Effects of Land Use Changes on Soil Quality and Native Flora Degradation and Restoration in the Highlands of Ethiopia

Implications for sustainable land management

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Abstract

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Land degradation is threatening biological resources and agricultural productivity, the mainstay of the economy in Ethiopia. Ensuring sustainable food and biomass supply while maintaining ecological integrity in Ethiopia requires two imperative efforts: (i) the sustainable use of productive land resources, and (ii) effective regeneration of degraded ecosystems. This thesis aims to (i) identify trends in soil quality and native flora degradation due to deforestation and subsequent cultivation using a chronosequence of farm fields converted from a tropical dry Afromontane forest; and (ii) investigate the possibilities for restoration of soil quality and native flora on degraded sites with the help of reforestation. The studies were conducted near and in the Munessa-Shashamane forest, which is located on the eastern escarpment of the Central Ethiopian Rift Valley.

The results showed that following deforestation and subsequent cultivation, soil organic carbon (SOC) and total N declined exponentially in the 0-10 cm layer of the soil. In the same soil layer, analysis based on ^{13}C natural abundance revealed that SOC of forest origin was declining by 740 kg C ha^{-1} yr^{-1}, while addition to the SOC from agricultural crops was about 240 kg C ha^{-1} yr^{-1}. The imbalance in SOC addition from the crops and loss of SOC of forest origin has led to the continuous decline of SOC in the bulk soil by 500 kg C ha^{-1} yr^{-1}. The loss of N from the surface soil of the farm fields was 66 kg N ha^{-1} yr^{-1} as compared with the fertilizer application rate of 35 kg N ha^{-1} yr^{-1} in the area, which, however, is seldom applied due to economic constraints for the farmers. Soil bulk density increased and pore space decreased progressively in the 0-10 and 10-20 cm soil layers with increasing cultivation period after deforestation. Other soil proprieties such as available P and K, exchangeable K, Ca and Mg, BS and CEC also changed significantly but at a slow rate. Most of the significant changes were limited to the top 0-10 cm layer. At the present level of management, the soils of the study area can be used for 25-30 years without loss of productivity. This 25-30 year period of sustainable use is much longer than most reports for tropical soils subject to similar land use changes, and this was attributed to the volcanic nature of the soils and the traditional low intensity tillage practice coupled with the parkland agroforestry used in the farming systems investigated. It was also observed that as tillage intensity shifts from the traditional low tillage to high intensity mechanized tillage the rate of soil degradation increases, which may reduce the period of sustainable use of the deforested sites.

Deforestation and subsequent cultivation of the tropical dry Afromontane forest investigated also endangered the native forest biodiversity, not only through the outright loss of habitat but also by deteriorating the soil seed banks. The results showed that the contribution of woody species to the soil seed flora declined from 5.7% after 7 years to nil after 53 years of continuous cultivation. However, soil quality and native flora degradation are reversible through reforestation. Reforestation of abandoned farm fields with fast-growing tree species was shown to restore soil quality. Tree plantations established on degraded sites also fostered the recolonization of diverse native forest flora under their canopies. An important result from studying the effects of reforestation is that good silviculture, particularly selection of appropriate tree species, can significantly affect the rate and magnitude of both soil quality and biodiversity restoration processes.

Key words: Andosols, biodiversity, deforestation, ecological restoration, land degradation, Munessa-Shashamane, reforestation, regeneration, soil seed bank, ^{13}C, ^{15}N, soil organic matter, sustainability, subsequent cultivation.

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Contents

Preface

Articles I-V

This thesis is based on the following papers, which are referred to in the text by their Roman numerals:

I. Mulugeta Lemenih, Karltun, E. and Olsson, M. 2004. Assessing soil chemical and physical property responses to deforestation and subsequent cultivation in smallholders farming system in Ethiopia. *Agriculture, Ecosystem & Environment* (In press).

II. Mulugeta Lemenih, Karltun, E. and Olsson, M. 2004. Soil organic matter dynamics after deforestation along a farmland chronosequence in southern highlands of Ethiopia. *Agriculture, Ecosystem & Environment* (Submitted).

III. Mulugeta Lemenih and Demel Teketay. Changes in soil seed bank composition and density following deforestation and subsequent cultivation of dry Afromontane forest in Ethiopia. (Manuscript).

IV. Mulugeta Lemenih, Olsson, M. and Karltun, E. 2004. Comparison of soil attributes under *Cupressus lusitanica* and *Eucalyptus saligna* established on abandoned farmland with continuously cropped farmlands and natural forest in Ethiopia. *Forest Ecology and Management* (In press).

V. Mulugeta Lemenih, Taye Gidyelew and Demel Teketay, 2004. Effects of canopy cover and understory environment of tree plantations on richness, density and size of colonizing woody species in southern Ethiopia. *Forest Ecology and Management* (In press).

1. Background

1.1. Introduction

Ethiopia with an area of 1,130,000 km^2 and about 67 million population is the third largest and most populous country in Africa (MoFED, 2002; Kidanu, 2004). The country's major natural renewable resources consist of land, water and natural vegetation that comprise enormous biodiversity (African Development Bank, 1997). In Ethiopia, as in most developing countries, the economy is primarily based on agricultural production. Agriculture accounts for 52% of the GDP (World Bank, 2002), 90% of the total export revenue (IMF, 2002), and employs about 85% of the labour force in the country (CSA, 1999). The agriculture in Ethiopia is predominantly subsistent in nature. Smallholder farmers with an average holding of less than one hectare account for over 90% of the agricultural area under crop production (Tsegaye, 1997), and 95% of the agricultural outputs (Legesse, 2003). The agricultural production system is mainly rainfed and traditional, which is characterized by low input of fertilizer and pesticides.

Moreover, long-term overall economic development policy in Ethiopia is planned as “Agricultural Development-Led Industrialization (ADLI)” (NCSS, 1993). The goal of this strategy is to achieve rapid and sustainable economic growth by improving the productivity of the agricultural sector. In recent years, various official documents of the government of Ethiopia (e.g. ‘Sustainable Development and Poverty Reduction Program - also called PRSP’ (MoFED, 2002) and ‘Rural Development Strategy’ (MoA, 2002)) all reiterate that agriculture is the driver of economic development and the key sector for reducing poverty and ensuring food security in the country. This strong reliance on agriculture as an economic driving force entails that natural resources of agricultural significance should be managed on a sustainable basis. It can only be sustainable management of the agricultural resources base that will enable the country to achieve the desired sustainable rural and economic development goals on the basis of its agricultural economy.

1.2. Deforestation, land degradation and the challenges of sustainable agricultural development in Ethiopia

It is obvious that the agricultural sector in Ethiopia is increasingly being confronted with the pressure from a rapidly growing population and diminishing natural resources; the problems that engender biophysical land degradation and hamper sustainable agricultural development in the country (EFAP, 1994; Bojo and Cassels, 1995; Herweg and Stillhardt, 1999). Land degradation is the process of progressive deterioration of biological (flora and fauna) and physical (soil, water, micro-climate, etc.) resources of the land, leading to declining productivity and unsustainable yields (Singh, 1995). The lag in agricultural productivity advancement behind population growth has caused intense land use conflicts, particularly between the agricultural and the forestry sectors in Ethiopia. To compensate for the low agricultural productivity, deforestation for arable land expansion has been the principal land use change employed in Ethiopia for centuries. Rate of deforestation in Ethiopia, which amounts to 163,000 - 200,000 ha yr^{-1}, is one of the highest in tropical Africa (Reusing, 1998). As a result, the natural forest cover in Ethiopia has declined considerably from approximately 40% to just less than 3% (Kuru, 1990; EFAP, 1994); a process that has further exacerbated land degradation in the country.

The traditional shifting cultivation practice has changed due to demographic and economic pressures, leading to permanent agriculture being practiced on deforested sites. There are several repercussions of such land use changes and intensification, the most important in Ethiopia's context being: (i) accelerated soil erosion and deterioration of soil nutrient status (FAO, 1986; Hurni, 1988, 1993; EFAP, 1994; Tekle, 1999); (ii) altered hydrological regimes and sedimentation of wetlands (Kuru, 1990; EFAP, 1994; Hawando, 1997); and (iii) loss of primary tropical forests and their biodiversity (NCSS, 1993; EFAP, 1994; Teketay, 1996; Reusing, 1998; Tekle, 1998).

Deforestation and conversion to permanent cultivation is the primary cause for dwindling tropical biodiversity and in Ethiopia the practice has already threatened a number of plant species, including the gene pool of wild populations of *Coffee arabica* L. (Tewolde, 1989, 1990; Kelbessa *et al*., 1992; Tadesse *et al*., 2001, 2002). Moreover, deforestation coupled with improper crop production practice on the mountainous topography that dominates the highlands of Ethiopia is considered to be the root cause of the excessive soil erosion in the country (Hurni, 1993; Janssen and Willkens, 1994; Tekle, 1999). An estimated 1.9 billion tons of soil, on average 42 tons ha^{-1}, are eroded annually from the highlands of Ethiopia (SCRP, 1984 – 1991; EFAP, 1994). This removal of surface soil by erosion results in soil organic matter (SOM) loss in the range of 1.17 to 78 million tons yr^{-1} (15 to 1000 kg ha^{-1} yr^{-1}), soil nitrogen loss from 0.39 to 5.07 million tons yr^{-1} and that of phosphorus from 1.17 to 11.7 million tons yr^{-1} (Hawando, 1997). Available reports (e.g. EFAP, 1994; Hawando, 1997) indicate that over 50% of the agricultural land in the highlands of Ethiopia is already severely affected by soil erosion.

Land degradation in Ethiopia is also exacerbated by soil nutrient depletion arising from continuous cropping together with removal of crop residues, low external inputs and absence of adequate soil nutrient saving and recycling technologies (Bojo and Cassels, 1995; Sahlemedhin, 1999). A continental study commissioned by the FAO in 38 sub-Saharan Africa (SSA) countries, including Ethiopia, showed that Ethiopia is one of the countries with the highest rates of nutrient depletion. The aggregated national scale nutrient imbalances were -41 kg ha^{-1} yr^{-1} for N, -6 kg for P and -26 kg for K (Stoorvogel and Smaling, 1990). It has long been known that SOM and other soil properties decline rapidly following tropical forest clearance, burning and subsequent cultivation (e.g. Nye and Greenland, 1960; Lal, 1976). However, the rates and magnitudes of the declines are highly variable depending on several factors such as soil type, climatic factors and land use intensity (e.g. Tiessen *et al*., 1994; Tinker *et al*., 1994; Parfitt *et al*., 1997). Nevertheless, in Ethiopia, one of the tropical countries with considerable deforestation experience (Kuru, 1990), few scientific studies have been made on soil quality and biodiversity changes following deforestation and land use intensification.

1.3. Ecological restoration to harness sustainable development in Ethiopia

In a country like Ethiopia where a rapidly growing human population is inducing overexploitation of the available productive natural resources, restoration of the vast degraded landscapes that exist in the country will have a valid and important role in harnessing sustainable development. According to Tekle (1998) reversal of land degradation and restoration of the productive capacity of the degraded land is a necessity and not an option in Ethiopia, especially if most of the livelihood and economic development are to continue to emerge from the agricultural economy.

In fact, restoration/rehabilitation of degraded lands is a subject that is receiving considerable attention in many parts of the world (Montagnini, 2001; Perrow and Davy, 2002; Bradshaw, 2002), especially in SSA (Chamshama and Nduwayezu, 2003). Underlying reasons for global interest in restoration include: (i) dwindling forest cover and forest products; (ii) environmental problems such as climate change, loss of biodiversity, pollution, desertification, etc. associated with natural forest cover reduction and conversion to intensive land use; (iii) decreasing land productivity owing to large areas of potentially productive lands languishing in a highly degraded state due to soil and water erosion, declining soil fertility and loss of soil organic matter; (iv) decreased infiltration and water retention capacity, increased runoff and disrupted hydrological cycles (floods and water shortages); and (v) increased sediment transport and water pollution (e.g. Rocheleau *et al.*, 1988; Lundgren and Taylor, 1993; Rowe *et al.*, 1994; Ayoub, 1998; Salami, 1998; Cairns, 2002; Chamshama and Nduwayezu, 2003).

There are diverse approaches and techniques to land and vegetation restoration (Brown and Lugo, 1994; Perrow and Davy, 2002). Past efforts to restore degraded agricultural lands in Ethiopia were predominantly characterized by an engineering approach or import and distribution of chemical fertilizers. Unfortunately, neither engineering solutions nor imports of chemical fertilizers proved satisfactory in solving the problem of land degradation in Ethiopia (Eyasu, 2002). For many reasons the engineering solutions were neither effective nor sustainable in Ethiopia (EFAP, 1994; Eyasu, 2002; Bekele, 2003), just as in many other African countries (Jaiyeoba, 2001). Similarly, large-scale adoption of chemical fertilizers to enhance crop productivity was not possible mainly due to economic constraints to the smallholder farmers.

The rationale for ecological restoration in Ethiopia must be broader to encompass other areas of rural development besides soil erosion or soil fertility. At present, the country is increasingly facing an acute shortage of forest product supply, alarming biodiversity crises and deteriorating socio-economic status in rural areas. A modest estimate for the current national scale wood deficit, mainly fuelwood, is over 33 million m^3 per annum; the very problem that triggered the turning of crop residues and cow dung to fuelwood substitutes (EFAP, 1994; Tekle, 1999). According to the FAO (1993a), at the current rates of utilization, demand for fuelwood will outstrip current vegetation resources in Ethiopia by the year 2010. Therefore, a sustainable supply of wood products is equally crucial to food supply and both must be considered components of sustainable economic and rural development in Ethiopia.

In order to address the problems of soil degradation, biomass scarcity and loss of biodiversity, reforestation/afforestation of degraded lands is often seen as the most sound rehabilitation technique in the tropics (e.g. Parrotta *et al*., 1997), and particularly in Africa including Ethiopia (Chatterson *et al*., 1989; Jaiyeoba, 2001; Lemenih and Teketay, 2004). There is now ample empirical evidence from wide geographical areas that substantiates the potential of reforestation or afforestation in restoration of the biophysical resources of degraded tropical lands (e.g. Lugo, 1997; Parrotta *et al*., 1997; Lamb, 1998; Harrington, 1999; Cannell, 1999), while providing diverse socio-economic and ecological services including wood supply (e.g. Lamb, 1998; Montagnini, 2001; Otsamo, 2000).

Reforestation activities have over a century of history in Ethiopia but despite this relatively long history**,** the total area of plantations in Ethiopia does not exceed more than 200,000 ha (Bekele, 2003). This is only equivalent to the area of natural forest deforested in a single year in Ethiopia. Furthermore, past reforestation activities have focused mainly on wood supply, particularly fuelwood, and little effort has been made to connect the reforestation programme to ecological restoration.

1.4. The rationale for the study and the research problem

The keys to sustainable economic development that depend on agriculture in Ethiopia are (i) the development, use and management of essential agricultural resources such as land, soil, water and forest on a sustainable basis; and (ii) successful restoration of the vast degraded landscapes in the country. However, despite the general recognition of the problem of land degradation and its impact on agricultural productivity at aggregate national scale, few scientific studies have been conducted at a small spatial scale such as the farm level to provide precise quantitative information on the extent of soil degradation problems in Ethiopia (Eyasu, 2002; Bekele, 2003). Due to the limited scientific studies at a small spatial scale, the accuracies and scales of the available aggregate soil degradation statistics in Ethiopia have been questioned (Eyasu *et al*., 1998; Eyasu and Soones, 1999; Eyasu, 2002). Moreover, the measurements of land degradation in Ethiopia have focused on soil erosion, and this has been treated in isolation from other aspects of soil management such as ensuring adequate fertility, SOM, vegetation cover and biodiversity (Eyasu, 2002).

Although erosion is one of the most production-limiting factors in the highlands of Ethiopia, a single factor-based generalization of land productivity constraints may be misleading for several reasons. Firstly, the magnitude of soil erosion differs significantly from site to site within the range 0-300 metric tons ha^{-1} yr^{-1} depending on the climatic conditions, soil type, land use/land cover, farming system, etc. (SCRP, 1984-1991; Omiti *et al.*, 1999). Secondly, there is currently increasing awareness that soil nutrient depletion from the agroecosystem is a very widespread problem and that it is an immediate crop production constraint in Ethiopia (Stoorvogel and Smaling, 1990; Stoorvogel *et al*., 1993; Bojo and Cassels, 1995; Eyasu, 2002). In fact, physical land degradation such as soil erosion often emanates from overexploitation of the soil and inappropriate cropping systems. Therefore, physical degradation such as erosion is a secondary feature emerging primarily from the loss of SOM, plant nutrients and vegetation cover (Eyasu, 2002). Yet the problem of soil nutrient loss has not been well linked to land degradation in Ethiopia to the same degree as drought and soil erosion (Eyasu, 2002). This is probably because soil fertility decline is a gradual process (Stoorvogel *et al*., 1993). Thirdly, proper understanding of soil quality degradation, for instance, as caused by continuous cropping at farm/field level, is generally limited in Ethiopia and the aggregate scale reports provide little practical guidance to design local conditions-dependent sustainable soil management technologies. To meet the increasing demands for food and to sustain environmental integrity in Ethiopia, soil quality must be maintained. Therefore, studies are needed for understanding and predicting the trends, magnitudes and rates of soil quality changes for the purposes of (i) monitoring effects of current land use changes and intensification; and (ii) designing appropriate management options that maintain soil quality for sustainable agricultural productivity in Ethiopia**.**

Another important area of research to ensure sustainable development in agriculture-dominated landscapes like that in Ethiopia is to assess the relationship between the practice of agriculture and other ecosystem components, particularly biodiversity. The term ‘biodiversity’ is usually used by conservation biologists to refer to the number of native plant species in a given system (species richness) (Simberloff, 1999). Technically, it refers to the variability in genetic structure, species composition (flora and fauna) and/or habitat properties of an ecosystem (Kellomäki *et al*., 2001). In this thesis, ‘biodiversity’ is used to refer to plant species richness in an ecosystem. The most obvious and major losses in biodiversity in Ethiopia occur through the destruction of habitat owing to the extensive deforestation and conversion of forests to agricultural land (Kelbessa *et al*., 1992; Tadesse *et al*., 2001, 2002). However, biodiversity loss is also taking place in a more subtle way, manifesting itself through the effects of progressive fragmentation and subsequent isolation of forest communities and depletion of resources needed for secondary forest succession such as soil seed banks, soil fertility and other habitat qualities following deforestation (Teketay, 1996; Kumar, 1999; Lemenih and Teketay, 2004). Habitat loss is usually accompanied by habitat degradation and fragmentation, which together accelerate biodiversity losses (Kumar, 1999). Nonetheless, the effect of deforestation followed by subsequent cultivation of different intensity on soil seed banks and the future of the forest flora are little documented in Ethiopia. Such knowledge on soil seed bank dynamics is particularly crucial in ensuring the conservation of biological resources in human-influenced ecosystems and also for planning successful restoration mechanisms for degraded land in the future.

Furthermore, in Ethiopia biophysical resources essential for rural and agricultural development have already been severely degraded from vast areas of land. Sustainable rural and agricultural development must reverse this trend and rebuild and augment the productive capacity of the diminishing agricultural resources base. The consideration of the socio-economic and ecological constraints of the rural areas in Ethiopia dictates that approaches to combat land degradation must emphasize strategies that will make sustainable (productive) use of the degraded lands while restoring soil fertility (Tekle, 1998). Reforestation/afforestation approaches for ecological restoration, which are receiving considerable attention in recent years, have been suggested as potential strategies for restoring degraded tropical lands and their biodiversity (e.g. Lugo, 1997; Parrotta *et al*., 1997; Otsamo, 2000; Montagnini, 2001) including Ethiopia (Lemenih and Teketay, 2004). However, scientific studies that evaluate the potential of plantation forests in ecological restoration in Ethiopia are limited. Poor capacity, institutional instability and ignorance of the forestry sector (Teklu, 2003) hamper scientific studies on the potential of reforestation/afforestation in ecological rehabilitation in Ethiopia.

