ significantly. The similarity in $\delta^{15}N$ of the two stands in the 10-30 cm soil layer might suggest that the several years of C_4 pasture input (Paper I) from the previous land use was responsible.

Soil N increased by 5.7 Mg ha^{-1} in the *C. lusitanica* stand and by 2.0 Mg ha^{-1} in the *P. patula* stand within the 0-50 cm soil layer (Fig. 6). Similarly, soil N has been found to increase in secondary succession on N-degraded farmland over 55 years (Inouye *et al.*, 1987). An increase in soil N of 2.5 Mg ha^{-1} was found under *C. lusitanica* in 15 years after reforestation of farmland in southern Ethiopia (Lemenih, Lemma & Demel, 2005). The amount of N gained or lost was closely related to C dynamics (Paper I), indicating that similar mechanisms apply to both. The increase in our study could be due to mechanisms behind uplift from the subsoil. Accumulation of nitrate-N has been found in the subsoil (50-200 cm soil layer) under unfertilised maize in Kenya (Buresh & Tian, 1998). This accumulation of subsoil nitrate has been attributed to the greater mineralisation of soil OM than uptake of N by unfertilised maize (Mekonnen, Buresh & Jama, 1997). The excess nitrate leaches to the subsoil and sorbs onto positively charged

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clay surfaces, retarding leaching losses (Hartemink *et al*., 1996). Fast-growing trees with high N demand, including *E. grandis*, can take up the N in the subsoil (Buresh & Tian, 1998). Other processes such as wet deposition might also have contributed to the N pool in the present study. A recent report indicates that the wet N deposition amounts to 2.05 kg ha^{-1} y^{-1} at Munesa Shashemene Forest in southern Ethiopia (Ashagrie, 2004).

Afforestation in our study did not significantly affect available P and exchangeable Mg^{++} in the soil layers. However, K$^+$ concentration increased significantly within the 0-30 cm soil layer in both *C. lusitanica* and *P. patula* stands, while exchangeable Ca^{++} increased significantly only in the 0-5 cm layer in the *C. lusitanica* stand (Paper II). The increase in K$^+$ was consistent with reports by Nowak, Downard & White (1991). Enhanced cycling of Ca^{++} and Mg^{++} has been observed under plantations of *E. grandis* (Tchienkoua & Zech, 2004). Different factors could have contributed to the significant increase in concentrations of some of the nutrients. The annual deposition (kg ha^{-1}) of nutrients by rainfall was found to be 0.08 for Mg, 0.18 for PO$_4$-P, 0.80 for K and 1.77 for Ca at Munesa Shashemene Forest in southern Ethiopia (Ashagrie, 2004). In that study, net throughfall of nutrients under *C. lusitanica* (kg ha^{-1}) was 0.44 for Mg, 3.22 for K and 0.36 for Ca. Potassium is released faster than other base cations from litter (Lisanework & Michelsen, 1994; Paper III). Decomposition is not a prerequisite for a release of K$^+$ since it is not highly associated with structural materials (O'Connell, 1988). The increase in exchangeable Ca^{++} under *C. lusitanica* in the 0-5 cm soil layer was consistent with the high concentration of Ca and relatively faster mineralisation rate of *C. lusitanica* needles (Paper III). Available P may limit the productivity and hence SOC sequestration in the second rotation of the plantations.

Assessing decomposition process in the forest floor and microcosm decomposition of leaf litter and fine roots

The relationship between litter chemistry and decomposition in Paper III was determined by comparing the chemistry of fresh litter with that of decomposed litter (from the Oe layer of the forest floor). Low concentrations of K, nearly the same or increased concentrations of Ca, Mg, P and S, and increased concentrations of N were found in decomposed litter compared to fresh leaf litter (Paper III). The results for K, Ca and Mg in decomposed litters are in line with those of other studies (O'Connell, 1988; Ribeiro, Madeira & Araujo, 2002), and are consistent with the higher mobility of K compared to Ca and Mg (Marschner, 1995). The Ca concentration was not affected by leaching in the early stage (Tian *et al*., 2003). Calcium is structurally bound and its release would be closely associated with microbial activity on related C fractions (Blair, 1988; Tian *et al*., 2003). The lower impact of afforested trees on the soil concentrations of exchangeable Ca^{++} and Mg^{++} than those of K$^+$ (Paper II) might be related to the slower release of these elements during decomposition. The concentrations of P and S were nearly the same or increased in decomposed litters. The N concentration was higher in decomposed litter compared to fresh leaf litter, in agreement with other studies,