1.5. Objectives of the study

The objectives of the present study were to:

1. Assess the rate and magnitude of changes in soil chemical and physical properties following deforestation and conversion into farmland of a tropical dry Afromontane natural forest, and to use these changes as indicators to assess the sustainability of the farming system (Paper I);
2. Investigate soil organic matter (SOM) dynamics based on natural ^{13}C and ^{15}N abundances on farmlands converted from a tropical dry Afromontane forest (Paper II);
3. Describe the impact on native flora of deforestation and subsequent cultivation of a tropical dry Afromontane forest through investigating changes in the species composition and density of soil seed banks (Paper III);
4. Examine soil attributes of fast-growing exotic plantations (*Cupressus lusitanica* (Mill.) and *Eucalyptus saligna* (Sm.)) established on abandoned farmland to evaluate the potential of plantation forestry for restoration of degraded soils (Paper IV); and
5. Investigate the role of plantation forests in fostering the recolonization of native flora, and how the canopy characteristics of the plantation species affect the process of native woody species recolonization (Paper V).

2. Theoretical Perspective

2.1. Sustainable land management and the tropical smallholder

Approach to the management and use of land resource is changing rapidly and dramatically towards sustainability at the global scale. The concept of sustainable management emerged essentially from the growing understanding of (i) limits to the availability and carrying capacity of land resources; and (ii) environmental, economic and social impacts of their improper uses (Dumanski *et al*., 1991; FAO, 1993b; Bell and Morse, 1999).

Concerning sustainable land management, greater international emphasis has been given to the tropical regions where poverty coupled with growing human population is triggering overexploitation of the limited natural resources base (Hartemink, 1998; Islam and Weil, 2000). Because of mounting demand for food crops, timber, pastureland and firewood, tropical forests are being degraded, cleared and converted to croplands at an alarming rate (Hall *et al*., 1993; Islam and Weil, 2000). The high rate of tropical deforestation and land use intensification are causing impacts that range from local to global scale, and the impacts are affecting processes that sustain the interacting systems of the global biogeosphere (Tinker *et al*., 1994). These impacts of tropical deforestation and land use intensification are imperative reasons to increase efforts to halt tropical deforestation and find a sustainable solution to it (Brady, 1994; Lal, 1996; Hartemink, 1997).

Countries in sub-Saharan Africa are specifically facing the greatest dilemma of rapidly degrading essential biophysical resources such as land, forest, water and energy, and the lack of appropriate technology necessary for increasing food production (Smaling *et al*., 1996). The situation in sub-Saharan Africa is further exacerbated by the high population growth in the region. Indeed, a special article in Agenda 21 (Chapter 14, Program area J) of the 1992 UN Conference on Environment and Development in Rio de Janeiro singled out the sub-continent as an area with a special focus for sustainable development.

Agriculture is the mainstay of the economy in most tropical countries and particularly in sub-Saharan Africa. Therefore, sustainable economic/rural development in these countries must inevitably deal with sustainable agriculture[1] (Smaling *et al.*, 1996; Kassa, 2003). According to Smaling *et al*. (1996) only sustainable agriculture is likely to provide the long-term benefits required for achieving development, environmental health, biodiversity conservation and poverty alleviation in this region.

On the other hand, despite the growing volumes about sustainability or sustainable land management, defining the term ‘sustainability’ remains problematic (Pretty, 1995). Following the Brundtland report of 1987 on ‘Our Common Future’ and the ‘Earth Summit’, there has been considerable debate about sustainability, much of which has centred around definitions. Since the Brundtland Commission’s definition[2] of sustainable development, there have been at least 70 more definitions constructed (Pretty, 1995). Bell and Morse (1999) remark that people differ in their concept of sustainability, and so do their visions of sustainable land management or sustainable agriculture. Indeed, major

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[1] Sustainability, sustainable land management and sustainable agriculture, though different in the strict sense of the terms, are also used synonymously in a general sense. In this thesis they are used in a general sense to refer to sustainable agriculture.

[2] Sustainable development is the management and conservation of basic natural resources, and the orientation of technological and institutional changes in such ways that the continued supply and satisfaction of human needs for present and future generations are ensured (Brundtland, 1987).

departures are because of the various groups and interests who are currently promoting different philosophies to guide agriculture (Francis and Youngberg, 1990). For instance, agronomists are interested in sustainable yield and pedologists may be concerned about soil quality or fertility, while agro-economists focus on farm income and equity. Furthermore, the definition of sustainability becomes too broad when spatial and temporal dimensions are included (Kassa, 2003). Consequently, some argue the relevance or need of defining sustainability to actually practise it (e.g. Gibbon *et al*., 1995; Pretty, 1995). Pretty (1995) suggests that it is easier to agree to describe goals for a more sustainable land use such as agriculture than to define sustainable agriculture. According to Pretty (1995), it would be more important to describe what is being sustained, to whose benefit, at what scale and how it is measured or monitored than to try to define it. On the other hand, Young (1997) and Hartemink (1998) are convinced that despite the various attempts at defining sustainability, the basic essence in all the cases remains the same and refers to the combination of productive use and maintenance/conservation of the productive capacity of a site on which the production depends.

The fact that sustainability is a concept that cannot be measured directly, coupled with the difficulty in defining it, makes monitoring land management progresses complex from a sustainability point of view (Kassa, 2003). Nevertheless, evaluation of land management practices, whether they are progressing towards or away from sustainability, can be made based on the performance of different components of sustainability (Dumanski and Smyth, 1993; Herrick, 2000; Nambiar *et al*., 2001). Certain attributes or components of land management may prove especially helpful in evaluating the sustainability of a particular system of land use because their status is highly relevant to performance and their instability in relation to known environmental pressures is highly predictable (FAO, 1993b; Pretty, 1995). Such attributes have been described as ‘Indicators’ of sustainability.

With respect to agricultural sustainability, different sets of indicators have been proposed. For instance, Gomez *et al*. (1996) proposed a framework for evaluating sustainability at farm level in the Philippines based on field indicators that take into account both the farmers’ satisfaction and resource conservation. High yield, low labour requirement, low input cost, high profit and stability are some of the features that are likely to enhance farmer satisfaction. Natural resource conservation is usually associated with soil depth, nutrient balance, organic matter content and biological diversity. Several others have also proposed and used similar indicators such as soil quality, nutrient budget, resource use efficiency, yield trend and variability, ecological impacts and so forth (e.g. FAO, 1993b; Vance, 2000; Nambiar *et al*., 2001; Arshad and Martin, 2002). Among the multitudes of possible indicators, many evaluators opt for objectively verifiable indicators, which are sensitive, specific, measurable, simple, usable and cost effective (Bell and Morse, 1999). The first task in assessing sustainability is, therefore, to define the indicators by which it is to be evaluated. In this study, the concept of sustainability was used both as an objective (measurable) and as a process (temporal), and the discussion was limited to the level of smallholder farming systems. The indicators used were classical soil attributes assumed to affect crop productivity and measurable with standard laboratory soil analytical procedures.

2.1.1. *Soil fertility and sustainable agriculture*

Soil is the foundation for nearly all land uses (Herrick, 2000). Together with water, soil constitutes the most important natural resource of our physical environment (Arshad and Martin, 2002). The wise use of this vital resource is essential to promote sustainable development, feed the growing world population and maintain environmental health (Wang and Gong, 1998; Hartemink, 1998; Arshad and Martin, 2002). The manner in which soils are managed has a major impact on agricultural productivity and sustainability (Scholes *et al*., 1994). In the past few decades alone, the global grain production growth rate has dropped from 3% in the 1970s to 1.3% in the early 1990s, which is one of the key indicators of declining soil quality on a global scale (Steer, 1998). Many agree that no agricultural system can be claimed to be sustainable without ensuring the sustainability of soil quality (fertility) (King, 1990; Arshad and Martin, 2002). Indeed, the maintenance or enhancement of soil quality is considered a key indicator of sustainable agricultural systems (Swift and Woomer, 1993; Bouma, 1994; Scholes *et al*., 1994).

Soil quality has been defined by many authors in recent years (e.g. Karlen *et al*., 1992; Doran and Parkin, 1994; Herrick, 2000; Arshad and Martin, 2002). Although the definitions are slightly different, all refer to the functions of the soil to supply plant nutrients and other physico-chemical conditions to plant growth, promote and sustain crop production, provide habitats to soil organisms, ameliorate environmental pollution, resist degradation and maintain or improve human and animal health (Wang and Gong, 1998). Generally, soil quality encompasses three basic components: physical, chemical and biological attributes of the soil (Ouedraogo, 2004). These chemical, physical and biological soil attributes determine the sustainable nutrient supply capacity of the soil for plant growth. Soil physical properties determine the capacity of the soil to provide plants with a foothold, moisture and air; and soil chemical conditions determine the capacity of the soil to provide plants with nutrition.

The term sustainable soil nutrition implies that plant nutrients and the soil physical environment suitable for plant growth remain at a steady state for the long-term. One way to insure sustainable soil nutrition is to make sure that all nutrients taken up by plants during growth are returned to the soil so that they can be used again by plants of the next production cycle. In this manner, a nutrient cycling is established (Fig. 1). In real agro-ecosystems, however, the cycle never closes because of losses and/or gains (Fig. 1). Managing the cycle to minimize losses plus supplying necessary inputs to compensate for inevitable losses is the key to soil fertility management and thus sustainable agriculture (King, 1990). The magnitude of nutrient losses and the extent of input substitutions vary considerably depending on the socio-economic and cultural setting of the agricultural system. Poor farmers in the tropics seldom lack the resources to sufficiently compensate for nutrient losses from the agroecosystem during production cycles. The maintenance or enhancement of soil fertility is even more difficult in cases such as the smallholders' cropping system in the highlands of Ethiopia, where crop residues and cow dung are considered important components of the harvest and removed from the crop fields. The removal of crop residues and cow dung not only interrupts recycling of plant nutrients but also contributes to a considerable depletion of the SOM, which exacerbates the decline in plant nutrient levels and soil productivity (Murage *et al*., 2000).

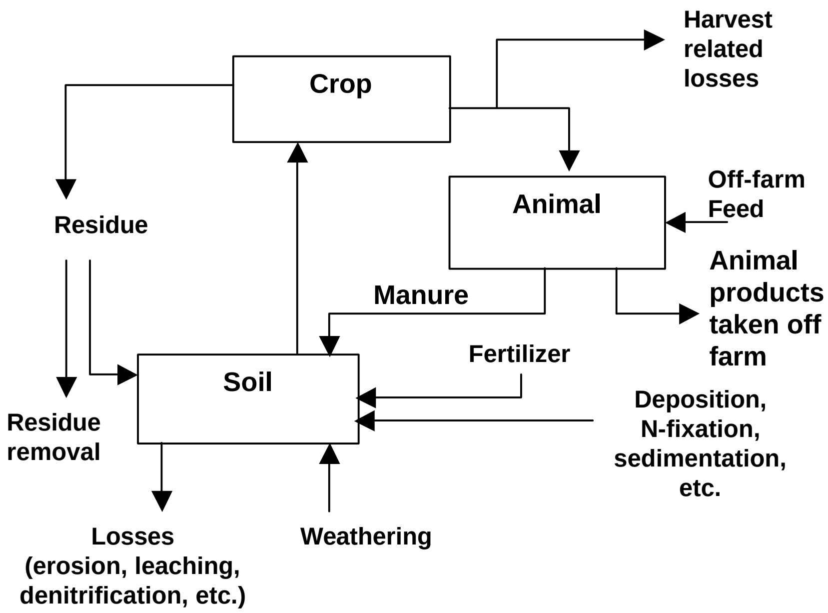

Fig. 1. System of nutrient flow on a farm (modified from King, 1990).

2.1.2. Soil organic matter as an indicator of sustainable agriculture

In all forms of agricultural systems, whether traditional or modern, SOM plays essential role in sustaining crop production and preventing land degradation (Ouedraogo, 2004). Several studies have given credence to the role of SOM in improving soil physical, chemical and biological properties (e.g. Paul and Clark, 1996; Fernandes *et al*., 1997). Because of its positive influence on several soil processes, crop productivity and environmental quality, SOM is often considered to be the single most important indicator of soil quality and sustainable land management (Roming *et al*., 1995; Vance, 2000; Doran, 2002). Moreover, SOM is a soil property that is generally most sensitive to crop management (van Noordwijk *et al*., 1997; Vance, 2000). Indeed, SOM management is envisaged to maintain soil fertility, crop productivity and promote sustainable agriculture (Ouedraogo, 2004), particularly in low input tropical agricultural systems (Sanchez *et al*., 1989; Kapkiyai *et al*., 1998). Soil organic matter is a relatively simple property to measure but may be characterised in many different ways.

Bolinder *et al*. (1999) suggest that it may be appropriate to measure the SOM content of the bulk soil, but in addition a number of other related properties of the SOM might prove to be more closely linked to changes in soil quality. These properties include C/N ratio, soil organic carbon, total nitrogen, light fraction and particulate organic matter, mineralizable carbon and nitrogen, microbial biomass, soil carbohydrates and soil enzymes. On the other hand, Sojka and Upchurch (1999) suggest a cautious approach towards the adoption of SOM as a more or less universal index of soil quality. According to Sojka and Upchurch (1999), even though there is evidence in many soils that an increase in SOM levels tends to improve the quality of the soil, there are many frequently negative environmental and crop production impacts, for instance an increased requirement of pesticide addition for efficacy, increased P solubility, etc. in soils with high SOM.

2.1.3. Using indicators to evaluate sustainability

Sustainability always refers to a temporal scale (FAO, 1993b) and implies equilibrium or steady state conditions in the states of the indicator variables over time (Hartemink, 1998). An agricultural system that continues to be productive for a long period of time without degrading its resource base can be claimed to be sustainable (Gliessman, 2001). Evaluation of sustainable agriculture, therefore, needs the assessment of how indicator variables are changing over time. Are the ecological foundations of the agricultural system being maintained or enhanced, or are they being degraded in some way? An agroecosystem that will someday become unproductive gives numerous hints of its future condition. These hints could be revealed by the direction of change (positive or negative, increase or decrease), magnitude of changes (percentage over a baseline value or rates of change), and/or duration of changes (Hartemink, 1998; Wang and Gong, 1998; Arshad and Martin, 2002) for the soil quality indicators. Measurement and comparison of selected indicators with desired values (critical limits or threshold levels) can be used to assess changes. Present values can also be compared with values at the commencement of the monitoring period (Arshad and Martin, 2002) or with historical data when available (Hartemink, 1998) or with soil quality attributes under reference ecosystems (Veldkamp, 1994; Feigl *et al*., 1995; Wang and Gong, 1998). By these measurements, it could be possible to identify whether an agricultural system is heading towards or away from sustainability. If the indicator variables are not showing signs of decline, the agriculture can be said to be sustainable, while if the indicator variables are showing signs of decline over time, the agricultural system is unsustainable (Gliessman, 2001).

2.2. Implication of tropical land use changes for biodiversity

In the aspiration to develop a sustainable society on the basis of agriculture and other biological resources, particularly forests, one must recognize that degradation of any biological resource or its habitat is not a sustainable practice. This is because biodiversity (i) is the foundation for sustainable development (Kumar, 1999); (ii) constitutes the basis for environmental health and determines the biogeochemical processes that regulate the Earth's system (Loreau *et al*., 2001); and (iii) is the basis for global and national economic and ecological security for the present and the future generations (Mugabe, 1998). In fact, it is biodiversity that enables many poor people, particularly those living in vulnerable ecosystems, to avert risks and insecurities today by diversifying their sources of livelihood. At present, rural people in the developing countries derive an estimated 90% of their daily needs directly from the biological resources (Kumar, 1999), and over 80% of the people in the developing countries depend on traditional herbal medicines obtained from the forests for primary health care (Farnsworth and Soejarto, 1991). Climatic and ecological changes that influence agricultural activities, economic development and human health are all regulated by biological diversity, particularly the forests. Consequently, the depletion of biodiversity will inevitably hamper sustainable development and endanger humanity's own future (Mugabe, 1998).

Tropical forests are storehouses of biodiversity (Brady, 1994). They are phenomenally rich in flora and fauna. According to Wilson (1988), over half the global number of species, which is estimated to be in the millions, is found in tropical forests. Tropical forests account for 52% of the total forest area of the world, of which 42% is dry forest, 33% is moist forest and 25% is wet and rainforest (Murphy and Lugo, 1986). Dry forests are next to rainforests in ecological complexity, which arises from the strong seasonal and inter-annual variability in rainfall, which permits the occurrence of very diverse flora and fauna (Khurana and Singh, 2001).

The largest proportion of tropical dry forests is found in Africa, where it accounts for 70–80% of the forested area (Murphy and Lugo, 1986; Teketay, 1996). Africa's rich biodiversity is estimated to comprise about 25% of global biodiversity in terms of ecosystems, species composition and genetic variety (Mugabe, 1998). Ethiopia hosts the fifth largest flora diversity, which is estimated to be between 6,500 and 7,000 species of higher plants, in tropical Africa (Sayer *et al.*, 1992), the richest avifauna in mainland Africa (Teketay, 2000), and is one of the 12 Vavilov Centres of crop diversity (Tedla and Gebre, 1998). Contrasting geo-climatic variations have induced rich floral and faunal diversity in Ethiopia. The highlands of Ethiopia alone contribute more than 50% of the tropical Afromontane vegetation in Africa (Yalden, 1983; Tamrat, 1993), of which tropical dry Afromontane forests cover the largest part (Teketay, 1996).

However, economic and demographic pressures are increasingly imposing non-sustainable development, which is driving greater proportions of tropical forests and their biodiversity to be either modified into more open and species-poor secondary forests or to be lost completely. Loss in biodiversity takes place in a variety of processes - direct and indirect (Fig. 2). In recent years, losses in biodiversity are proceeding largely due to the widespread loss and transformation of natural landscapes, for example such as occurs when forestland is converted to agricultural uses (e.g. Kumar, 1999; Laurance *et al*., 1998, 2002; Tole, 2001). Moreover, the transformation of natural landscapes accelerates biodiversity losses indirectly through the anthropogenic activities that degrade the self-repairing capacity of an ecosystem such as soil seed banks, soil fertility, etc. (Garwood, 1989; Brown and Lugo, 1990, 1994; Teketay, 1996). To protect biodiversity against the impact of these factors, it is necessary to have early warning of changes; hence the need for monitoring (Noss, 1990). In this thesis, the focus is on the soil seed bank aspects of biodiversity threats from land use changes that involve deforestation and subsequent cultivation.

Secondary succession on deforested and/or degraded sites begins from different regeneration pathways, namely soil seed bank, seed rain, seedling bank and coppices from damaged trees (Brokaw, 1985; Garwood, 1989; Teketay, 1996). Soil seed banks (SSB) are aggregations of viable seeds accumulated in or on the soil over several years and potentially capable of replacing adult plants (Bakker, 1989). Soil seed banks play a critical role in vegetation maintenance, succession, ecosystem restoration, differential species management and conservation of biological and genetic diversity (van der Valk and Pederson, 1989; Hills and Morris, 1992). The soil seed bank is one of those biophysical key factors that will determine the success of ecosystem recovery from disturbances (Bakker *et al*., 1996; Teketay, 1996; Tekle, 1998).

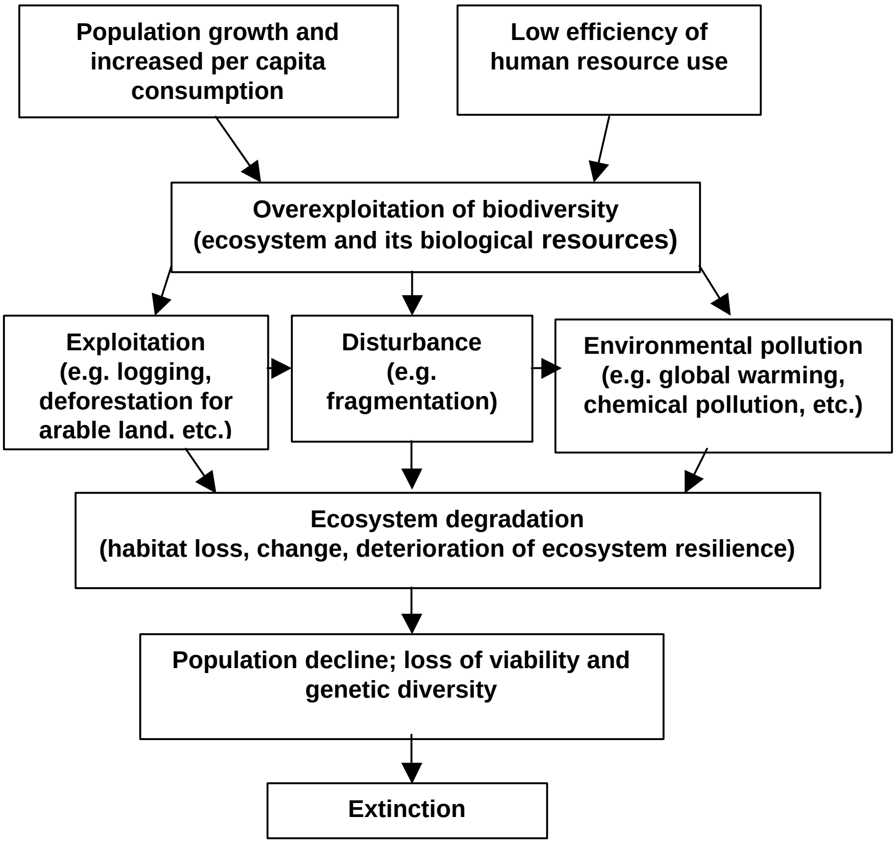

Fig. 2. Processes of human impacts that lead to biodiversity loss (Modified from Kumar, 1999).

Most plant communities include populations of viable seeds buried in the soil (Leck *et al*., 1989). Considerable proportions of such seeds are capable of germinating as soon as they are exposed to suitable conditions. Some of the seeds may exhibit a variety of dormancy types and persist in the soil for considerable periods of time. Persistence of seeds in the soil bank differs for various species depending on the nature of the seed (longevity), the environmental factors including seed predation, and anthropogenic factors such as land use (Bakker *et al*., 1996; Teketay, 1996). Some seed types are transient (Thompson *et al*., 1996), and they cannot persist over a year in the seed bank. These types of seeds easily vanish from the soil and their contribution to vegetation succession is probably low in landscapes affected by prolonged human use. The long-term persistent SSB (over 5 years) may often be available for use in restoration, and the more persistent they are, the more likely they are to be usable (Davy, 2002).

Disturbance or damage to an ecosystem is likely to affect all aspects of its successional status including soil seed banks. In particular, human land uses following deforestation are increasingly recognized as an important determinant of vegetation succession after abandonment (Compton *et al*., 1998), and persistent effects of prior land use have been reported in plant community assembly in temperate (Foster, 1992; Motzkin *et al*., 1996; Keersmaeker *et al*., 2004) and tropical (Foster *et al*., 1999) forests. Several studies have indicated that more heavily degraded sites due to human land uses start succession with lower recruit (seed bank) availability than do less degraded sites (Brown and Lugo, 1990, 1994; Teketay, 1996, 1997a & b, 1998; Lemenih and Teketay, 2004). The strength of the human land use effect in slowing or hampering forest succession is strongly related to the nature, duration and intensity of the land use (Brown and Lugo, 1990; Teketay, 1997a, 1998; Honnay *et al*., 1999). For instance, Davy (2002) indicates that conversion of natural ecosystem to plantation forestry allows good persistence of native soil seeds, while repeated cultivation of land leads to the destruction of the indigenous soil seed banks.

Obviously, agricultural land expansion is the major driver of tropical deforestation (Sanchez and Garduno, 1994) including Ethiopia (EFAP, 1994). Under the traditional shifting cultivation practice, both the small plot size slash and the short cropping duration might not significantly hamper vegetation recovery either from seed banks and/or from seed dispersal, also referred to as "seed rain", from the nearby natural stands. However as the intensity of use and the scale of clearance increase, fragmentation, depletion of SSB and the general deterioration of the growth condition of a site will hamper the recovery of the vegetation (Brown and Lugo, 1990, 1994; Teketay, 1997a, 1998).

2.3. Restoration ecology and sustainable development

Ecological restoration is a necessity where the basic needs of a society for survival are threatened due to biophysical land degradation. It provides theory and techniques to restore various types of degraded ecosystems (Kumar, 1999), and helps in reducing the length of time for which habitat remains in the degraded state (Dobson *et al*., 1997; Lamb, 1998). Ecological restoration also assists as a management tool for reversing the continuous loss of biodiversity (Dobson *et al*., 1997; MacMahon, 1997; Kumar, 1999).

2.3.1. Ecological restoration: definitions

Human inputs to the recovery of productive function and structure of degraded ecosystems differ according to the nature, degree, extent of degradation and the availability of resources. There are different words ascribed to human activities to recover land from degradation. Some of the common terms include: (i) restoration; (ii) rehabilitation; (iii) reclamation; and (iv) remediation (Bradshaw, 1997, 2002; Harrington, 1999). Restoration is defined as the return of an ecosystem (both the structure and the function) to a close approximation of its condition prior to disturbance (Miller *et al*., 1995; Bakker *et al*., 1996). The definition of ecological restoration has been broadened by the Society for Ecological Restoration to "Ecological restoration is the process of assisting the recovery and management of ecological integrity. Ecological integrity includes a critical range of variability in biodiversity, ecological processes and structures, regional and historical context, and sustainable cultural practices" (Society of Ecological Restoration, 1996). The attempt in restoration is to recreate, direct and accelerate natural succession to bring a 'full recovery' of the species composition, structure and function of the original ecosystem (Miller *et al*., 1995; Bradshaw, 2002). Apparently, complete restoration is rarely a realistic goal, because many of the ecosystems are so severely degraded that determining and reproducing the pre-disturbance state is difficult or even impossible (Otsamo, 2000). However, the important aspect of restoration is the intent to recreate an ecosystem based on biological models (Bradshaw, 1987; Harrington, 1999).

Rehabilitation is a broader term that refers to any attempt at repairing or restoring a damaged ecosystem, without necessarily attempting a complete restoration to any specific prior conditions or status (Harrington, 1999; Kumar, 1999; Bradshaw, 2002). In essence both restoration and rehabilitation are similar, but unlike restoration, rehabilitation contains little or no implication of recreating the original ecosystem (Bradshaw, 2002). The word 'rehabilitation' is used to indicate any act of improvement from a degraded state (Wali, 1992; Miller *et al*., 1995). Reclamation denotes rehabilitative work carried out on severely degraded sites, such as sites disturbed by open-cast mining, large-scale construction or in a sense of reclaiming land from the sea. The term has also been used in connection with conversion of degraded *Imperata* grasslands to fast growing forest plantations in Asia (Lamb and Tomlinson, 1994). Remediation is 'the act of remedying', i.e. rectifying or making good, and is often used in connection with cleaning of a site from toxic wastes. The emphasis in remediation is on the process rather than the end-point reached (Bradshaw, 2002).

Recently, the term restoration has been used widely in most ecosystem management activities aiming at re-establishing the functional and/or structural components of ecosystems. The use of tree plantations as foster ecosystems to re-establish native woody species, native fauna and soil fertility has been described as restoration (e.g. Lamb, 1998; Harrington, 1999; Ashton *et al*., 2001; Yirdaw, 2002; Duncan and Chapman, 2003). In this thesis the term 'restoration' is adopted to refer to the use of tree plantations in the recovery of soil attributes and native vegetation on degraded sites.

2.3.2. Restoration strategies

There are many theories and methods for accelerated restoration of degraded land and their biodiversity. Actions to restore degraded lands may comprise the fostering of beneficial factors and the removal of inimical ones (MacDonald *et al*., 2002). Available ecological restoration strategies can be grouped into two classes, as 'passive' or 'active' depending on the degree of human involvement (Allen, 1995; Laycock, 1995; McInvar and Starr, 2001). A passive approach seeks to restore the ecosystem by leaving the system alone, hoping that it will regain desirable structure and function through natural succession, i.e. by relying on the self-regenerating potential of ecosystems following the removal of degrading agents. An example of a passive technique is area closure (e.g. Tekle, 1998). Conversely, an active restoration approach involves active human intervention to complement and reinforce the self-regenerating potential of an ecosystem. An example of this kind of restoration is tree planting. Passive approaches are less effective for restoring highly degraded ecosystems (Laycock, 1995) and thus active restoration methods are often necessary (McIver and Starr, 2001). The choice of methods for restoration may depend on a wide variety of social, economic, cultural, biological and environmental factors (Miller *et al*., 1995; MacDonald *et al*., 2002). Nonetheless, the choice of method will significantly affect the speed with which the restoration process proceeds. For instance, most degraded landscapes in the highlands of Ethiopia have been remarked to be extremely poor in many aspects of self-regenerating potential such as soil status, SSB and seed dispersal (Lemenih and Teketay, 2004). Indeed, the very limited self-regenerating potential of severely degraded sites is rarely enough to initiate and expedite the restoration processes using a passive approach alone and thus calls for strategies that could augment and expedite the restoration processes.

2.3.3. Plantation forests in ecological restoration

Recently, several studies from the tropics have reported that both soil fertility and diverse native flora and fauna can be restored under fast growing timber plantations established on degraded tropical sites (e.g. Lugo, 1992, 1997; Fisher, 1995; Lugo *et al*., 1993; Parrotta *et al*., 1997; Fang and Peng, 1997; Datta, 1998; Hayes and Samad, 1998; Decher and Bahain, 1999; Sullivan *et al*., 1999; Senbeta and Teketay, 2001; Yirdaw, 2001; Zanne *et al*., 2001; Yirdaw, 2002; Senbeta *et al*., 2002; Chen *et al*., 2003; Jenkins *et al*., 2003). This phenomenon has led several ecologists to propose fast-growing plantation species as an ecological management tool for the restoration of degraded lands in the tropics (e.g. Lugo, 1997; Parrotta *et al*., 1997; Lamb, 1998; Harrington, 1999).

According to Harrington (1999), although the phrase "planting for ecosystem restoration" is of recent origin, many of the earliest large-scale plantings were made for what is now referred to as restoration. Furthermore, restoration by revegetation is just a modification of the long-standing tradition of folk wisdom that utilizes the soil ameliorating function of trees, while making productive gain from the forest crop. The long-known practices of swidden agriculture or shifting cultivation, for instance, take advantage of the soil improving potential of forest fallows (Nye and Greenland, 1960; Ewel, 1985; Sanchez, 1976; Jordan, 1985). This implies that the approach may have a higher probability of acceptability by the rural people than other approaches such as engineering techniques.

There are several mechanisms behind soil fertility enhancement and native species recolonization under plantation forests. Some of the mechanisms involved in soil influence by trees include: (i) enhanced mineral weathering, (ii) input of organic matter, (iii) nutrient pumping and recycling, (iv) symbiotic N-fixation, (v) interception of particles and dusts (e.g. aerosols) in the air, and (vi) improved soil structure through root action. There are also indirect beneficial effects of plantation forests through changed microclimatic conditions under the stand canopy, e.g. a decrease in soil erosion and fostering effects for the recolonization of diverse flora and fauna (e.g. Fisher, 1995; Kelly *et al.*, 1998; Raulund-Rasmussen *et al*., 1998; Olsson, 2001).

Similarly, some of the mechanisms by which planted trees foster recolonization of native flora and fauna include: (i) provision of perches for visiting birds and a corridor for wildlife, which are major seed and fruit dispersal agents (Parrotta *et al*., 1997; Wunderle, 1997); (ii) canopy shading, which improves the microclimatic conditions of the forest floor through a variety of functions (moderating soil moisture, reducing wind desiccation, moderating air humidity, protecting the emerging seedlings against strong direct sun radiation) (Keenan *et al*., 1997); (iii) improvement of the impoverished and often nutrient-poor soils of degraded lands by reducing bulk density, increasing the availability of nutrients, and accumulating more organic material through litter fall and root turn-over (Lugo, 1992; Parrotta, 1992, 1999; Fisher, 1995); and (iv) suppressing potentially competitive grasses that are common to degraded open lands and increasing the probability of native shade-demanding species emerging (Guariguata *et al*., 1995; Otsamo, 2000). Therefore, establishment of plantation forests not only reverses degradation of a site (e.g. Lugo, 1992; Parrotta, 1992; Fisher, 1995) but also creates a refuge for incoming seeds and emerging seedlings of native woody flora. Eventually, if the planted species are gradually and carefully removed without damaging the woody understory regeneration, a secondary forest could develop quickly (Parrotta *et al*., 1997).

3. Materials and methods

3.1. Location

The study was conducted near and in the Munessa-Shashamane forest, which is located on the eastern escarpment of the Central Ethiopian Rift Valley. The Munessa-Shashamane forest lies within latitudes 7°12’N and 7°32’N, and longitudes 38°45’E and 38°56’E at about 240 km south of Addis Ababa (Fig. 3). The Munessa-Shashamane forest is managed by a government enterprise called

Shashamane Forest Industry Enterprise (SFIE). Four of the studies were carried out in the Gambo district area of the SFIE, while the fifth study was conducted in the Degaga district area. The forest comprises approx. 25,000 ha of disturbed natural forest and 6,791 ha of plantation forests (Teshome and Petty, 2000). Geologically, the area is largely associated with the Wonji fault belt and carters. The main topographical feature is that the escarpment that is aligned North-Northeast to South-Southwest. This escarpment extends to over 4000 m above sea level at the Arsi-Bale massif and descends gradually to the plain beneath to the Central Rift Valley lakes at about 1500 m above sea level.

3.2. Soils

The soils of the area are closely related to their parent materials and their degree of weathering. The main parent materials are basalt, ignimbrites, lava, gneiss, volcanic ash and pumice (Makin *et al*., 1975). The large volcanoes within the southern Rift Valley of Ethiopia belong to the late Tertiary origin (Anonymous, 1988). The highland areas bordering the Rift Valley are characterized by deep, moderately weathered dark reddish brown soils of clay loams, which are all associates of the Rift Valley volcanic soils. Based on soil physical descriptions and chemical analysis, the soils around the lower elevation range of the Gambo district, where the major part of the study was carried out, are classified as Mollic Andosols (FAO, 1998) or Humic Haplustands (Soil Survey Staff, 1999). The soils around Degaga district are Typic Palehumults (Soil Survey Staff, 1999; Solomon *et al*., 2002). In Africa, Andosols occur in Ethiopia, Kenya, Tanzania, Rwanda, Cameroon and Madagascar. In the montane highlands of eastern Africa in general (Lundgren, 1978), and in the productive zones of the highlands and Rift Valley of Ethiopia in particular, Andosols cover large areas of land and support many subsistence-farming systems with large human and animal populations.

3.3. Climate

Climatic conditions in the highlands of Ethiopia are generally largely a result of differences in altitude. There are decreases in mean annual temperature and increases in mean annual rainfall with increasing elevation. The climate of the eastern escarpment of the Central Ethiopian Rift Valley, where the Munessa-Shashamane forest lies, falls into four eco-climatic zones that include semi-arid, warm sub-humid, humid, and cold-humid eco-climatic zones (Makin *et al*., 1975). However, the studies were mainly confined to the warm sub-humid and humid eco-climatic zones of the elevation gradient. The rainfall in the area is bimodal, with the main rainy season in the period from July to October and the short rainy season between March and May. Average annual rainfall amounts to 1200 mm at the warm sub-humid zone and 1370 mm at the humid zone (Anonymous, 1990). Temperature varies between the mean annual maximum of 25 ^{o}C and mean annual minimum of 10 ^{o}C across the elevation gradient covered by the Munessa-Shashamane forest (Teshome and Petty, 2000).

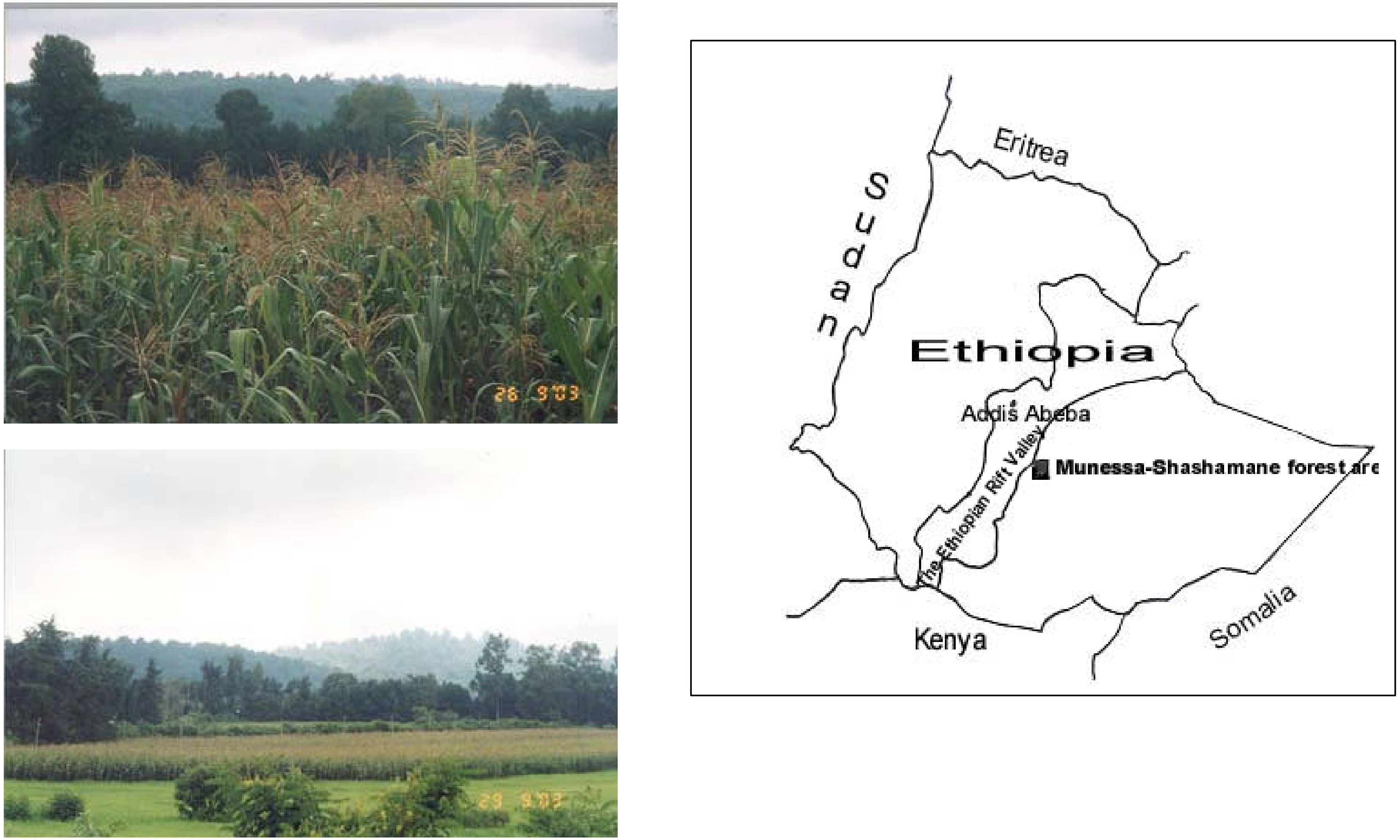

Fig. 3. Munessa-Shashamane forest on the eastern escarpment of the Central Ethiopian Rift Valley (Left: top and bottom), and a map of Ethiopia with location of the study area (Right).