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Table 1. *Organic fractions (mg g⁻¹) of leaf litter (LL) and decomposed litter (DL) of plantation and native forest*

Forest type	WSE	NPE	Holocellulose	Lignin	Ash	Arabinose	Xylose	Mannose	galactose	Glucose
C. lusitanica										
LL	149.4	57.4	385.3	408.0	76	36.8	22.7	31.4	36.8	158.0
DL	17.6	34.0	345.4	603.0	146	22.2	25.8	25.8	28.1	108.9
E. grandis										
LL	189.8	140.7	307.0	362.5	62	32.0	37.3	8.5	30.9	169.5
DL	23.5	52.2	265.0	659.3	234	22.2	36.6	18.3	28.7	107.0
P. patula										
LL	143.9	85.7	370.4	400.0	55	32.8	28.6	59.3	39.2	225.4
DL	23.5	23.3	135.3	825.8	357	26.4	37.3	35.8	31.1	124.4
Native forest										
LL	119.6	43.9	445.7	390.8	89	27.4	40.6	14.3	35.1	165.8
DL	21.6	20.2	250.7	707.5	258	21.6	29.6	16.2	27.0	93.0

* WSE- water soluble extractives, NPE- non polar (dichloromethane) extractives

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and may be attributed to microbial immobilisation (Ribeiro, Madeira & Araujo, 2002; Tian *et al*., 2003). The relative increase in N concentration in the decomposed litter together with the increase in soil N concentration in surface soil layers following plantation (Paper II) suggests that N release took place very early during the decomposition. However, the non-significant increase in available P concentration following afforestation might suggest either a slow release of P during decomposition of litter or early release followed by fixation on soil mineral surfaces into non-available form.

Concentrations of WSE, NPE, holocellulose, arabinose, galactose, xylose (in native forests and *Eucalyptus*) and mannose (in *Pinus* and *Cupressus*) were lower in the decomposed litter than in fresh leaf litter, whereas concentrations of lignin, xylose (in *Cupressus* and *Pinus*) and mannose (in native forest and *Eucalyptus*) were higher in the decomposed litter (Table 1). The decrease in WSE concentration in the decomposed litter is consistent with its early leaching and microbial utilisation (Berg & McClaugherty, 2003). The less pronounced decline in NPE concentration than in WSE in the decomposed litter may indicate a relatively slower leaching and degradability of NPE, which consists of fats, oils and waxes (Preston *et al*., 1997). The decline in holocellulose concentration and increase in lignin concentration in the decomposed litter was in agreement with other litterbag studies (Johansson, 1994; Tian, Takeda & Azuma, 2000). Microorganisms easily degrade isolated, unshielded holocellulose, which may explain the lower holocellulose concentration in the decomposed litter compared with the fresh leaf litter (Berg & McClaugherty, 2003). The increase in lignin concentration may be due to an increase in more recalcitrant compounds, and/or the loss of the easily degradable fractions. Increased xylose concentration in decomposed litter of *C. lusitanica* and *P. patula* may indicate a slower release of xylose, which agrees with slow decomposition of xylan in Scots pine needles (Berg & McClaugherty, 2003). The increase in mannose concentration in decomposed litter of native forest and *E. grandis* stands may suggest re-synthesis by fungi operating under the two stands (Tian *et al*., 2003).

The CPMAS ^{13}C NMR spectra showed differences between decomposed and fresh leaf litter (Fig. 7). The proportion of alkyl, methoxyl, aromatic, phenolic and carbonyl C increased and that of *O*-alkyl C declined in decomposed litter compared to fresh litter except in the *E. grandis* stand, which showed a decline in alkyl and nearly the same *O*-alkyl C (Paper III). The increase in alkyl and decrease in *O*-alkyl C intensity was consistent with other studies (Quideau *et al*., 2005). The decline in *O*-alkyl C was probably due to the preferential decomposition of holocellulose, as indicated by a decline in the proximate holocellulose fractions. The increase in alkyl C was due to accumulation of recalcitrant leaf cutins and waxes and also a contribution from microbial growth (Quideau *et al*., 2005), while there was a decline in extractable alkyl C (NPE). On the other hand, the decrease in alkyl C in *E. grandis* decomposed litter (Fig. 7) was also consistent with the decrease in alkyl C during composting of some tree leaf litters (Almendros *et al*., 2000). The intensity of *O*-alkyl C in *E. grandis* spectra was nearly the same in fresh and decomposed litter, which is consistent with a study by Quideau *et al*. (2005) on decomposition of oak leaf litter during 1.5 years. Non-polysaccharide