3.4. Vegetation

The vegetation of Munessa-Shashamane forest is one of the conspicuous remnants of the once dense tropical dry Afromontane vegetation that covered the highlands of Ethiopia (Fig. 3). The main forest blocks of the Munessa-Shashamane forest are located on the westerly aspect of the eastern escarpment of the Central Ethiopian Rift Valley. The vegetation covers the altitudinal range from 1500 m above sea level at its bottom in the Rift Valley to over 4000 m above sea level in the Arsi-Bale Mountains and the associated plateau. The Munessa-Shashamane forest, just like other east African vegetation as a whole, can be divided into vegetation zones according to altitude and humidity (Lundgren, 1971). At the Rift Valley plain, open *Acacia* woodland dominates, and this gradually turns into dry open deciduous woodland of a transitional vegetation type (Eriksson *et al*., 2003). At mid-altitude along the escarpment, i.e. between 2100-2600 m above sea level which is also the altitudinal range of the study sites, tropical dry evergreen montane forest dominates. Different plant communities comprise this section. At the lower sub-humid part a *Podocarpus falcatus* - *Croton macrostachyus* mixed forest exists, which gradually converts into the humid zone dominated by *Podocarpus falcatus* forest. These vegetation communities along the elevation transect are all referred to as ‘Montane forests’ in many classification systems (von Breitenbach, 1961, 1963; Brown and Cocheme, 1969; Chapman and White, 1970; White, 1983; Friis, 1992; Tamrat, 1993). In the sub-humid and humid section of the escarpment, *Podocarpus falcatus* makes up the dominant upper tree story. Other major co-dominant tree species include *Croton macrostachyus, Ekebergia capensis, Celtis africana* and *Prunus africana.*

3.5. Deforestation and agricultural land expansion in the area

The Munessa-Shashamane forest has been under continuous pressure from deforestation for a long period, a process that is still ongoing. Early reports concerning deforestation of Munessa-Shashamane forests date back to the 1940s and 1950s (Russ, 1946; Mooney, 1954). According to Russ (1946) and Mooney (1954), commercial logging as well as agricultural land expansion were important agents of deforestation at that time. In recent decades, human influxes to the area from the central and southern highlands have intensified the pressure on the Munessa-Shashamane forest (Anonymous, 1990; Tolera, 1996). Consequently, forest clearance in search of crop and grazing lands has increased (Seifu, 1998; see also Fig. 4). The pressure from increasing human population and unplanned utilization has decreased the area and productivity of the forest (Anonymous, 1990). According to Seifu (1998), cropland is increasing at the rate of 2.8% per annum in the area, while natural forests and woodlands are declining at estimated rates of 1.7% and 2.6% per annum, respectively. To improve the productivity of the forest, plantation forest has been introduced to the Munessa-Shashamane forest since the late 1960s.

Beside the increased pressure from the growing population, an unstable political system has aggravated the rapid decline of the Munessa-Shashamane forest, particularly since the 1970s (Seifu, 1998). The period after 1970 marked the most politically unstable time in the modern history of Ethiopia. This instability significantly impacted on the natural forest resources of Ethiopia (Bekele, 2003), including the Munessa-Shashamane forest (Seifu, 1998). Nevertheless, the Munessa-Shashamane forest is still one of the prominent remnants of the tropical dry evergreen montane forests found in the highlands of Ethiopia. Today, the Munessa-Shashamane forest is one of the forests in Ethiopia designated as High Priority Forest Areas (HPFA) for protection by the Ethiopian government.

Fig. 4. Deforested and cultivated forestland in the Munessa-Shashamane forest area, Ethiopia. Scattered on-farm trees are remnants from the natural forest. The trees form the traditional parkland agroforestry system of the farming system in the area.

3.6. Plantation development

Starting from the late 1960s, plantation development has been carried out in the Munessa-Shashamane forest (Lundgren, 1971). The plantations are established predominantly on freshly cleared but previously degraded parts of the natural forest, while some stands were also established on abandoned farm fields. The natural forest sites were manually cleared including cutting of bushes, climbers and other ground vegetation (Teshome and Petty, 2000). Plantation forests are increasing over time at an annual planting rate of 6% in the Munessa-Shashamane forest (Seifu, 1998). At present, plantation forest covers some 6,791 ha of land in the area (Teshome and Petty, 2000).

The bulk of the plantations in the Munessa-Shashamane forest comprise monocultures of *Cupressus lusitanica*, *Eucalyptus* spp. and *Pinus patula. C. lusitanica* and *P. patula* are planted mainly for sawn timber, while *Eucalyptus* spp. are planted mainly for pole production. The plantations are established from seedlings, which is a common plantation establishment practice in Ethiopia. The stands are characterized by wide spacing, commonly at 2.5 x 2.5 m (1600 tree/ha). Most stands, except *Eucalyptus* spp., are often clear-cut between the ages of 25-30 years. Most of the stands, other than *Eucalyptus,* are also thinned two to three times before the final clear-cut harvest. *Eucalyptus* stands are often harvested on a short rotation basis, usually 10-15 years (Lemenih and Bekele, 2004).

3.7. Farming system and cropping practices in Munessa

The farming system of the area is predominantly subsistence farming based on mixed crop-livestock production. Cattle provide inexpensive and easily accessible inputs required for crop production such as draught and threshing power in the agricultural production system, while crop production supports the livestock by providing crop residues that supplement the feeds required by the livestock. After crop harvest, cattle are allowed to graze on the weeds and remaining crop stalks on the croplands. However, most of the main grazing is carried out in the forest and on communal grazing lands. Manure from the cattle is mainly used for homestead gardens, while those farm fields away from the home garden, which are the focus of the present study, often receive little or no manure. Tillage involves a simple oxen-plough that cultivates the soils to a shallow depth (oral communication with the farmers). Major crops grown in the area are maize, sorghum, wheat and barley, mainly with one harvest per year. Fertilizer application is very limited. Whenever applied, the type of fertilizer used in the area is diammonium phosphate (DAP) in combination with urea. According to the farmers in the study area, the rate of fertilizer application is 50 kg ha^{-1} yr^{-1} of each type. However, most farmers sometimes use half the amounts recommended or often even none due to economic constraints. The farming system is a traditional parkland agroforestry system with scattered trees on farms (Fig. 4). The trees are preserved from the original forest during clearance. The species and density of the trees vary across the eco-climatic zones. Some of the trees preserved on-farm in the area have been recognized as soil improvers (Jiru, 1989; Gindaba, 1997).

3.8. The study approach

Two approaches are often used in studying ecosystem dynamics. The first and ideal type of approach is ‘temporal monitoring’, where the dynamics of ecosystem components (e.g. soil, plant, etc.) are examined over time at a single site. This is feasible where long-term data are available and changes in ecosystem components over time can be directly measured. Unfortunately, such long-term data are rarely available in the tropics (Sanchez *et al*., 1985; McDonagh *et al*., 2001) and this is also the case in Ethiopia. Furthermore even if available, such data usually come from on-station experiments that do not reflect the environment and management conditions of the farmers’ fields (McDonagh *et al*., 2001). Therefore, it is rarely possible to follow this approach (Bhojvaid and Timmer, 1998; Hartemink, 1998), and there are few such studies (Sanchez *et al*., 1985). An alternative approach is to use the spatial analogue and chronosequence methods (Young, 1991; Bhojvaid and Timmer, 1998). The spatial analogue method involves spatial sampling on sites that are subject to different land uses but operating within a similar environment and on similar soil types. The chronosequence method is a synchronized spatial sampling from neighbouring sites of different ages managed on similar soils, and under similar climatic conditions and management practices (Young, 1991; Hartemink, 1997). These approaches have been widely used in several contexts such as to: (i) assess long-term effects of global climate change (Tate, 1992); (ii) long-term changes in soil productivity (Martin *et al*., 1990; Arrouays *et al*., 1995); (iii) evaluate effects of deforestation and subsequent cultivation (Sanchez *et al*., 1985; McDonagh *et al*., 2001); (iv) assess soil C dynamics due to long-term land uses (Balesdent *et al*., 1988; Dominy *et al*., 2002); and (v) study nutrient dynamics and carbon storage changes (Marques and Ranger, 1997; Garten, 2002).

Chronosequence or spatial analogue methods have the danger of confounding time with possible spatial variability, and assume that all measured differences reflect the effects of time or management and not inherent spatial variability. Although the limitations of using such approaches are well acknowledged (e.g. Sanchez *et al*., 1985; Bhojvaid and Timmer, 1998), they have been and still are widely used in studying different aspects of ecosystem dynamics (e.g. Vernberg, 1988; Greenland and Swift, 1991; Marques and Ranger, 1997; Bhojvaid and Timmer, 1998; McDonagh *et al*., 2001). A major advantage of these techniques is that they provide data on long-term changes in soil, plant or other ecosystem components within a reasonable time. In situations like Ethiopia in particular, where data on long-term experiment are very rare, the chronosequence and spatial sampling approaches are valuable alternatives to study ecosystem dynamics in a temporal perspective. According to Young (1991) the analysis of soil fertility gradients using soil survey data along a chronosequence or spatial sampling under different land use/management regimes could yield important information on where, and to what extent, a soil fertility decline is taking place and a position could be reached from which to take action to arrest or reverse the problem.

In this thesis, for some of the studies (Papers I, II and III) the chronosequence approach was adopted, while for the rest (Papers IV & V) a spatial analogue approach was employed. A necessary assumption made in these research approaches is that soil conditions or other parameters of interest for all the sites studied should be similar before changes in the land use have been introduced. This is because observed differences in present soil conditions or other parameters can be interpreted as being caused by the present land–use practices only if the conditions were assumed to be comparable prior to the introduction of the new land management. Similarities between sites in soil properties that are known to be little influenced by land use and time can be used to justify comparability for soil studies in a chronosequence or spatial analogue system. Soil type and particle-size distribution are properties that are frequently used as indicators of comparability of two or more spatially sampled soils (Sanchez *et al*., 1985; Lilienfein *et al*., 2000).

Soil types at the study sites were similar and particle size also showed a fairly similar distribution with soil depth (Papers I and IV). The sensitivity of soils derived from volcanic parent materials to land use changes is a problem here. Land use changes that involve deforestation and subsequent cultivation have been shown to affect soil processes in soils of volcanic origin so drastically that even characteristics that are considered permanent, such as soil type and particle size distribution, can be significantly affected (Wielemaker and Lansu, 1991). Therefore, differences can be observed in particle size distribution, which can be

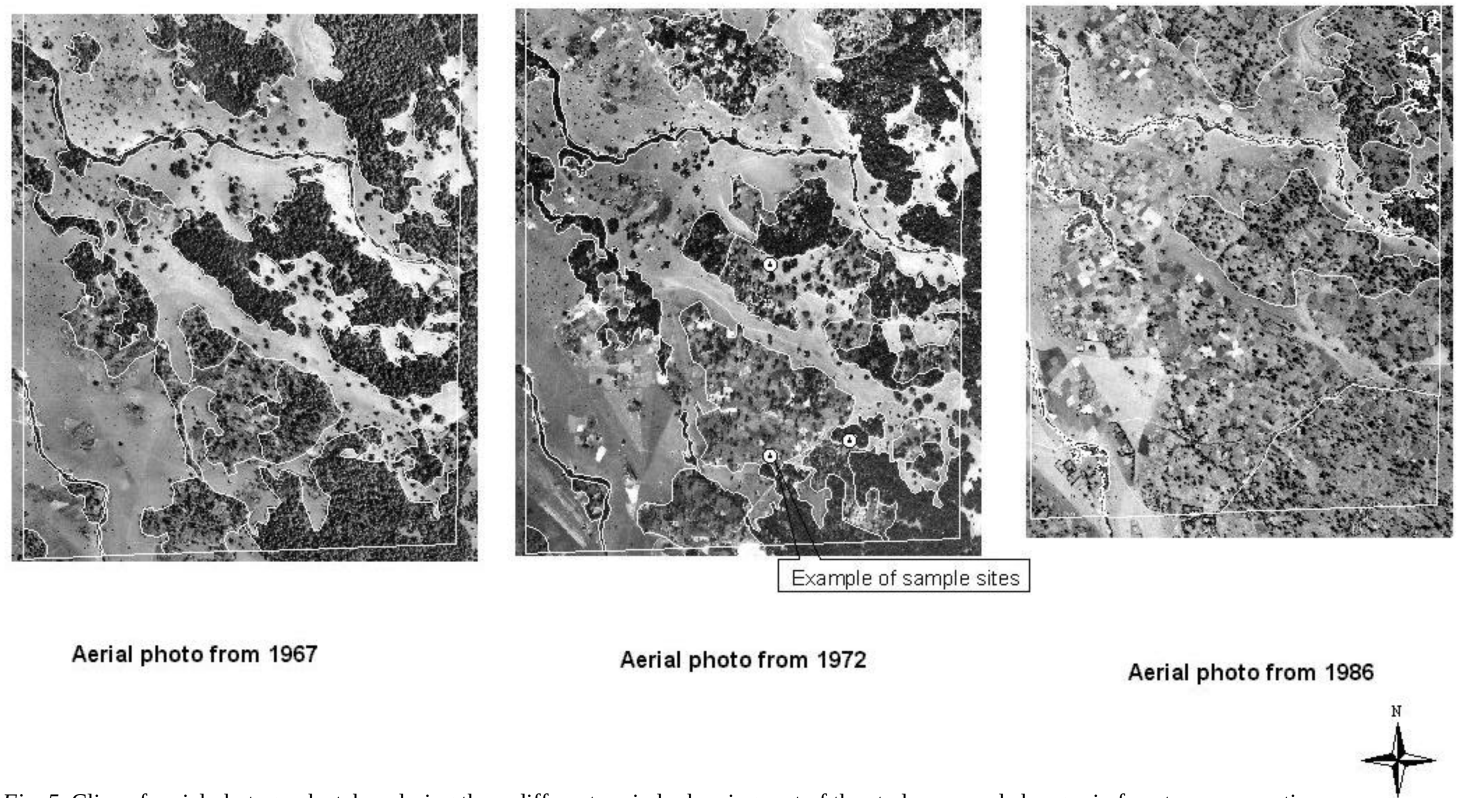

Fig. 5. Clips of aerial photographs taken during three different periods showing part of the study area and changes in forest cover over time.

attributed to the impact from deforestation and farming practices such as repeated tillage. In this thesis, the similarity in all particle size distributions in the sub-surfaces (those parts of the soils that are little affected by the changing landmanagement), and particularly similarity in the clay fraction of all depths (Sanchez *et al*., 1985), supports the assumption that soil conditions prior to the shifts in land management were more or less similar (Papers I, IV).

Whenever spatial sampling is used, a precise understanding of the historical land use changes is another prerequisite to guide the sampling and interpretation of the results. However, construction of historical land use changes and determination of the time of their introduction is often problematic. Remote sensing, satellite imagery, aerial photographs and radar images can be invaluable in getting information on the time of land-use introduction whenever available. Images and photographs can be used to detect and map historical land cover/land use changes and to determine their ages.

Availability and/or cost may limit the use of remote sensing and thus gaps in time are inevitable between two or more sets of remote sensing data. To compensate for the gaps in information from remote sensing data, interviews with local people are a valuable complement. Besides filling the gaps in information, interviewing also clarifies any ambiguities in the interpretation of remote sensing data. In this study, information on the historical land cover/land use and the respective ages of the sites was obtained from aerial photographs taken during three different periods (Fig. 5) complemented with informal interviews with the local farmers and institutions in the surroundings of the study area.

3.9. Soil and vegetation sampling

3.9.1. Studies of soil attributes on the farm fields (Papers I and II)

A chronosequence of closely located farm fields with 7, 10, 26, 34 and 53 years since conversion from natural forest and the adjacent natural forest were selected. One criterion in the selection of the fields was that they had been cultivated only with C_4 crops (maize and sorghum in this case) since clearance. Soil samples were then collected in five replicates from a square 20 x 20 m^2 plot established in the natural forest and at each farm field site. Pits were dug at the four corners and in the centre of the square plots and samples were removed from 0-10, 10-20 and 20-40 cm increments with a hand trowel uniformly along each depth.

Two additional samples from 60-70 and 90-100 cm depths were taken from the pits dug at the centre of the square plot. The soil profiles in each pit at the centre of the square plots were also described according to FAO guidelines for soil profile description (FAO, 1990). The soil samples were air-dried, mixed well and passed through a 2 mm sieve for chemical analysis. Separate soil core samples from the 0-10, 10-20 and 20-40 cm depths were taken with a sharp-edged steel cylinder of 5 cm height and 7.2 cm diameter forced manually into the soil for bulk density determination.

The major part of the soil analysis was carried out in Addis Ababa (Ethiopia) at the National Soil Research Laboratory. Analyses of soil C, total N and stable isotopes were made in Sweden at the Swedish University of Agricultural Sciences, Uppsala (Sweden). Soil pH was measured in water and 1M KCl suspension of 1:2.5 (soil: liquid ratio) potentiometrically using glass-calomel combination electrodes. Organic C and total N were determined using a LECO-1000 CHN analyser and the results were reported on an oven-dry basis. Testing with HCl in all the soil pits did not show signs of carbonate, so we assumed that the total C obtained in the analysis closely estimates the organic C contents of the soils. For ^{13}C and ^{15}N analyses, a few grams of the sieved mineral soil from each sample were drawn after thorough mixing and grinding into a fine powder. The C and N contents and the natural abundance of ^{13}C and ^{15}N were measured on the powdered samples on an automated on-line C and N analyser coupled with an isotope ratio mass spectrometer. Stable isotope abundances are expressed using δ notation in per mil (‰), as the deviation of the isotopic ratio of the sample from that of an arbitrary standard as shown in Paper II. Available P (Olsen) was analysed according to standard methods (Olsen *et al*., 1954), exchangeable bases (Ca, Mg, K and Na) were analysed after extraction using 1M ammonium acetate at pH 7.0. Ca and Mg in the extracts were analysed using an atomic absorption spectrophotometer, while Na and K were analysed by flame photometer (Black *et al*., 1965).

After displacement of the exchangeable base-forming cations using 1M ammonium acetate, the samples were washed using ethanol and the ammonium on the saturated exchange sites was subsequently replaced by the addition of Na. Cation exchangeable capacity (CEC) was thereafter estimated titrimetrically by distillation of ammonium that was displaced by sodium (Chapman, 1965). Percentage Base Saturation (BS) was calculated by dividing the sum of the charge equivalents of the base-forming cations (Ca, Mg, Na and K) by the CEC of the soil and multiplying by 100. Particle size analysis was performed using the Boycous hydrometric method, after destroying organic matter using hydrogen peroxide and dispersing the soils with sodium hexameta phosphate (Black *et al*., 1965). The USDA particle size classes, viz. Sand (2.0-0.05 mm), Silt (0.05-0.002 mm) and Clay (<0.002 mm), were followed when assigning textural classes. Bulk density was determined after drying the core samples in an oven at 105 ^{o}C, and ethanol displacement was used for particle density determination. Percentage pore space was computed from the bulk and particle densities determined (Brady and Weil, 2002).

3.9.2. Study on soil seed banks (Paper III)

The same chronosequence of farm fields and the natural forest used for the studies in sub-section 3.9.1. above were used for the SSB study as well. Five random plots, one square metre each, were selected at each site. Three subplots of 15 cm x 15 cm were marked in a triangular shape at the centre of each plot. Soil samples were removed from 0-3, 3-6 and 6-9 cm soil layers of each subplot using a sharp knife and spoon. Soil samples of the corresponding soil layers from the three sub-plots of each plot were mixed in a plastic bag and later divided into three equal parts of which one was randomly chosen as the working sample for incubation in a glasshouse. The rationale for taking sub-plots was to capture spatial heterogeneity of soil seed distribution. From the natural forest, a fourth layer containing litter was also sampled. The soil samples were then transported to the Ethiopian Agricultural Research Organization (EARO) Headquarters in Addis Ababa where the germination trial was conducted in a glasshouse. Soil samples were spread to a thickness of about two centimetres on plastic trays lined with cotton cloth and kept continuously moist. A seedling emergence method was used to assess the composition of the SSB over a period of one year. Emerging seedlings that were readily identifiable were counted, recorded and discarded. Seedlings that were difficult to identify were first counted, labelled and transplanted and grown separately until they could be identified. Each month, the soil samples were stirred to stimulate seed germination. For a comparison of soil seed bank flora with the above-ground vegetation in the adjacent natural forest, 10 sample plots of 314 m^2 area each were randomly selected in the natural forest. All woody species found in these plots were first identified. Thereafter, sub-plots of 2.5 m x 2.5 m, marked at the centre of each plot, were used to identify all herbaceous species including grasses and sedges.

3.9.3. Study on soil attributes under plantation forests (Paper IV)

To study the effect of reforestation of abandoned farm field on soil attributes, soil samples were collected from *E. saligna* and *C. lusitanica* stands, farm fields subject to traditional farming (TF) and mechanized farming (MF) and from the adjacent natural forest. The plantations were established on an abandoned part of the MF site. The adjacent natural forest was used as a reference site in the investigation. Five replicate soil samples were taken from 0-10, 10-20 and 20-40 cm increments from all the sites. Soil analyses followed the same procedure as in Paper I.

3.9.4. Study on regeneration of native forest flora (Paper V)

Five stands were selected to encompass exotic and indigenous species as well as broadleaved (open canopy) and coniferous (dense canopy) species from the plantations in the Degaga district of SFIE. The plantation species involved were *Cordia africana* (broadleaved & indigenous), *Eucalyptus saligna* (broadleaved & exotic), *Pinus patula* (coniferous & exotic), and *Cupressus lusitanica* (coniferous & exotic). In each stand, canopy closure percentage (CCP) and leaf area index (LAI) of the plantation, air and soil temperatures and soil moisture at the forest floor were measured in 10 plots. Canopy closure percentage was measured at the centre of each plot using a convex model spherical densiometer held at chest height (Lemmon, 1956) and LAI was measured using a Li-Cor LAI-2000 Plant Canopy Analyser. Air temperature (at 0.30 m height above the ground) and soil temperature (at 5 cm soil depth) of the forest floor were monitored for 15 days under the canopies of the plantation species and in the natural forest using air and soil thermometers. Soil moisture content of the forest floor was determined from five composite soil samples taken from the upper 0-5 cm soil layer of each plot using a core sampler of 204 cm^3 volume.

In each of the 10 randomly selected plots, the identities and number of individuals of naturally regenerating woody species were assessed and diameter at breast height (DBH) and heights of all regenerates with a DBH greater than 1.0 cm were measured and recorded under the plantation forests. For the seedling/sapling populations in the natural forest, height and DBH measurements were restricted to individuals within 1.0 to 10.0 cm DBH ranges.

3.10. Data analyses

3.10.1. Soil attributes

The data obtained from the soil analyses were subjected to one-way Analysis of Variance (ANOVA) for each sample depth separately to detect whether differences in the soil attributes studied differed significantly between the land uses. Least significant difference (LSD) was used for mean separation for those properties that were found to be significantly different. The level of significance used was 0.05. In Papers I and II, the soil C of the bulk soil (g kg^{-1}), the total N (g kg^{-1}), $\delta^{13}C$, $\delta^{15}N$, the fraction of soil organic carbon (SOC) (g kg^{-1}) derived from forest origin (C_f), and the fraction of SOC (g kg^{-1}) derived from agricultural crop (C_C) in the surface 0-10 cm layer were related to cultivation period and also to each other using exponential and linear regression functions. In cases where soil C stocks were reported in g m^{-2} calculations were performed using the equation:

$$C = z r_b c \qquad (1)$$

where:

C = carbon stock (g m^{-2}) of a sample soil layer
z = thickness (m) of a sample soil layer,
$ñ_b$ = bulk density (kg m^{-3}) of a sample soil layer,
c = carbon content (g kg^{-1} soil) of a sample soil layer.

Total N (g m^{-2}) stocks were also computed using a similar equation. Differences in soil bulk density between sites affect comparisons of SOC stocks between different land uses/land covers by influencing the amount of soils that are sampled from a fixed soil depth (Solomon *et al*., 2002; Mendham *et al*., 2003). Such differences in soil bulk density between the natural forest and other sites (the farm fields and the plantations) in the present study were accounted for by adjusting the thickness of sampled layers from each site with respect to equivalent weights of soils under the natural forest applying the equation:

$$z_{corr} = \frac{r_{forest}}{r_{LU}} z \qquad (2)$$

where:

z_{corr} = adjusted thickness of sample soil layers under plantations or on farm fields,
$ñ_{forest}$ = bulk density of the sampled soil layers under the natural forest,
$ñ_{LU}$ = bulk density of the sampled soil layers under plantations or on farm fields,
z = thickness of soil layers used during field sampling (Solomon *et al*., 2002).

3.10.2. Degradation index

Commonly, when evaluating the impact of land use change on soils, the soil status under a new land use is compared with a pre-existing steady-state base line, ideally the native vegetation, which can then be expressed as degradation (also called deterioration) index (DI) (Adejuwon and Ekanade, 1988; Islam and Weil, 2000). To compute DI in the present study, the difference between mean values of individual soil properties from the farm fields or the plantations and the baseline values of similar soil properties under natural forest were computed and expressed as a percentage of the values under the natural forest. These percentage changes were summed across all soil properties to compute the soil DI for each site, which was used as an index of soil degradation or improvement. Values of pH, C/N ratio and exchangeable Na were not included in this calculation because the criterion of ‘more is better’ is uncertain over the range of values in this study for these soil properties.

3.10.3. Threshold levels

Changes in the soil properties are indicators of the way the soils are managed, i.e. whether the soil management is sustainable or not. However, whether the changes observed have reached the level that affects sustainable crop production, also known as threshold/critical level, cannot be determined from the analysis of soil attributes alone. This assessment is comparable to the qualitative land evaluation procedures in which soil qualities are matched with crop requirements (FAO, 1976; Hartemink, 1998). FAO’s qualitative land evaluation procedure was used to evaluate whether threshold levels had been reached for the observed soil changes in this study. Maize, one of the dominant crops grown in the area, was selected for the evaluation. Requirements of maize for pH, CEC, BS%, P, K and N levels were obtained from Landon (1991) and Stahr (2000), both of which refer to tropical environments. In particular the latter, which was the main source of the information, is from Africa.

3.10.4. Fractionation of soil organic carbon based on ^{13}C abundance

The fraction of organic C derived from the natural forest (C_f) and the agricultural crops (C_c) along the chronosequence was calculated based on the $\delta^{13}C$ values of the soil samples taken from the forest (δ_{fs}) and the agricultural fields (δ_{as}), as well as average $\delta^{13}C$ values of the forest vegetation (δ_f) and the samples of agricultural crops (δ_c). The average $\delta^{13}C$ value of the agricultural crop materials (maize & sorghum) collected from the study sites was -10.59‰ ± 0.023. We did not sample the vegetation in the natural forest, but used a value from a recent study made in the same forest (Solomon *et al*., 2002), which reported $\delta^{13}C$ value of -26.5‰ ± 0.9. The organic C derived from the agricultural crop (C_c) and the natural forest (C_f) was calculated and expressed as a percentage of the total SOC from the farm fields by the following equation:

$$C_c[\%]=\left(\frac{(d_{as}-d_{fs})}{(d_c-d_f)}\right)100\,; \qquad C_f[\%]=100-C_c[\%] \qquad (3)$$

where:

C_c = organic C fraction derived from the agricultural crop (C_4 plants),
δ_{as} = $\delta^{13}C$ value of the soils taken from the farm fields,
δ_{fs} = $\delta^{13}C$ value of the soils taken under the natural forest (C_3 plants,
δ_C = average $\delta^{13}C$ value of the agricultural crops,
δ_f = average $\delta^{13}C$ value of the forest vegetation,
C_f = organic C fraction derived from the natural forest.

3.10.5. Estimating loss of soil organic carbon of forest origin

The loss of forest-derived SOC denoted as Loss (C_f)[%] as a result of deforestation and subsequent cultivation along the chronosequence was calculated as follows:

$$Loss(C_f)[\%] = 100 - C_f[\%]\left(\frac{C_{as}}{C_{fs}}\right) \qquad (4)$$

where C_{as} and C_{fs} denote SOC of the bulk soils in the farm fields and the forest soils respectively.

3.10.6. Soil seed banks

To compare whether the variations in species composition and viable seed densities of the SSB from the different study sites were significantly different, one-way analyses of variance (ANOVA) were performed. The mean number of species per plot was used as an index of species richness, and the mean seed density per plot was used as a measure of abundance during the ANOVA. Least significant difference (LSD) was used for mean separation for those properties that were found to be significantly different. Jaccard's Similarity Coefficient (JSC) (Krebs, 1985) was also calculated to assess the similarity in the species composition between the SSB and the standing vegetation in the adjacent natural forest. The coefficient, which is expressed in equation 5, was calculated based on the presence-absence relationship between the number of species common to the two sites and the total number of species from the two sites. This means that the coefficient expresses the ratio of the common species to all species found on the two sites:

$$JSC = \frac{j}{(a+b-j)} \qquad (5)$$

where j is the number of species common to both sites, and a and b are the number of species found on sites a and b, respectively. The coefficient has a value from 1 to 0, where 1 demonstrates complete similarity and zero complete dissimilarity.

3.10.7. Regeneration of native forest flora

To facilitate interpretation of the results, simple metrics of quantifying the diversity, abundance and growth were employed. The average number of observed species per plot was used as an index of species richness, while the average stem density of the naturally regenerating woody species was used as a measure of abundance. Growth was characterized by computing mean DBH and mean height of all regenerating woody species with DBH of more than 1.00 cm combined, and by sorting the naturally regenerating woody species into shrub and tree life forms per plot and stand. The means of thickest DBH and tallest heights were also computed by averaging the DBH and heights of individuals with the dominant diameter and height per plot and stand. The data sets on CCP, LAI, air temperature, soil temperature, composition, density, mean DBH and mean height were subjected to separate statistical analyses using one-way ANOVA. For those parameters found to be significantly different, LSD (5%) was run to separate between overstory species with statistically significant differences.

4. Results and discussion

4.1. Soil physical and chemical property responses

Soil bulk density has increased in the 0-10 and 10-20 cm layers relative to the length of time the soils were subjected to cultivation. Along with the increase in soil bulk density, soil porosity showed marked declines in both soil layers with increasing period under cultivation (Paper I). Levels of SOC and total N in the surface soil (0-10 cm) were significantly lower, and declined increasingly with cultivation time on the farm fields, compared to the soil under the natural forest. The changes in the physical soil attributes on the farm fields can be attributed to the impacts of frequent tillage and the decline in SOM content of the soils. The decline in SOC and total N, although commonly expected following deforestation and conversion to farm fields, might have been exacerbated by the insufficient inputs of organic substrate from the farming system due to residue removal and burning on the farm fields**.**

Despite the significant declines in soil bulk density, soil porosity, SOM and some other soil attributes, the overall rate of change in soil properties due to the land use change was not as rapid as first anticipated for such a low input, tropical farming system as the one studied (Paper I). Some of the soil nutrients on the farm fields had higher concentrations for quite some time compared to the soil under the adjacent natural forest (Fig. 6). The levels of some of the soil nutrients did not fall below their values in the soil of the adjacent natural forest even after 53 years of subsequent cultivation following deforestation (Fig. 6).

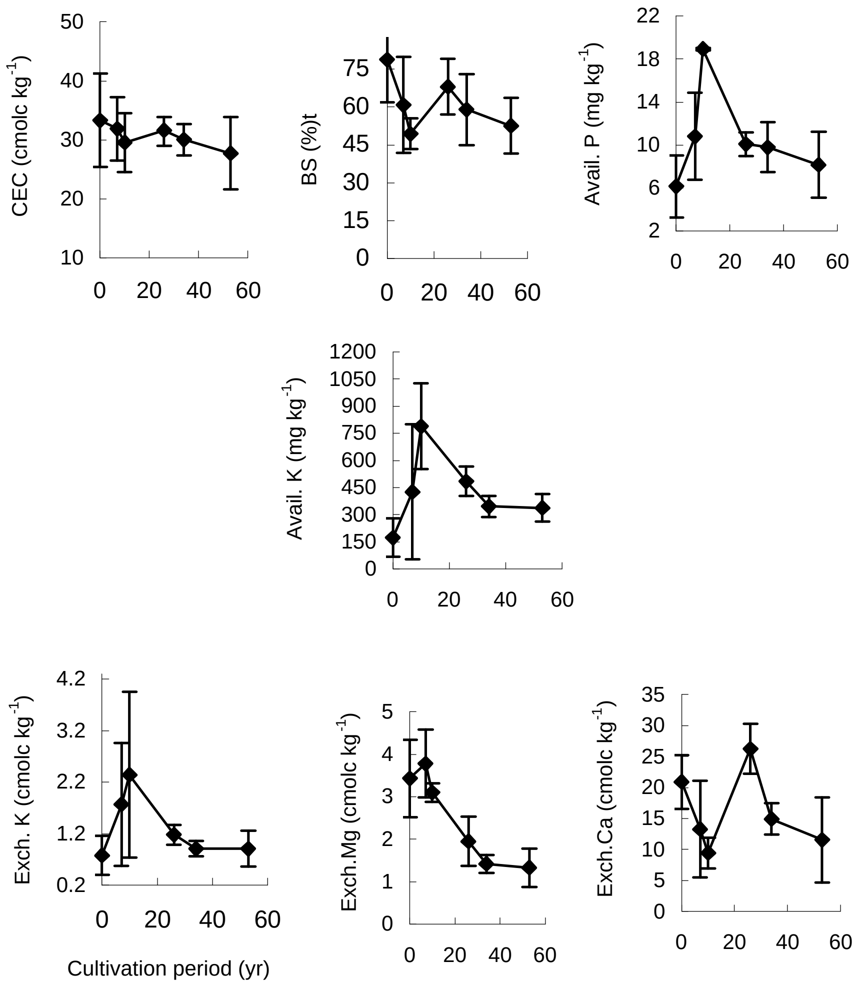

Fig. 6. Changes in mean values of CEC, BS, available P, available K, Exchangeable K, Mg and Ca of Andosols along a chronosequence of farm fields converted from a tropical dry Afromontane forest, Ethiopia (the value 0 along the x-axis denotes the natural forest).

In contrast to the continuous decline in the 0 -10 cm layer, SOC contents in the 10-20 and 20-40 cm soil layers on the farm fields were higher than the corresponding depths under the natural forest. Consequently, the SOC stocks in the upper 40 cm soil did not differ significantly between the farm fields and the natural forest until after 26 years of continuous cultivation (Fig. 7). The higher SOC contents in the sub-soil layers of the farm fields might probably have emerged from SOM incorporation from surface layers to sub-soil layers as a result of the mixing effect of tillage. Furthermore, the substantial amount of organic materials added from root biomass after the slash and burn (Vitousek and Sanford, 1986; van Noordwijk *et al.*, 1997) coupled with the strong organic matter stabilizing nature of Andosols may explain the higher SOM stocks in the sub-soils of the farm fields (Parfitt *et al.*, 1997; van Noordwijk *et al.*, 1997).

Partriculraly, the woody roots left in the below ground after slashing decompose gradually (van Noordwijk *et al*., 1997) and continue enriching the SOM for some time after the forest clearance (Fig. 7).

The overall slow degradation response to deforestation and subsequent cultivation of the soils of the study confirms the results from several other studies carried out on Andosols (e.g. Cotching *et al*., 1979; Saggar *et al*., 1994; Parfitt *et al.*, 1997). The cited studies reported good resistance of Andosols to degradation when subject to land use changes that involve deforestation and subsequent cultivation. For instance, Popenoe, (1960) found a slight reduction of SOC and total N contents following clearing, burning of natural forest and subsequent cultivation, and an increased fertility on the cleared site than natural forest in Guatemala. Krebs *et al*. (1974) and Krebs (1975) also could not find any evidence on Andosol deterioration after comparing soils under natural forest, sugar cane, coffee and pasture following 20 years of cultivation in Costa Rica. Parfitt *et al*., (1997) reported greater stability of organic matter as well as less degradation response of Andosols after 100 years of continuous cropping and pasture land use compared with an Inceptisol exposed to only 20 years of similar uses.

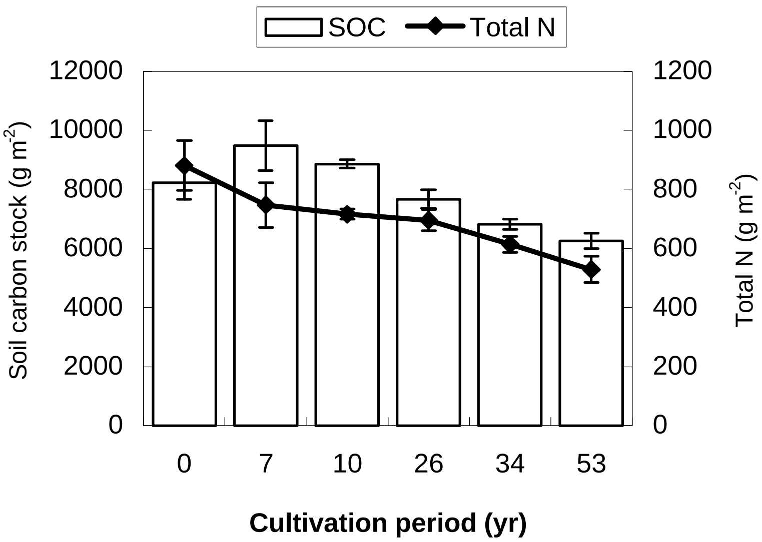

Fig. 7. Changes in soil C and total N stocks (g m^{-2}) in the top 0- 40 cm depth of Andosols along a chronosequence of farm fields converted from a tropical dry Afromontane forest, Ethiopia (the value 0 along the x-axis denotes the natural forest).

The resistance of Andosols to changes in their soil chemical properties is attributed to the resilience of the soil fertility through rapid weathering of poorly crystallised phases in the volcanic ash (Lundgren, 1978), and the nature of the clay particles and amorphous minerals that make up volcanic Andosols. The allophane, imogolite and ferrihydrite clay particles, and the large amorphous Al/Fe oxides that make up volcanic Andosols have substantial exchange surface areas, and therefore adsorb and stabilize organic matter and soil nutrients (Zunino *et al*., 1982; Boudot *et al*., 1989; Mizota and van Reeuwijk, 1989; Saggar *et al*., 1994; Saggar *et al*., 1996). These properties render Andosols the capacity to accumulate, stabilize or protect and gradually mineralise the organic matter and plant nutrients that are accumulated in the soils during the forested period as well as from the slash and burn, thus displaying high fertility for a relatively prolonged period (Paper I).

However, the continuous deterioration of the physical soil attributes could affect the productivity of the soils by altering the hydrological regimes (e.g. infiltration & water-holding capacity), crop rooting depth and soil susceptibility to erosion in the long-term. Despite the initial increases in SOC, the lack of sufficient organic substrate inputs from the farming system is well reflected in the gradual decrease in soil C and total N stocks in the long run (Fig. 7). Similarly, the impacts of continuous nutrient mining owing to restricted external inputs is clearly reflected in the gradual decline of soil properties such as available P and K after their peaks at 10 years (Fig. 6). Therefore, the overall direction of change in the soil condition with span of time under cultivation was degradation, albeit slowly (Paper I). This degradation trend is also reflected in the DIs of the farm fields (Fig. 8). Therefore, to sustain crop production in the long-term in this area, improved soil fertility management should be introduced.