O-alkyl C may have accumulated, since proximate holocellulose concentration was reduced in decomposed litters. Consistent with findings by Inbar, Chen & Hadar (1989), there was an increase in aromatic and phenolic C regions, which was not common in many studies. This result along with increase in methoxyl C intensity might suggest a relative increase in aromatic structures. The increase in proximate lignin fraction may be associated with an increase in cutin, which contributes mainly to alkyl C; tannin to phenolic C and lignin structures to aromatic C and phenolic C of ^{13}C NMR spectra with the extent of decomposition (Preston *et al*., 1997).
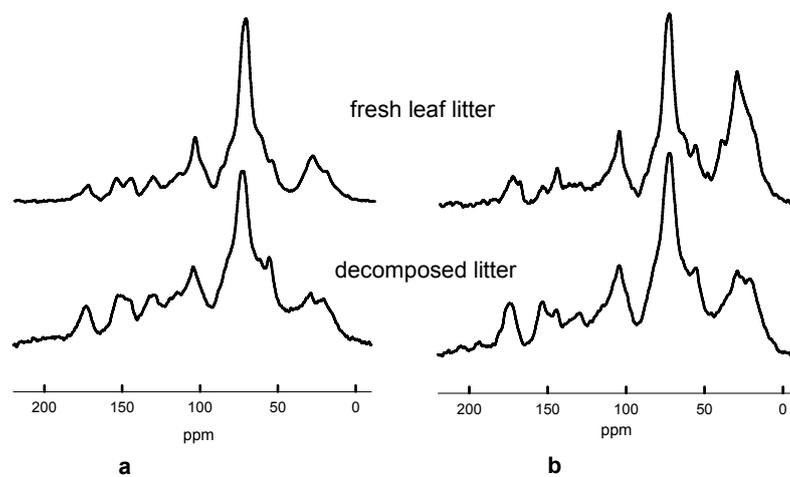
Fig. 7. CPMAS ^{13}C NMR spectra of fresh leaf litter and decompose litter of (**a**) *P. patula* and (**b**) *E. grandis*

Leaf litter and fine roots of the exotic trees (*C. lusitanica*, *P. patula* and *E. grandis*) and native forests differed in their chemical composition. The leaf litter in general had higher concentrations of Ca and solubles, and lower concentrations of lignin (except *P. patula*) than fine roots (Paper III). These findings are consistent with results from other tropical tree species (Bloomfield, Vogt & Vogt, 1993). The ^{13}C NMR spectra also showed higher proportions of alkyl C regions and lower *O*-alkyl C, aromatic C and phenolic C regions in the leaf litter than in fine roots (Paper III). Findings by Lessa, Andersson & Chatson (1996) agree with the higher proportions of alkyl C regions and lower O-alkyl C regions in leaf litter than fine roots, but not with the lower proportions of aromatic C and phenolic C regions than in fine roots.

The mean cumulative C mineralised from leaf litter (mg C g^{-1} original C) was 308.5 for *C. lusitanica*, 250.2 for native forest, 165.2 for *E. grandis* and 155.7 for *P. patula* during 120 days of laboratory decomposition, whereas the mean cumulative C mineralised from fine roots was 210.8 for *P. patula*, 161.1 for native forest, 128.4 for *C. lusitanica* and 126.5 for *E. grandis* (Paper III). The pattern of C mineralisation rates for leaf litter and fine roots (Fig. 8), which were initially high and declined with the progress of time, is explained by the relative increase in