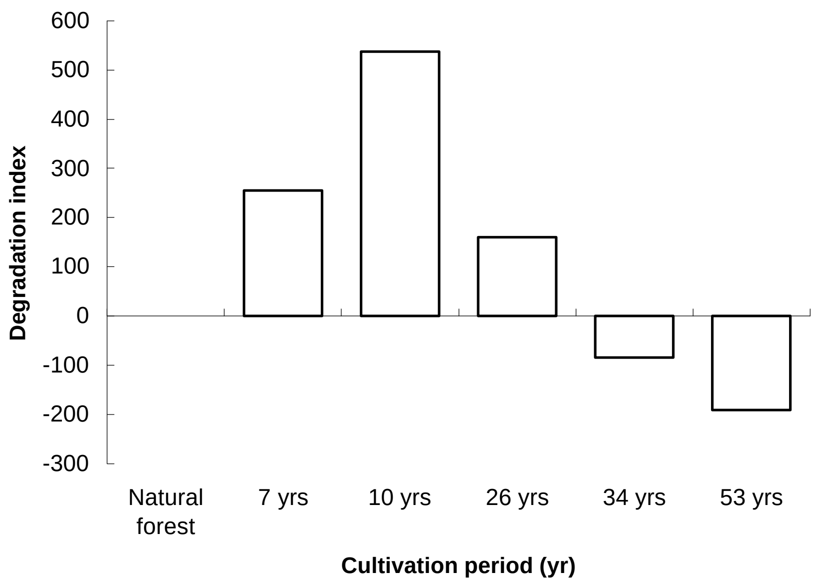

Fig. 8. Degradation index of Andosols along a chronosequence of farm fields with 7, 10, 26, 34 and 53 years of continuous cultivation following conversion from a tropical dry Afromontane forest in Ethiopia. Each deterioration index was calculated as the sum of the percentage deviation of bulk density, pore space, soil C, total N, CEC, BS%, available and exchangeable base cations (except Na) of the upper 0-10 cm soil layer from their respective values under the natural forest.

4.2. Soil organic matter dynamics as shown by the ^{13}C and ^{15}N natural abundance

A key component for sustaining production in low input tropical agricultural systems is the maintenance of a good level of SOM (Sanchez *et al*., 1989; Woomer *et al*., 1994; Kapkiyai *et al*., 1998). A better understanding of SOM dynamics in tropical agricultural systems can give insights into processes that are important in the management of the SOM pool. In this study ^{13}C and ^{15}N natural abundance techniques were used to assess SOM dynamics following deforestation and subsequent cultivation along a chronosequence of farm fields originally converted from natural forest sites (Paper II).

The results showed that $\delta^{13}C$ values of the SOC in the 0-10 cm soil layer of the farm fields increased with the time under cultivation (Fig. 9a). On the other hand, during the 53 years of continuous cultivation, the SOC of the bulk soil declined by 50.4% (equivalent to an average loss of 500 kg C ha^{-1} yr^{-1}), while the SOC fraction of forest origin declined by 74.6% (equivalent to an average loss of 740 kg C ha^{-1} yr^{-1}). The SOC gains from the agricultural crop origin remained very low, only 240 kg C ha^{-1} yr^{-1}, compared with the large loss of the SOC of forest origin. The crop-derived SOC in the surface soil showed a rapid increase during the first 10 years of cultivation after conversion, and remained fairly constant thereafter along the chronosequence (Fig. 9b). This kind of rapid increase in crop-derived SOC during early cultivation periods after deforestation followed by stabilization afterwards has been reported in other similar studies (e.g. Balesdent *et al*., 1988; Collins *et al*., 1999). After 53 years of continuous cultivation following deforestation, the SOC of agricultural origin accounted for 49.5% of the bulk soil SOC content, i.e. slightly less than the amount of SOC contributed from forest origin. The lower contribution to SOC from the agricultural crop compared with that from forest origin even after a relatively prolonged period of time under cultivation is also consistent with several other similar studies (e.g. Balesdent *et al*., 1988; Vitorello *et al*., 1989; Collins *et al*., 1999; Solomon *et al*., 2002). For instance, on a Brazilian Oxisol that had been under sugarcane cultivation for 50 years following deforestation, the amount of SOC derived from the sugarcane crop amounted to 36% of the SOC of the bulk soil (Vitorello *et al*., 1989).

The low contribution of the agricultural crop to the SOC could probably be explained by (i) low incorporation of crop residues from the cropping system owing to residue removal and repeated burning; and (ii) the rapid turnover in the organic substrates derived from crop residue whenever added (McDonagh *et al*., 2001). This low carbon input from the agricultural crop could not compensate for the large mineralization of the forest-derived organic matter on the farm fields. This phenomenon eventually resulted in a progressive decline in the SOC of the bulk soils along the chronosequence (Fig. 9b). The fraction of the SOC attributed to crop origin is mainly that contributed by crop roots and possibly some crop residues that escaped burning. This may imply that crop residue management in the farming system can potentially increase the SOC contribution from the cropping system (Vance, 2000), which may help to reduce the progressive decline in SOC in the soil. Across a range of sites, a strong and positive relationship between return of crop residues (including those with high lignin and/or high C/N ratios) and soil organic matter has been found (e.g. Duff *et al*., 1995; Grace *et al*., 1995), which eventually contributes to sustainable crop yields (Grace *et al*., 1995).

The results also showed that $\delta^{15}N$ consistently increased with cultivation period, coupled with the changes in $\delta^{13}C$. The increased $\delta^{15}N$ values of the SOM on the farm fields would indicate higher nitrogen losses from the agroecosystem compared to the natural forest system (Högberg, 1990; Penuelas *et al*., 1999). This is consistent with the large loss of total N, which amounted to 59%, from the farm field soils during the 53 years of continuous cultivation after deforestation. The lower $\delta^{15}N$ value in the natural forest indicates low activities of N-losing processes, which is due to the relatively closed nutrient cycling and minimal disturbance in the natural forest system (Eshetu, 2000; Eshetu and Högberg, 2000; Eshetu, 2004). The higher $\delta^{15}N$ values on the farm fields could be a result of nitrification in combination with nitrate leaching, denitrification, ammonia volatization in the agricultural system as a result of tillage, aeration, residue burning and temporary waterlogging (Paper II).

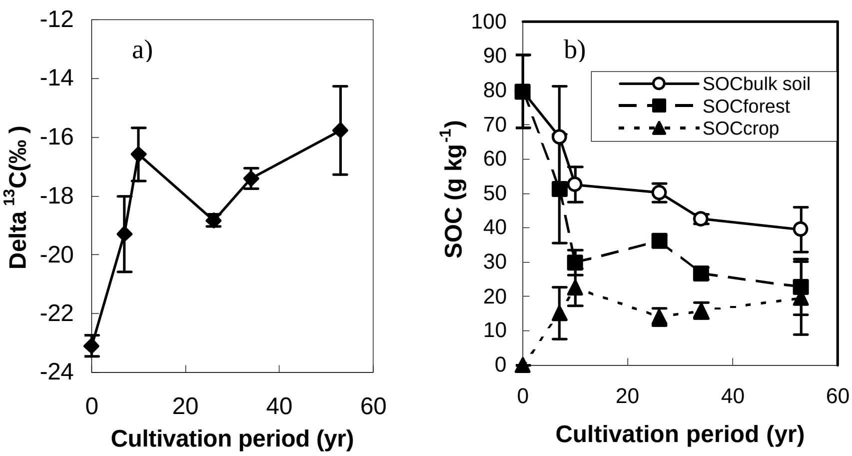

Fig. 9. Effect of deforestation and subsequent cultivation of Andosols on (a) $\delta^{13}C$, and (b) SOC of the bulk soil (SOCbulksoil), forest-derived C (SOCforest) and agricultural crop-derived C (SOCcrop) in the 0 -10 cm soil layer along a farm field chronosequence converted from a tropical dry Afromontane forest in Ethiopia (the year zero along the x-axis denotes natural forest).

The SOC of the bulk soil, the SOC fraction derived from the forest and total N of the 0-10 cm soil layer declined exponentially and attained new apparent equilibria after 20-25 years (Papers I & II). This concurs with results from other studies on cultivation of virgin pastureland (e.g. Grace *et al*., 1995) and natural forest sites (e.g. Dominy *et al*., 2002). The initial rapid loss of large SOC and total N demonstrates the presence of an easily mineralizable organic substrate (labile pool) of forest origin that was exhausted in 25 to 30 years after deforestation. This exhaustion is what has caused the levelling off of these parameters. Generally, the loss of total N from the soils of farm fields was considerable compared with the amount added as fertilizer (Paper II).

Given the parallel and relatively high irregularities along the chronosequence of the farm fields in SOC of the bulk soil, the C: N ratio, the SOC of forest origin and $\delta^{13}C$ are probably worth noting (Paper II). Although this is not possible to ascertain with the data collected for this study, the phenomenon could be attributed to the effect from on-farm trees and their management (see for example Fig. 4). A study in Kenya has shown that carbon inputs from on-farm trees in a parkland agroforestry system can significantly alter $\delta^{13}C$ signature of the SOC of soils taken close to the trees compared to the SOC taken relatively far away from the influence of trees (Nyberg and Högberg, 1995). Parkland tree management such as lopping, pruning, felling, etc. in the farming system of the study area is a continuous process (Fig. 4), which continuously incorporates some organic materials from the trees into the soil system. This is a tree-derived source of SOC different from the carbon originating from the natural forest. This may suggest that the use of natural abundance of ^{13}C techniques to study SOM dynamics under traditional parkland agroforestry system should be exercised with caution.

In general, the results showed that despite the characteristic low tillage and traditional parkland agroforestry system on soils of volcanic origin, subsequent cultivation of soils cleared out of a dry tropical montane natural forest caused a substantial reduction of SOM levels in the top plough layer. This reduction could probably be explained by reduced input of organic residues, a higher soil disturbance as a result of tillage and absence of SOM management. As soil organic matter is the main supplier of soil N, S and P in low input farming systems, a continuous decline in the SOM content of the soils is likely to affect the soil productivity and sustainability in the long-term.

4.3. How sustainable is the farming system?

Sustainable agriculture implies an agricultural system that can be maintained in a steady state over time (Hartemink, 1998; Quimby *et al*., 2002). Soil being a vital component of sustainable agriculture, by monitoring changes in important soil quality indicators such as soil physical and chemical properties the sustainability of an agricultural system can be judged (Greenland, 1975; Hartemink, 1998). Sections 4.1 and 4.2 present evidence of significant changes in the soil quality attributes of the studied soils following deforestation and subsequent cultivation. The overall degradation trend observed, albeit slowly, in the soil quality partly reflects the lack of proper soil management during the cultivation period. In fact, significant changes in soil attributes following deforestation and subsequent cultivation are probably inevitable. The question now is what these changes indicate for the sustainability of crop production in the area, because the observed changes in soil attributes as shown by soil analytical data may not necessarily imply a decline in crop yield, and thus may not reflect a non-sustainable farming system.

The sustainability evaluation showed that despite significant changes, most soil chemical properties did not reach threshold values/critical limits for maize cultivation for the time period considered in this study. Nonetheless, the 34 and 53 year-old fields are close to marginal states for the cultivation of maize (Paper I). According to a classification of sustainable land management presented by FAO (1993b), the soils in this study may be classified as long-term sustainable, i.e. they can be used for at least 25 years before causing soil degradation that limits crop production at the present management level.

However, continuous use of the farm fields for over 25-30 years as is practised in the study area at present does not appear to be sustainable as shown by the increasingly marginal status of the fields older than 30 years for maize production. This was consistent with the DIs computed for the soils of the farm fields (Fig. 8). The cumulative degradation index (DI) values of the 34 and 53 year-old fields were negative, showing that overall soil condition in these farm fields is getting below the condition of the soil under the natural forest. These declines in soil fertility are caused by the subsequent removal of nutrients through crop and residue harvest and leaching coupled with low levels of fertilizer application, which cause negative nutrient balances. As a result of such output-input nutrient imbalances, crop demands for nutrients have to be offset by the soil nutrient pool, a process that leads to degradation. Farmers in the study area, like other farmers in the country, practise low-input agriculture mainly due to economic constraints. Furthermore, the organic matter that could have helped to sustain soil fertility by mineralization, retention of nutrients and enhancing the water-holding capacity of the soils, also declined along the chronosequence due to continuous cropping and residue harvest. This process of declining SOM caused significant impacts not only on the continuous decline in plant nutrient pools such as the pool of total N, but also on the soil physical properties such as bulk density and pore space. This suggests that nutrient mining, i.e. continuous crop harvesting without replenishing soil nutrients, coupled with poor soil management have caused the declining state of the farming system.

When compared to most tropical soils that could serve for agriculture only for few years (e.g. Tiessen *et al*., 1994), the soils of the study area could be claimed to be very resistant to degradation. On the other hand, the overall decline in agricultural productivity in the whole country may not be surprising given the sustainable use of the soils only for 25-30 years when compared with the low input continuous cultivation of most soils in the highlands of Ethiopia for over hundreds of years.

4.4. Tillage intensity and rate of soil degradation

An investigation was also made on the effects of tillage intensity by sampling soils from neighbouring sites subject to traditional low tillage farming (TF) and high intensity mechanized farming (MF) (Paper IV). The results showed a wide range of soil property divergences between the soils subject to the two farming systems. The soil under MF had significantly higher bulk density of 1.07 g cm^{-3} compared to 0.85 g cm^{-3} under the TF and 0.66 g cm^{-3} under the adjacent natural forest in the 0-10 cm layer. In the 10-20 cm layer as well, the soil under MF had significantly higher bulk density of 1.11 g cm^{-3} compared to 0.83 g cm^{-3} under the

TF and 0.73 g cm^{-3} under the natural forest. The changes in bulk density were also associated with changes in soil pore space. The soil pore space for the 0-10 cm layer was 49.1% under the MF and 59.7% under the TF, compared with 65% under the natural forest.

Similarly, the soil under MF exhibited greater total C, N and base cation decreases compared to the soil under TF. The soil under MF had lower values for total C (-8.92 g kg^{-1}), total N (-1.0 g kg^{-1}), CEC (-7.59 $cmol_c$ kg^{-1}), exchangeable Ca (-3.95 $cmol_c$ kg^{-1}), available K (-108.60 mg kg^{-1}) and exchangeable Na (-0.08 $cmol_c$ kg^{-1}) compared to the soil under TF in the 0-10 cm layer. Differences in the subsoil layer followed the same patterns as the surface soil, although over narrow ranges. The exception was the trend followed by available K. The MF site had, however, a higher average (but non-significant) content of available P than the soil under TF.

The large differences in soil properties between the two tillage intensities were attributed to several factors. Firstly, in the MF case site-clearance was conducted with bulldozers that removed most of the litter and vegetation biomass as well as some of the topsoil layer from the site, in contrast to the traditional clearance method where nutrients were returned in ash from slash and burn during the clearance phase. This might have contributed to the initial nutrient pool differences, which may also be reflected in soil properties during the subsequent years (Sanchez *et al*., 1985). Moreover, in the slash and burn system, some trees are selectively left on-farm compared to the complete removal of vegetation by mechanized clearing. The contributions of on-farm trees to the soil ecosystem and nutrient budgets are considerable (e.g. Jiru, 1989), and might have made some contribution to the observed differences in the soil nutrient status between the two farming systems.

The difference in organic matter loss between the two farming system is most likely related to the difference in the intensity of tillage. Tillage and annual ploughing are known to affect soil organic matter mineralization by stirring the soil, disrupting aggregates and increasing aeration. Moreover, tillage intensity was found to correlate well with the rate of soil organic matter turnover. Usually reduced tillage such as zero tillage causes a low impact on soil organic matter (Reicosky and Lindstrom, 1993, Balesdent *et al*., 1990). Similarly, big losses of total C and N on the MF site in the present study were probably related to the increased tillage intensity due to tractor ploughing, which turns over larger and deeper soil materials, disrupting soil aggregates and increasing aeration over a larger soil volume as opposed to the reduced tillage practices of the TF owing to the simple farm implement used. The increased loss of organic matter might also explain the losses of base cations on the MF site because of accelerated leaching and erosion accompanying organic matter losses. On the other hand, the high level of available P found on the MF site compared to the TF site indicated P accumulation due to large phosphate-containing fertilizer additions in the former site.

4.5. Impact of deforestation and subsequent cultivation on soil seed bank

Soil seed banks (SSB) assessment was made to provide information on whether seeds from the original forest could persist after deforestation and subsequent cultivation of different intensities (times) and could contribute to future succession of native woody vegetation after abandonment. Similar to most reports from SSB studies in the temperate (e.g. Bossuyt and Hermy, 2001) and tropical (e.g. Teketay, 1996) regions, it was observed that there was a low overall similarity between the species recovered from SSB samples including those taken from natural forests with the above-ground species in the natural forest (Paper III). In spite of the low viable seeds in the SSB, the study also showed that deforestation and subsequent cultivation were obvious threats to the forest biodiversity not only through the clearance of forest (loss of habitat), but also by deteriorating the SSB of native forest flora. Contributions of woody species (trees and shrubs) to the soil seed flora declined from 5.7% after seven years to nil after 53 years of continuous cultivation.

A second obvious effect from repeated cultivation was that the SSB of the farm fields quickly accumulated seeds of weed species common to open cultivated fields such as *Galinsoga* spp., replacing seed banks of forest flora. Of the germinated flora from the seed bank samples, herbaceous plants accounted for nearly 78% on average (Paper III). Woody species, particularly the native woody species, were represented by less than 10% of the germinants. The density of germinated herbaceous plants was much higher on the farm fields than the natural forests. This observation was consistent with other seed bank studies from the highlands of Ethiopia (Teketay, 1997a & b, 1998) or outside Ethiopia (e.g. Davy *et al*., 1998; Dunsford *et al.*, 1998). In fact, besides anthropogenic disturbance, a combination of factors such as the inherent properties of the seeds (e.g. dormancy and longevity), environmental conditions of the site where the seeds lie and presence of predators can determine the length of time that seeds remain viable in the soil (Granström, 1986; Teketay, 1996). Nevertheless, for a given state of natural conditions, human impacts often accelerate depletion of forest flora SSB, and consequently arrest or retard forest succession following abandonment (Davy, 2002). The strength of human disturbance to affect SSB appears to be related to the duration (and intensity) of land uses (Brown and Lugo, 1990; Teketay, 1998; Honnay *et al*., 1999). This was shown in the progressive decline of the woody species seed composition in the seed bank in this study (Paper III). In a similar study in the UK, 20 years of repeated cultivation removed all vestiges of the seed bank of lowland *Calluna* heath, even though the seeds of the dominant *Calluna vulgaris* can have considerable longevity in undisturbed heathland soils (Davy *et al*., 1998).

Generally, conversion and subsequent cultivation of land previously covered with Afromontane forests not only diminishes soil seed reserves of woody species but also alters the composition of the seed banks. This destruction and alteration of SSB impose challenges in future efforts to recover native forest flora following farm field abandonment. The effect is not only through the limitation of the availability of seeds to initiate the succession of native forest flora, but also through the changed composition that provides an advantage for the competitive weed flora to dominate a site. This ultimately hampers the succession rate of the less competitive native woody species, particularly when left unaided (Harrington, 1999; Lemenih and Teketay, 2004).

The aforementioned impacts of deforestation and repeated cultivation on SSB are worth stressing considering the regeneration ecology of the climax Afromontane forest species. Most climax woody species in the Afromontane forests of Ethiopia are shade-tolerant and also possess seeds that are recalcitrant. Consequently, these species predominantly employ a seedling bank strategy of regeneration, where they accumulate abundant seedlings under the protective canopy of mature forests (Pohjonen, 1989; Teketay, 1996, 1997b). This implies that future successes in restoring native climax species on degraded agricultural lands will rely on fresh seed recruitment on the one hand and a nurse species to become established to provide the required shade for the emerging seedlings of the native climax woody species on the other. Quantity and rate of seed dispersal are in turn affected by several factors, e.g. by the dispersal ecology of the key species and the degree of isolation of the degraded landscape with respect to the seed-donating natural forest community. Unfortunately, long distance dispersal of tropical woody species seeds is rare mainly due to their size and structural limitations (Bakker *et al*., 1996; Lyaruu, 1998; Cubina and Aide, 2001), including woody climax species in the Afromontane forests of Ethiopia (Teketay, 1996; Lemenih and Teketay, 2004). As is the case in Ethiopia, where natural forests have already disappeared significantly, most agricultural landscapes have been increasingly isolated from remnant forest stands. Therefore, restoration of native woody species through natural dispersal will be an extremely slow process if not impossible, and active restoration strategies should be planned rather than relying on natural succession (Lemenih and Teketay, 2004).