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more recalcitrant compounds and decreasing degradability by microorganisms (Yang *et al*., 2004). The amount of cumulative C mineralised from leaf litter and the associated rate constants (k) were positively correlated with Ca concentration. Increasing Ca concentration in a growth substrate has a positive effect on the growth of fungi (Lindeberg 1944 *cit*. Johansson, 1994). Moreover, substrate pH influences fungal growth and Lindeberg suggested that Ca influences the pH in a favourable way for fungal growth. Thus Ca in leaf litter might possibly enhance decomposition by these mechanisms. Arabinose and galactose concentrations were positively correlated with the cumulative amount of mineralised C, whereas WSE concentrations were positively correlated with the k value (Paper III). The correlation with cumulative mineralised C is consistent with early microbial degradation of arabinan and galactans (Berg & McClaugherty, 2003). The correlation of k with WSE might indicate that major differences in the decomposition rate were due to the first decomposition phase, which may be controlled by WSE (Fig. 8b). The similarity of the decomposition rates of fine roots during the latter days may indicate that the recalcitrant material decomposes at fairly similar rates.
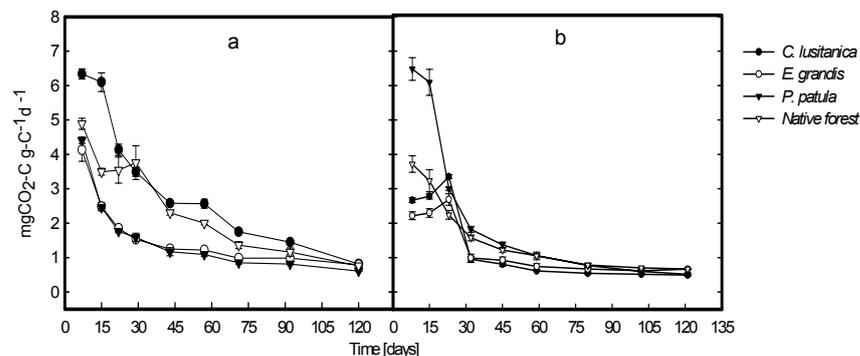

Fig. 8. Rate of C mineralisation (mg CO_2 g C^{-1} d^{-1}) of (a) leaf litter and (b) fine roots.

Cumulative mineralised C was lower for fine roots than for leaf litter for *C. lusitanica*, *E. grandis* and native forest, whereas it was higher for *P. patula* (Paper III). In agreement for *P. patula*, in some previous studies fine roots were shown to decompose faster than leaf litter (Hobbie, 1996; Ostertag & Hobbie, 1999), while the finding that *C. lusitanica*, *E. grandis* and native forest fine roots decomposed slower than leaf litter agrees with results for other tree species (Bloomfield, Vogt & Vogt, 1993; Usman *et al*., 2000). This shows that there are tree-specific differences in leaf litter and fine root decomposability.

Assessing factors that contribute to differences in soil C under exotic trees (Paper IV)

In order to assess factors that contribute to the differences in tree species impact on SOC stock, SOC was modelled under *C. lusitanica*, *P. patula* and *E. grandis*, which are grown under similar management, edaphic and climatic conditions. Factors considered included litter input, litter quality and microclimate. After

parameterisation, the biomass module of the CO2Fix model using turnover parameters estimated litter input into the soil module (Paper IV). The modelled leaf litter reasonably approximated the measured litterfall (mainly leaf litter) at the site at the age of 20 except for *E. grandis*. This difference in *E. grandis* was related to the attainment of current annual increment early in the tree growth. The modelled root litter input values also agreed with the fine root (<2 mm) biomass at the site at age 20, which could be assumed to be the major source of root litter. Moreover, the measured litterfall was in the range for tropical mountain forests (Vitousek, 1984), and Indian tropical forests (Pragasan & Parthasarathy, 2005). The fine root (<2mm) biomass measured was also within the range for montane forest (Cavelier, 1992).
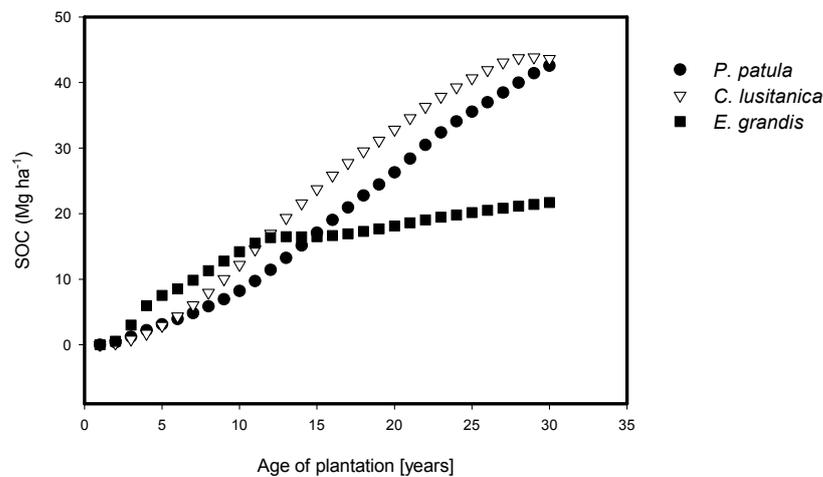
Fig. 9. Simulated SOC sequestration under tree plantations with age.