An active restoration technique recently receiving considerable attention is the use of plantation forest as a nurse crop (foster ecosystem) to catalyse the recolonization of native woody flora on degraded tropical sites (e.g. Lugo, 1992, 1997; Parrotta, 1992, 1995, 1999). The establishment of plantation forest accelerates vegetation succession by modifying several aspects of a degraded site such as moderation of the microclimate and soil fertility, suppressing competitive grasses and providing habitats or perches for animal and avian seed dispersers (Keenan *et al*., 1997; Parrotta *et al*., 1997; Wunderle, 1997).

4.6. Plantation forests in restoration ecology

4.6.1. Effects on soil attributes (Paper IV)

The results of soil assessment under the plantations of *E. saligna* and *C. lusitanica* showed a positive effect of reforestation of degraded lands on soil attributes, although considerable differences were observed based on the tree species involved (Paper IV). After 15 years of plantation establishment, the soil under the *C. lusitanica* stand in the 0-10 cm layer showed lower soil bulk density, increased soil C, total N, CEC, base saturation (BS), available K, exchangeable K, Ca, and Mg compared to the soils of the MF and TF and the soil planted with *E. saligna*. On the other hand, for the soil planted with *E. saligna* most soil conditions (in the upper 0-10 cm layer) such as pH, total C, total N, BS, CEC, available P, available K, and exchangeable Ca were lower than in the TF situation. However, the overall difference of the soil planted with *E. saligna* with the soil under MF was not very evident. Mean values for some soil attributes in the surface 0-10 cm soil layer showed the following orders:

Bulk density:	natural forest < *Cupressus* < TF < *Eucalyptus* < MF
pH:	MF > *Cupressus* > Natural forest > TF > *Eucalyptus*
Total C:	natural forest > *Cupressus* > TF > *Eucalyptus* > MF
Total N:	natural forest >*Cupressus* > TF > *Eucalyptus* > MF
CEC:	natural forest > *Cupressus* > TF >*Eucalyptus* > MF
BS:	natural forest > *Cupressus* > MF > TF >*Eucalyptus*

There was a marked decrease in soil pH as well as BS under *Eucalyptus* compared to the other sites. This is consistent with several other studies (e.g. Balagopalan *et al*., 1991; Parrotta, 1999). *Eucalyptus* growth form, specifically the oblong conical shape of the canopy, has been alleged to trigger high rates of base cation leaching underneath the canopy, which ultimately leads to low pH (Balagopalan *et al*., 1991). According to Rhoades and Binkley (1996), however, soil acidification results from intense base cation depletion due to storage in biomass by the tree species. Both intrinsic species characteristics and growth rate (biomass accumulation) have been shown in many instances to affect rate of soil nutrient (cation) uptake by trees (e.g. Eriksson, 1996; Alriksson, 1998). Therefore, the low pH and BS under *Eucalyptus* may be exacerbated by the rapid growth rate and large biomass productivity of the species, for instance, as compared with *C. lusitanica* (Paper IV).

The relatively small accumulation of SOC under *Eucalyptus* is also consistent with several other studies (Marry and Sankaran, 1991; Lourzada *et al*., 1997; Garcia-Motiel and Binkley, 1998). Most of these studies attributed the phenomenon to (i) poor nutrient status (high C: N ratio, lignin and tannin contents), and low decomposition of eucalypt litter (e.g. Bernhard-Reversat, 1987; Marry and Sankaran, 1991; Jonsson *et al*., 1996; Lourzada *et al*., 1997), and (ii) poor decomposition-facilitating environment in *Eucalyptus* stands (Kardell *et al*., 1986; Bi *et al*., 1992; Grove *et al*., 2001). For instance, Jonsson *et al.* (1996) reported a little contribution of litter from *Eucalyptus* spp. to soil C but a significant lowering of pH compared with several other species in Tanzania.

So far, most differences in soil attributes between the sites in the present study are confined to the topsoil layer. Differences in soil attributes between the plantations, the farm fields and the natural forest in the 10-20 cm and 20-40 cm subsoil layers were not evident. This is also consistent with observations from several other studies where soil attributes have been assessed following reforestation/afforestation of former arable lands (e.g. Garten, 2002; Paul *et al*., 2003).

From the results obtained here, two important points emerged. The direction and magnitude of changes in soil attributes under the plantations were species-dependent, and reference ecosystem-dependent. The soil attributes under *C. lusitanica* showed an overall change towards the direction of the soil attributes under the adjacent natural forest compared to the soil attributes under the two farming situations. For instance, in the 0-10 cm layer the soil under *C. lusitanica* showed changes in bulk density (-0.25 g cm^{-3} and 0 g cm^{-3}), soil C (+21.9 g kg^{-1} and +13 g kg^{-1}), total N (+2.02 g kg^{-1} and +1.02 g kg^{-1}), CEC (+10.04 $cmol_c$ kg^{-1} and +2.45 $cmol_c$ kg^{-1}) and BS (+2.78% and +7.08 %) compared with the soils of the MF and TF situations respectively. On the other hand, the soil attributes under *E. saligna* diverged away from the soil attributes under the natural forest, and in some instances were even poorer than the soils subject to continuous farming. For instance, in the 0-10 cm soil layer the soil under *E. saligna* showed divergence of the magnitudes for pH (-1.1 and -0.8), BS (-16.24% and -23.32%) and CEC (-6.74 $cmol_c$ kg^{-1} and -1.28 $cmol_c$ kg^{-1}) compared to the soils of the MF and TF situations respectively.

Species-dependent changes in soil attributes have been reported from several studies (Binkely and Sollins, 1990; Parrota, 1992; Brown and Lugo, 1994; Smith, 1994; Fisher, 1995; Garcia-Montiel and Binkley, 1998; Montagnini, 2001). Differential impacts on soil attributes between plantation species may emerge from differences in nutrient recycling capacity (i.e. different quantity and quality of above- and below-ground litter inputs) and nutrient use efficiency (i.e. biomass production and nutrient immobilization in the biomass) of the respective species (Cuevas and Lugo, 1998; Montagnini, 2001). Therefore, proper selection of plantation species on the basis of long-term knowledge about their performance, economic and environmental benefits is a very important element of silvicultural decision in restoration of degraded sites with the ehlp of reforestation/afforestation.

Furthermore, the results from this study showed that the conclusion on whether soil fertility declined or improved under plantation forests differed considerably depending on the reference ecosystem used. For instance, the differences in several soil attributes between the soil planted with *E. saligna* and the soil of MF situation were not evident, while the differences with the soil of the TF situation were very clear (Paper IV). So *E. saligna* can be judged as unfit for soil improvement compared with the soil subject to the TF situation, where the reduced tillage coupled with the relatively low annual crop harvest of the farming system did not cause large soil degradation. Conversely, *E. saligna* can be neutral or even be accepted for restoration compared with the MF situation, where high intensity tillage soil disturbance and relatively large annual harvests have caused high soil degradation. Other reports have also shown that the direction of soil attribute changes under *Eucalyptus spp.* depends on the soil status of the starting ecosystem (Second Citizens Report, 1985; Hailu, 2002). When planted on degraded lands *Eucalyptus* can have positive effects, whereas when grown on newly cleared forest sites the effects are reported to be adverse (Second Citizens Report, 1985).

Generally, the results from this study confirm recommendations of plantation forestry as a facilitator of soil fertility restoration at degraded tropical sites (e.g. Lugo and Brown, 1990; Fisher, 1995; Montagnini, 2001). The positive changes in soil attributes under the plantation stand of *C. lusitanica* compared with the soils under continuous farming situations can be explained by (i) higher inputs of organic substrates from the plantations than from the cropping system; (ii) reduced decomposition of both newly added C and the old soil C owing to lower soil disturbance in the plantations than the farm fields and changed microclimate; and (iii) reduced frequency of harvest-related losses because of the long rotation periods for the plantations compared to the agricultural crops. Nevertheless, most soil attributes (particularly soil C and total N) under the plantations, even for *C. lusitanica,* which had a high positive impact on soil fertility, are still lower than their values in the soil of the adjacent natural forest. This means that 15-17 years are not long enough for the plantation stands to offset the soil fertility lost due to deforestation and cultivation in the area. In fact, several authors have stressed that time since reforestation/afforestation has a significant effect on the magnitude of soil fertility improvement (Sanchez *et al*., 1985; Trouve *et al*., 1994; Bhojvaid and Timmer, 1998; Paul *et al*., 2003). According to Sanchez *et al*. (1985) and Bhojvaid and Timmer (1998), three distinct stages of soil development can be recognized following plantation establishment: (i) an initial establishment phase (0-5 years) characterized by either nominal soil changes or even a decline in soil properties; (ii) a brief transitional phase (5-7 years) characterized by a canopy closure of the tree plantations and a rapid change in soil properties; and (iii) fallow enrichment phase (7-30 years) characterized by a gradual stabilization of soil properties. Similarly, several studies that assessed change in soil C stocks following afforestation and reforestation of former arable soil showed loss of soil C at early stages of plantation development (< 10 years) as there is relatively little input of C from biomass. However, this trend gradually improves as the plantation matures to a phase where C continues to accumulate (Trouve *et al*., 1994; Post and Kwon, 2000; Paul *et al*., 2002). Therefore, the relatively young age since the establishment of the plantations studied may explain why their soil fertility status remained lower than that of the natural forest soil.

4.6.2.Effects on recolonization of native woody flora (Paper V)

The empirical evidence from this study showed that diverse native woody species recolonize underneath plantation stands (Fig. 10). About 33 woody species were recorded from under the plantation stands with density ranging from 3,600-6,280 stems ha^{-1} (Paper V). This is consistent with several other studies from outside Ethiopia (e.g. Parrotta, 1995; Lugo, 1997; Parrotta *et al*., 1997; Loumeto and Huttel, 1997; Harrington and Ewel, 1997; Chen *et al*., 2003) and in Ethiopia (e.g. Yirdaw, 2001; Senbeta and Teketay, 2001; Senbeta *et al.*, 2002; Yirdaw and Luukkanen, 2003). Like most other similar studies, the present study showed marked differences in the density, diversity and sizes of colonizing native woody species under the different plantation species. Difference in canopy density coupled with canopy-influenced variations in understory environmental factors in the plantation stands were found to be responsible for much of the variations in the regeneration parameters assessed (Paper V). Plantation species with lighter canopies (i.e. low CCP or LAI) had higher understory mean diurnal air and soil temperature and higher diurnal air and soil temperature fluctuations than those with heavier canopies. Those species with lighter canopies also advanced greater diversity and density of naturally regenerated native woody species, as well as vigorous DBH and height than species with denser/heavier canopies (Fig. 10). For instance, the dense canopy species *C. lusitanica* (CCP = 94.2±1.74 or LAI = 3.84±0.15) had the lowest density of naturally regenerated native woody species compared to the open canopy species *C. africana* (CCP= 54.0±12.95 or LAI = 0.61±0.25). Sizes (diameter and height) of the regenerates were also larger under the light canopy species of *C. africana* followed by the *E. saligna* and *P. patula*, whereas no naturally regenerating native woody species were recorded with sizes over one cm DBH in the plots surveyed under the dense canopy stand of *C. lusitanica* (Paper V).

Fig. 10. Contrasting density and sizes of naturally regenerating native woody species under three plantation species in southern Ethiopia. Top left is *C. africana*, top right *E. saligna* and bottom centre *C. lusitanica* plantations.

These results demonstrated that the degree of shade by canopy density, rather than the tree species per se, is of overriding importance in determining the type, abundance and particularly the size of plant species that can exist in the understory of plantation forests. The findings confirm results from several other similar studies from the tropical (e.g. Ashton *et al*., 1998; Otsamo, 1998, 2000) and temperate regions (e.g. Hill, 1979, 1987; Hill and Wallace 1989; Cannell, 1999). A study in the temperate regions showed that a ground flora of vascular plants is eliminated once the forests intercept 80–90% of the incoming radiation (Hill 1979), but if light interception is kept to 80% or less, a similar ground flora can develop in forests with different overstory tree species grown under similar soils and climate (Hill, 1979; Hill and Wallace 1989; Cannell, 1999). Coniferous plantations can also have ground floras similar to those of broadleaved plantations, provided the conifers are thinned or consist of species that cast light shade (Hill, 1987; Hill and Wallace, 1989). The same effect of natural and artificial gaps has been witnessed in plantation forests in the tropics (Ashton *et al*., 1998; Otsamo, 1998, 2000; Yirdaw and Luukanen, 2003).

Generally, broadleaved species that have open crowns can bring forth dense and more vigorous regenerations than conifers (Paper V). Nonetheless, there are sufficient plantation management options available to make most plantation landscapes the homes of a rich diversity of flora and fauna, regardless of their intrinsic nature (Cannell, 1999). Management techniques such as planting density (spacing), thinning and pruning (Otsamo, 1998) can help to manipulate canopy characteristics and modify understory light conditions in any plantation settings. By doing so, optimum canopy opening can be obtained for optimum possible penetration and absorption of light, which induces rich plant diversity at the plantation forest floors.

5. Conclusions

5.1. Deforestation, soil fertility degradation and sustainable agriculture in Ethiopia

The results of this study showed that fertile Andosol forest soils cleared for crop production in the tropical dry Afromontane forest region of Ethiopia lose productivity within 25 to 30 years of subsequent cultivation after deforestation. Major declines were observed for soil organic matter, which is the principal source of plant nutrients such as N in low input tropical farming systems like the one investigated. Despite the clear decline in soil fertility within just 25 to 30 years, farmers continue to cultivate the lands for a prolonged time under low input systems due to demographic and economic pressures. This process of prolonged use with low inputs has exacerbated soil quality decline leading to soil degradation, which may ultimately lead to land abandonment.

The 25 to 30 years of sustainable use of the deforested sites is relatively long compared to most reports from similar studies in the tropics. This longer use of the studied soils compared to most other tropical sites is attributed to the nature of the studied soils, i.e., to their Andic properties and partially also to the nature of the farming system. In other highly weathered tropical soil groups found in the country, the sustainable practice of agriculture after deforestation may be expected to be shorter and/or the degradation process worse than that observed for Andosols in the present study.

Furthermore, the depletion of the natural forest ecosystems in Ethiopia, as only less than 3% of the country’s land area is covered with natural forests today, implies that horizontal expansion of croplands to fertile forest sites is no longer an option. Therefore, strategies to feed the expanding population in the country will have to seek a sustainable solution that better addresses soil management. More research on nutrient management with indigenous (e.g. traditional agroforestry, composting, crop rotation, biomass transfer, etc.) improved (e.g.

chemical and organic fertiliers, improved fallows, etc.) techniques should be integrated into a strategy for sustainable agricultural development in the country. In addition, improvement in the management of the soil resources for sustainable agricultural use would be one of the most useful strategies that could help to protect the remnant patches of the tropical dry Afromontane forest and their biological diversity from agricultural land expansion related threat in Ethiopia.

5.2. Impact of deforestation and repeated cultivation on soil seed bank: implication for biodiversity conservation

Ethiopian forests that host rich biodiversity are rapidly disappearing due to high human pressure and a weaker economy, which trigger agricultural land expansion at the expense of forest. As shown in this study, agricultural land expansion and intensification threatens the native forest flora not only through the outright destruction of the forest but also through the impact on the soil seed bank; the future of the forest flora. Besides the depletion of the soil seed banks, vegetation recovery after intensive agricultural land use is hampered due to the depletion of other resources needed for natural regeneration such as soil fertility, inhospitable abiotic (e.g. light, temperature, water, moisture, pH, or nutrient availability) and biotic site conditions (dominance of competitive grasses) at degraded agricultural sites, and lack of sufficient seed dispersal due to increasing isolation from nearby natural forests. This will ultimately lead to permanent destruction of the native forest flora. Therefore, there is urgent need for the protection and management of the few remnant patches of natural forests and their biodiversity in Ethiopia from the threat of agricultural land expansion.

5.3. Plantation forests in restoration ecology

This study showed that restoration of soil attributes and native forest flora on degraded sites in Ethiopia can be fostered with the help of fast-growing tree plantations. However, it was also observed that considerable differences exist between the plantation tree species involved both in fostering the regeneration of native woody species and restoring soil attributes. Therefore, one of the most important silvicultural precautions in using plantation forestry for ecological restoration is the decision on which species to use. The choice of species needs careful consideration and should be based on knowledge of the species' effects on soil attributes and local biodiversity. In terms of soil fertility, knowledge of tree species nutrient use efficiency and recycling capacities (litter production, root turnover, litter decomposition, etc) may be required. With regard to fostering native flora, tree characteristics such as branching habit, leaf orientation, leaf density, freedom from allelopathic potential and the like will be variables worth screening for. Unfortunately, species that may be good in terms of soil fertility may not necessarily be good for fostering biological diversity, as shown for *C. lusitanica* in this thesis. This calls for careful planning in species selection based on the target of the restoration programme. In fact, there are normally sufficient plantation management options available such as thinning, spacing, pruning, species mixing and others to make most plantation ecosystems homes for a rich local flora and fauna.

On the other hand, the ability to successfully restore the degraded lands in Ethiopia will contribute significantly to the envisaged agriculture-based sustainable economic and rural development. Furthermore in Ethiopia, with its predominantly rural structure, weak economy and very high reliance on natural resources for livelihood, particularly in the rural areas, but with its very limited forest resources, restoration activities using plantation forest may have several advantages. Some of these advantages may include (i) supplying forest products, particularly fuelwood and construction wood; (ii) augmenting rural income and national economy by making productive use of the degraded lands; (iii) slowing down the increasing pressure on the meagre natural forests that have high significance for their biodiversity; (iv) supplying additional goods and services such as fodder and forest grazing for the large livestock population in the country, fruits and other wild foods for the society that will contribute to food security; (v) reducing the erosive power of the torrential monsoon climate, regulating stream flows and restoring dried-up streams to secure water demands of the nation; and (vi) enhancing the greenness and improving the recreational values and panorama of the degraded landscapes and making them attractive and hospitable for the community.

References

Abebe, M. 1998. *Nature and Management of Ethiopian Soils*. Alemaya University of Agriculture, I.L.R.I., Addis Ababa, Ethiopia.

Adejuwon, J.O. and Ekanade, O., 1988. A comparison of soil properties under different land use types in a part of the Nigerian cocoa belt. *Catena* 15: 319-331.

African Development Bank (AfDB), 1997. Country environmental profile: Ethiopia, Final report. Environment and Sustainable Development Unit, African Development Bank. pp. 65. Available at http://www.afdb.org/african_countries/pdf/cep-et.pdf. Web page viewed in October, 2003.

Allen, E. B. 1995. Restoration ecology: limits and possibilities in arid and semiarid lands. In: Roundy, B..A., McArthur, E.D., Haley, J.S. and Mann, D.K. (eds). *Proceedings of the Wildland Shrub and Arid Land Restoration Symposium,* Las Vegas, NV, October 19-21, 1993. General Technical Report INT-GTR-315. US Department of Agriculture, Forest Service, Ogden, UT, pp. 7-15.

Alriksson, A. 1998. *Afforestation of Farm Field Soil Changes and the Uptake of Heavy Metals and Nutrients by Trees*. Doctoral thesis, Swedish University of Agricultural Sciences, Department of Forest Soils, Uppsala, Sweden.

Anonymous, 1990. Munessa-Shashamane State Forest Project Management Plan. Forest management division /SFCDD/NRCDM/MOA, Addis Ababa, Ethiopia.

Arrouays, D. Balesdent, J. Mariotti, A. and Girardin, C. 1995. Modelling organic carbon turnover in cleared temperate forest soils converted to maize cropping by using C-13 natural abundance estimates. *Plant and Soil* 173: 191-196.

Arshad, M.A. and Martin, S., 2002. Identifying critical limits for soil quality indicators in agro-ecosystem. *Agriculture, Ecosystem & Environment* 88: 153 –160.