The simulated SOC at age 20 was 32.8 Mg ha^{-1} for *C. lusitanica*; 26.3 Mg ha^{-1} for *P. patula*; and 18.1 Mg ha^{-1} for *E. grandis* (Fig. 9). In agreement with the measured SOC (Paper I), the simulated SOC under *C. lusitanica* was greater than under *P. patula* and *E. grandis*. When 20 years of simulated litter input from trees to the soil during the development of the trees (Fig. 10) were compared, *C. lusitanica* had higher total litter input than *P. patula* and *E. grandis* and moreover higher fine wood litter than both other tree species. The relatively greater importance of fine woody litter than non-woody litter was associated with a slower rate of microbial attack, *i.e.* slower decomposition (Hyvönen & Ågren, 2001). The simulated total litter (Fig. 10) was nearly equal for both *P. patula* and *E. grandis*, but *P. patula* had more fine woody litter than *E. grandis*, which accounted for their differences. Chemical litter quality indices used in the model explained 1.2 to 1.4% of the SOC differences under *C. lusitanica* and other trees, whereas it explained about 0.3% of the differences in the accreted SOC between *P. patula* and *E. grandis* (Paper IV). Microclimatic differences were accounted for by making an approximate temperature reduction under *C. lusitanica* (Paper IV), which had higher LAI and lower under-canopy temperature, based on literature

values (Lemenih, Gidyelew & Teketay, 2004). Results suggested that microclimates had little impact on the differences in SOC between tree species.
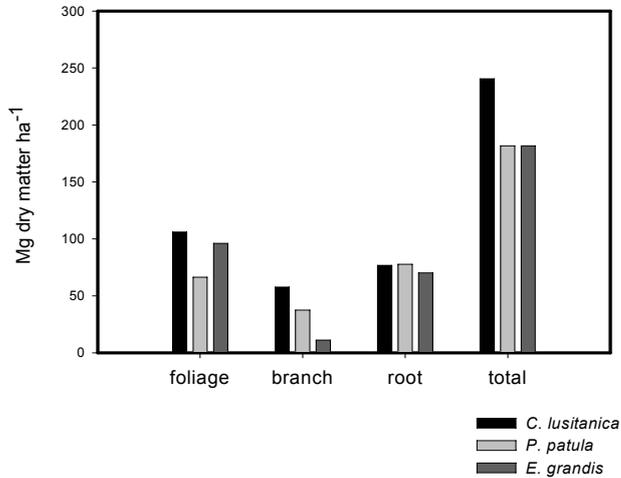

Fig. 10. Simulated litter input (dry matter) under tree species within the 20-year period.

Conclusions

Forest plantations on degraded farmland can influence SOC and nutrient concentrations, but studies of these impacts are few in the tropics and are inconsistent. The results are controlled by many factors, for instance soil type, soil texture, management history, site history, the extent to which labile SOC pools were depleted in farmland, and litter quality and quantity of the plantations established. This thesis evaluated the conversion of native forest to smallholder cropping land and, more particularly, the impact of exotic tree plantations on soil carbon and nutrient status of former cropped land, and attempted to assess some tree-related factors that contributed to the changes under plantations in the highlands of southwestern Ethiopia. Based on the specific location and the systems investigated, *i.e.* farmland, native forest, and three stands consisting of exotic trees species, the following conclusions were drawn:

- The conversion of native forest into farmland results in a reduction in the original SOC content, but this is partly compensated for by carbon originating from litter and harvest residues from farm crops. This suggests that the variation in net SOC losses reported in different studies may depend on *e.g.* farm residue management. Farm management that allows organic matter to return to the soil may thus lead to lower C losses and lower CO_2 emissions. The decline in SOC content in this study was accompanied by a decline in total nitrogen, available phosphorous and base cations in farmland. Loss of soil nutrients and hence low and

decreasing productivity are typical features of smallholdings in Ethiopia. Farmers are primarily concerned about the crop production for the forthcoming season only, because of their immediate financial and nutritional needs. The sustainability of the farming system from the long-term fertility viewpoint is at stake and has to be addressed.

- Fast-growing exotic trees (*Cupressus lusitanica*, *P. patula* and *E. grandis*) established on abandoned farmland lead to SOC accretion at a rate dependent on tree species. *Cupressus lusitanica* showed a higher potential for soil carbon sequestration than *P. patula* and *E. grandis*. The differences in carbon sequestration at sites with similar management, edaphic and climatic conditions mainly relate to litter inputs and decomposition. Modelling confirmed that higher litter inputs with a relatively high proportion of fine woody material take place under *C. lusitanica* than under *P. patula* and *E. grandis*. Studies on decomposition showed that *C. lusitanica* leaf litter decomposes faster than leaf litter from native forest and other tree plantations, whereas its fine roots decompose slower than those of *P. patula* and native forests and similarly to those of *E. grandis*. Fast decomposition of leaf litter may imply a faster transfer of fresh carbon to mineral soil, which contributes to the higher SOC under *C. lusitanica*. The slower decomposition of *C. lusitanica* fine roots may also contribute to higher SOC sequestration because fine root litter is directly placed into mineral soil. In contrast, *P. patula* with its relatively lower litter input, slow needle decomposition and faster fine root decomposition, shows a relatively lower SOC sequestration. Tree species differ in their capacity to sequester soil carbon; hence, knowledge of the tree species producing the litter, litter fall rates and fine root litter decomposition and root turnover is crucial. The carbon sequestration pattern under all tree species may be influenced by the clay content of the soil and the presence of kaolinites, iron oxides and other iron-rich amorphous minerals. Iron oxide and iron-rich amorphous minerals stabilise soil organic carbon through organo-mineral complexes, while 1:1 clays (kaolinite) and oxides favour micro-aggregate formation, which provides physical protection for soil carbon.

- Since reforestation and afforestation are being considered as means to mitigate CO_2 emissions, it is encouraging that a shift from abandoned farmland to fast-growing plantations of exotic trees with a higher SOC storage can be achieved within two decades. These findings could serve as a basis for sequestration projects through afforestation and reforestation and may encourage sub-Saharan countries with similar environmental conditions to participate in the management of the global carbon cycle through Clean Development Mechanism (Article 12) of the Kyoto Protocol, although sub-Saharan countries *per se* make a negligible contribution to total global CO_2 emissions. However, large-scale plantations may interfere with land tenure, land use laws, agricultural policies affecting forest resources, and cultural characteristics in these countries, and integrated land use policy and planning are required.

Planting trees for C sequestration may work best if coordinated with fuel wood and construction material demands or in a way to yield economic returns.

- In addition to soil carbon sequestration, afforestation with fast-growing exotic trees affects some soil nutrients. An increase in soil N is closely associated with the increase in SOC, showing that similar mechanisms apply to both. The natural ^{15}N technique indicates that exotic trees restore a 'native forest like' nitrogen cycling process. In addition, afforestation may affect the replenishment of some soil nutrients while not affecting others. The overall effect on the nutrient status may depend on amount of nutrients to be recycled, the recycling efficiency of the tree, litter input, litter decomposition and the release of nutrients. In general, available P may be the most limiting nutrient in the study sites, and may constrain the productivity of the following rotation.

- The trees species differ in total litter input and fine woody litter that mainly explained SOC content of tree species as demonstrated using a modelling approach. The decomposition rate of plant litter is an important component in forest ecosystems. Leaf litter decomposes at a different rate than the corresponding fine roots, as the decomposition rate of leaf litter and fine roots is controlled by different substrate quality factors. At the study site, Ca may play an important role in regulating the decomposition of leaf litter through its influence on microorganisms or it may be an indicator of the size of some unknown real quality variable. The main transformations that occur during litter decomposition in general are a reduction in holocellulose, water-soluble extractives and non-polar extractives and enrichment of recalcitrant compounds with alkyl structures (cutins), compounds with aromatics structures like tannins and lignin aromatic structures.

Final remarks

Botanical classification into coniferous vs. broadleaved trees has often been used to explain the influence of tree species on soil or litter decomposition rates. However, this study shows that great variation exists between the two conifers *C. lusitanica* and *P. patula* in their effect on soil, and in their needle litter and fine root decomposition.

Further research is needed to address the effect of these fast-growing trees on soil fauna and microbial biomass and diversity, mycorrhizal association, and fine root turnover rates. In addition, tree species influence may differ with different site properties, agro-climatic zones, management and plant age. Therefore, additional studies on carbon and nutrient dynamics following reforestation and afforestation combined with long-term decomposition studies of litter from leaves and fine roots, plus monitoring of litter fall inputs and fine root biomass, would enhance our understanding of the influence of these tree species on soil.

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