Ashton P.M.S., Gamage S., Gunatilleke I.A.U.N., and Gunatilleke C.V.S. 1998. Using Caribbean pine to establish a mixed plantations: testing effects of pine canopy removal

on plantings of rain forest tree species. *Forest Ecolology and Management* 106: 211-222.
Ashton, M.A., Gunatilleke C.V.S, Singhakumara, B.M.P. and Gunatilleke I.A.U.N. 2001. Restoration pathways for rain forest in southwest Sri Lanka: a review of concepts and models. *Forest Ecolology and Management* 154: 409-430.
Ayoub, A.T. 1998. Extent, severity and causative factors of land degradation in the Sudan. *Journal of Arid Environments* 38: 397- 409.
Bakker, J.P. 1989. *Nature Management by Grazing and Cutting.* Kluwer, Dordrecht.
Bakker, J.P., Poschlod, P., Strykstra, R.J., Bekker, R.M., and Thompson, K. 1996. Seed banks and seed dispersal: important topics in restoration ecology. *Acta Botanica Neert*. 45: 461-490.
Balagopalan, M., Thomas, T. P., Mary, M. V., Sonkar, S., and Alexander, T. G., 1991. Soil properties in Teak, Bombax and Eucalyptus plantations of Trichur forest division, Kerala. *Journal Tropical Forest Science* 5: 35-43.
Balesdent J, Wagner G H and Mariotti A 1988. Soil organic matter turnover in long-term field experiments as revealed by Carbon-13 natural abundance. *Soil Science Society of America Journal* 52: 118-124.
Balesdent, J., Mariotti, A. and Biosgontier, D., 1990. Effect of tillage on soil organic carbon mineralization estimated from ^{13}C abundance in maize fields. *Journal of Soil Science* 41: 587-596.
Bekele, M. 2003. *Forest Property Rights, The Role of The State, and Institutional Exigency: The Ethiopian Experience*. Doctoral thesis, Swedish University of Agricultural Sciences, Department of Rural Development Studies, Uppsala, Sweden.
Bekele, W. 2003. *Economics of Soil and Water Conservation. Theory and Empirical Application to Subsistence Farming in the Eastern Ethiopian Highlands*. Doctoral thesis (AGRERIA 404), Swedish University of Agricultural Sciences, Department of Economics, Uppsala, Sweden.
Bell, S. and Morse, S. 1999. *Sustainability Indicators. Measuring the Immeasurable. Earthscan Publishers*. London.
Berg, B. 2000. Litter decomposition and organic matter turnover in northern forest soils. *Forest Ecolology and Management* 133: 13-22.
Bernhard-Reversat, F. (ed.). 2001. *Effect of Exotic Tree Plantations on Plant Diversity and Biological Soil Fertility in the Congo Savanna: with Special Reference to Eucalypts*. Center for International Forestry Research, Bogor, Indonesia.
Bernhard-Reversat, F. 1987. Dynamics of litter and organic matter at the soil-litter interface in fast-growing tree plantations on sandy ferrallitic soil (Congo). *Acta Oecologia* 14: 179-195.
Bhojvaid, P. P. and Timmer, V. P., 1998. Soil dynamics in an age sequence of *Prosopis juliflora* planted for sodic soil restoration in India. *Forest Ecolology and Management* 106: 181-193.
Bi, H. Turvey, N.D. and Heinrich, P. 1992. Rooting density and tree size of *Pinus radiata* (D. Don) in response to competition from *Eucalyptus oblique* (L'Herit). *Forest Ecolology and Management* 49: 31-42.
Binkley, D. and Sollins, P. 1990. Factors determining differences in soil pH in adjacent conifer and alder-conifer stands. *Soil Science Society of America Journal* 54: 1427-1433
Black, C.A. Evans, D.D., White, J.L., Ensminger, L.E. and Clark, F.E. 1965. *Methods of Soil Analysis. Part 1. Physical and Mineralogical Properties, Including Statistics of Measurement and Sampling.* Am. Soc. Agro., Inc., Madison, Wisconsin.
Bojo J., and Cassels, D. 1995. Land degradation and rehabilitation in Ethiopia: a reassessment. AFTES Working paper No. 17. World Bank, Washington DC.
Bolinder, M.A., Angers, D.A. Gregorich, E.G. and Carter, M.R. 1999. The response of soil quality indicators to conservation management. *Canandian Journal of Soil Science* 79: 37-45.
Bossuyt B. and Hermy M. 2001. Influence of land use history on seed banks in European temperate forest ecosystems: a review. *Ecography* 24: 225-238.
Bossuyt, B. Deckers, J and Hermy, M. 1999. Migration of herbaceous plant species across ancient-recent forest ecotones in central Belgium. *Journal of Ecology* 87: 628-638.

Boudot, J.P., Bel Haje. B.A., Steiman, R. and Seigle-Murandi, F., 1989. Biodegradation of synthetic organo-metallic complexes of iron and aluminium with selected metal to carbon ratios. *Soil Biology & Biochemistry* 21: 961-966.

Bouma, J. 1994. Sustainable land use as a future focus for Pedology (a guest editorial). *Soil Science Society of America Journal* 58: 645-646.

Bouma, J. 2002. Land quality indicators of sustainable land management across scales. *Agriculture, Ecosystem & Environment* 88: 129-136.

Bradshaw, A.D. 1987. The reclamation of derelict land and the ecology of ecosystems. In: Jordan, W.R. Gilpin, M.E. and Aber, J.D. (eds). *Restoration Ecology*. Cambridge University Press, Cambridge. Pp. 53-74.

Bradshaw, A.D. 1997. What do we mean by restoration? In: Urbanska, K.M., Webb N.R. and Edwards P.J. (eds). *Restoration Ecology and Sustainable Development*. Cambridge University Press, Cambridge. pp. 8-16.

Bradshaw, A.D. 2002. Introduction and philosophy. In: Perrow, M.R. and Davy, A. J. (eds). *Handbook of Ecological Restoration: Vol. 1. Principles of Restoration*. Cambridge University Press, Cambridge. pp. 3-9.

Brady, N. C. and Weil, R. R. (Eds). 2002. *The Nature and Properties of Soils*, 13th Edition**.** Prentice Hall, NJ.

Brady, N.C. 1994. Alternative to Slash-and Burn: a global imperative. In: Sanchez P.A. and van Houten H. (eds). *Alternative to Slash-to-Burn Agriculture.* Symposium ID-6. Vol. 9, Supplement to Transactions. Proceedings of the Transactions of 15th World Congress of soil Science, July 10-16, Acapulco, Mexico. INEGI and CAN (Mexico).

Brokaw, N.V.L. 1985. Tree falls, regrowth, and community structure in tropical forests. In: *The Ecology of Natural Disturbance and Patch Dynamics*. pp 53-69 (Pickett, S.T.A. and P.S. White, eds.). Academic Press, New York.

Brown, L.H. and Cocheme, J. 1969. A study of the agroclimatology of the highlands of Eastern Africa. FAO/UNESCO/WMO.

Brown, S. and Lugo, A.E. 1990. Tropical secondary forests. *Journal of Tropical Ecology* 6: 1-32.

Brown, S. and Lugo, A.E. 1994. Rehabilitation of tropical lands: a key to sustaining development. *Restoration Ecology* 2: 97-111.

Brundtland, G.H. 1987. *Our Common Future.* Report of the World Commission on Environment and Development. Oxford University Press. Oxford. 398pp.

Bulygin, S.Y. and Lisetskty, F.N. 1991. Soil microaggregation as an index of erosion resistance. *Pochvovedeniye* 12: 98-104.

Cairns, J. Jr. 2002. Rationale for restoration. In: Perrow, M.R. and Davy, A. J. (eds). *Handbook of Ecological Restoration: Vol. 1. Principles of Restoration*. Cambridge University Press, Cambridge. pp. 10-23.

Campbell, J. 1991. 'Land or peasants? The Dilemma Confronting Ethiopian Resource Conservation'. *African Affairs* 90: 5-21.

Cannell, M.G.R. 1999. Environmental impacts of forest monocultures: water use, acidification, wildlife conservation, and carbon storage. *New Forests* 17: 239-262.

Chamshama, S.A.O. and Nduwayezu, J.B. 2003. Rehabilitation of degraded sub-humid lands in Sub-Saharan Africa: A synthesis. Background paper presented for VITRI/ETFREN/UFRO-SPDC workshop, Trees, Agro-forestry and climate change in dryland Africa (TACCDA), Hyytiälä, Finland, 29 June- 4 July 2003.

Chapman, H.D. 1965. Cation exchange capacity. In: Black *et al*., C.A. (ed). *Methods of Soil Analysis*. Agronomy 9: 891-901. Am. Soc. Agro., Inc., Madison, Wisconsin.

Chapman, J.D. and White, F. 1970. *The Evergreen Forests of Malawi.* Comm. For. Inst., University of Oxford.

Chatterson, M.T., Gulick, F.A. and Resch, T. 1989. Desertification-rethinking forestry strategy in Africa: Experience drawn from USAID activities. In: *Role of Forestry in Combating Desertification.* FAO Conservation Guide No. 21: 91-102. FAO: Rome.

Chen, X., Li, B-L. and Lin, Z-S. 2003. The acceleration of succession for the restoration of the mixed-broadleaved Korean pine forests in Northeast China. *Forest Ecolology and Management* 177: 503-514.

Cohen, A.L.B., Singhakumara, M.P. and Ashton, P.M.S. 1995. Releasing rain forest succession: a case study in the *Dicranopteris linearis* fernlands of Sri Lanka. *Restoration Ecology* **3**: 261–270.

Collins, H.P., Blevins R.L., Bundy L.G., Christenson, D.R., Dick, W.A., Huggins, D.R. and Paul, E.A. 1999. Soil carbon dynamics in con-based agroecosystems: results from carbon-13 natural abundance. *Soil Science Society of America Journal* 63: 584-591.

Compton, E.J. Boone, D.R. Motzkin, G. and Foster, R.D. 1998. Soil carbon and nitrogen in a pine-oak plain in central Massachusetts: Role of vegetation and land-use history. *Oecologia* 116: 536-542.

Constable, M. and Belshaw, D., 1989. The Ethiopian highlands reclamation study (EHRS): major findings and recommendations. In: *Proceedings of the National Workshop on Food Strategies for Ethiopia*, Addis Ababa. pp. 142-179.

Cotching, W.E., Allbrook, R.F. and Gibbs, H.S. 1979. Influence of maize cropping on the soil structure of two soils in the Waikato district, New Zealand. *New Zealand Journal of Agricultural Research* 22: 431-438.

CSA (Central Statistical Authority), 1999. Statistical Abstract (1998). Addis Ababa.

Cubina, A. and Aide, M.T. 2001. The effects of distance from forest age on seed rain and soil seed bank in a tropical pasture. *Biotropica* 33: 260-267.

Cuevas, E. and Lugo, A.E. 1998. Dynamics of organic matter and nutrient return from litterfall in stands of ten tropical tree plantation species. *Forest Ecolology and Management* 112: 263-279.

Datta, A. 1998. Hornbill abundance in unlogged forest, selectively logged forest and a forest plantation in Arunachal Pradesh, India. *ORYX* 32: 285-294.

Davy, A.J. 2002. Establishment and manipulation of plant populations and communities in terrestrial systems. . In: Perrow, M.R. and Davy A. J. (eds). *Handbook of Ecological Restoration*. *Vol. 1. Principles of Restoration*. Cambridge University Press, Cambridge. pp. 223 –241.

Davy, A.J., Dunsford, J.J. and Free, A.J. 1998. Acidifying peat as an aid to the reconstruction of lowland heath on arable soil: lysimeter experiments. *Journal of Applied Ecology* 35: 649-659.

Decher, J. and Bahain, L.K. 1999. Diversity and structure of terrestrial small mammal communities in different vegetation types on the Accra Plains of Ghana. *Journal of Zoology* 247: 395–408.

Dobson, P.A., Bradshaw, A. D. and Baker, A.J.M. 1997. Hopes for the Future: Restoration ecology and conservation biology. *Science* 277: 515-522.

Dominy, C. S., Haynes, R. J. and van Antwerpen. R. 2002 Loss of soil organic matter and related soil properties under long-term sugarcane production on two contrasting soils. *Biology & Fertility of Soils* 36: 350–356

Doran, J.W. and Parkin, T.B. 1994. Defining and assessing soil quality. In: Doran, J.W. Coleman, D.C., Bexdicek, D.F. and Stewart, B.A. (eds). *Defining Soil Quality for a Sustainable Environment.* SSSA Special Publication No. 35. Soil Sci. Soc. America and Am. Soc. Of Agro., Madison, WI, pp. 3-21.

Doran, J.W. and Zeiss, M.R. 2000. Soil health and sustainability: managing biotic component of soil quality. *Applied Soil Ecology* 15: 3-11.

Doran, J.W., 2002. Soil health and global sustainability: translating science into practice. *Agriculture, Ecosystem & Environment* 88: 119-127.

Duff, B. Rasmussen, P.E. and Smiley, R.W. 1995. Wheat/fallow systems in semi-arid regions of the Pacific NW America. In: Barnett, V. Payne, R., Steiner, R. (eds), *Agricultural Sustainability: Economic, Environmental and Statistical Considerations*. Wiley, New York, pp. 85-109.

Dumanski, J. and Smyth, A.J. 1993. The issues and challenges of sustainable land management. *International Workshop on Sustainable Land Management for the 21st Century,* University of Lethbridge, Alberta, Canada.

Dumanski, J. Eswaran, H. and Latham, M. 1991. Criteria for an international framework for evaluating sustainable land management. Paper presented at IBSRAM International Workshop on Evaluation for Sustainable Development in the Developing World. Chiang Rai, Thailand.

Duncan, R. S. and Duncan, E.V. 2000. Forest succession and distance from Forest edge in an Afro-Tropical Grassland. *Biotropica* 32: 33-41.

Duncan, R.S. and Chapman, C.A. 2003. Consequences of plantation harvest during tropical forest restoration in Uganda. *Forest Ecolology and Management* 173: 235-250.

Duncan, S..R. and Chapman A.C. 1999. Seed dispersal and potential forest succession in abandoned agricultural in tropical Africa. *Ecological Applications* 9: 998–1008

Dunsford, S.J., Free, A.J. and Davy, A.J. 1998. Acidifying peat as an aid to the reconstruction of lowland heath on arable soil: a field experiment. *Journal of Applied Ecology* 35: 660-672.

EFAP (Ethiopian Forestry Action Program), 1994. Ethiopian Forestry Action Program, Vol. (2). The Challenge for Development. Ministry of Natural Resources Development and Environmental Protection, Addis Ababa.

Eriksson, H. M. 1996. *Effects of Tree Species and Nutrient Application on Distribution and Budgets of Base Cations in Swedish Forest Ecosystems*. Doctoral thesis, Swedish University of Agricultural Sciences, Uppsala, Sweden.

Eriksson, I., Teketay, D. and Granström, A. 2003. Response of plant communities to fire in an acacia woodland and a dry Afromontane forest, southern Ethiopia. *Forest Ecolology and Management* 177: 39-50.

Eshetu, Z. 2000. *Forest Soils of Ethiopian Highlands: Their Characteristics in Relation to Site History. Studies Based on Stable Isotopes.* Doctoral thesis, Swedish University of Agricultural Sciences, Department of Forest Ecology, Umeå.

Eshetu, Z. 2004. Natural ^{15}N abundance in soils under young-growth forests in Ethiopia. *Forest Ecolology and Management* 187, 139-147.

Eshetu, Z. and Högberg, P. 2000. Effects of land use on ^{15}N natural abundance of soils in Ethiopian highlands. *Plant and Soil* 222: 109-117.

Ewel, J.J. 1985. Designing agricultural ecosystems for the humid tropics. *Annu. Rev. Ecol. Syst*. 17: 245-271.

Eyasu, E. 2002. *Farmers' Perceptions of Soil Fertility Change and Management.* SOS-Sahel and Institute for Sustainable Development, Addis Ababa, Ethiopia.

Eyasu, E. and Soones, I. 1999. Perspectives on soil fertility change. A case study from southern Ethiopia. *Land degradation & development* 10: 195-206.

Eyasu, E. Morse, S. and Belshaw, D.G.R. 1998. Nitrogen and phosphorous balances of Kindo Koisha farms in southern Ethiopia. *Agriculture, Ecosystem & Environment* 71: 93-113.

Fang, W. and Peng, S.L. 1997. Development of species diversity in the restoration process of establishing a tropical man-made forest ecosystem in China. *Forest Ecolology and Management* 99: 185-196.

FAO, 1976. A framework for land evaluation. Soils Bulletin 3. FAO, Rome, Italy.

FAO, 1986. Highland reclamation study, Final report, Vol. 1 and 2. FAOI, Rome.

FAO, 1990. Guidelines for soil profile description. Food and Agricultural Organization of The United Nation, Rome, Italy.

FAO, 1990. Guidelines for soil profile description. Rome, FAO, Italy.

FAO, 1993a. Ethiopia: Livestock Sector Development Project, Report No. 24//93 CP-ETH 45 SR, FAO, Rome.

FAO, 1993b. FESLM: An international framework for evaluating sustainable land management. World soil resources report 73. FAO, Rome.

FAO, 1998. World reference base for soil resources. World Soil Resources Reports 84. Food and Agricultural Organization of The United Nation, Rome, Italy.

Farnsworth, N. R. and Soejarto, D. D. 1991. Global importance of medicinal plants. In: Akerele, O., Heywood, V. and Synge, H. (eds). *Conservation of Medicinal Plants*. Cambridge, UK; Cambridge University Press. pp. 35-50.

Fearnside, P.M. and Barbosa, R.I., 1998. Soil carbon changes from conversion of forest to pasture in Brazilian Amazonia. *Forest Ecolology and Management* 108, 147-166.

Feigl, B.J., Milillo, J. and Cerri, C.C., 1995. Changes in the original and quality of soil organic matter after pasture introduction in Rondonia (Brazil). *Plant Soil* 175, 21-29.

Fernandes, E.C.M., Motavalli, P.P., Castilla, C. and Mukurumbira, L. 1997. Management of soil organic matter dynamics in tropical land-use systems. *Geoderma* 79: 49-67.

Fisher, R. F. 1995. Amelioration of degraded rain forest soils by plantations of native trees. *Soil Science Society of America Journal* 59: 544-549.

Foster, D. R. 1992. Land-use history (1730–1990) and vegetation dynamics in central New England, USA. *Journal of Ecology* 80: 753–772.

Foster, D. R., M. Fluet, and E. R. Boose. 1999, Human or natural disturbance: landscape-scale dynamics of the tropical forests of Puerto Rico. *Ecological Applications* 9: 555–572.

Francis, C.A. and Youngberg, G. 1990. Sustainable agriculture-an overview. In: Francis, A.C., Flora, B. C. and King, D.L. (eds.). *Sustainable Agriculture in Temperate Zones*. John-Wiley & Sons, Inc. USA. pp. 1-23.

Friis, I. 1992. Forests and forest trees of northeast tropical Africa. *Kew Bulletin Additional Series* 15: 1-396.

Friis, I. and Lawesson, J. 1993. Altitudinal zonation in the forest tree flora of Northeast tropical Africa. *Opera Bot*. 121: 125-127.

Garcia-Montiel, D. C. and Binkley, D., 1998. Effect of *Eucalyptus saligna* and *Albizia falcataria* on soil processes and nitrogen supply in Hawaii. *Oecologia* 113: 547-556.

Garten, Jr. C.T. 2002. Soil carbon storage beneath recently established tree plantations in Tennessee and South Carolina, USA. *Biomass & Bioenergy*, 23: 93-102.

Garwood, N.C. 1989. Tropical soil seed banks: A review. In: Leck, M.A., V.T. Parker and R.L. Simpson (eds). *Ecology of Soil Seed Banks*. Academic Press, San Diego, CA. pp 149-209.

Gibbon, D., Lake A. and Stocking, M. 1995. Sustainable development: A challenge for agriculture. In: Morse, S. and M. Stocking (eds.). *People and Environment*. UCL Press, UK.pp 31-68.

Gindaba, J. 1997 *Decomposition and Nutrient Release From Leaves of Croton macrostchyus and Millettia ferruginea for Soil Improvement in Agroforestry Systems.* MSc thesis Report No. 1997:24, Swedish University of Agricultural Sciences, Skinnskatteberg.

Gliessman, S.R. 2001. The ecological foundation of Agroecosystem sustainability. In: Gliessman, S.R. (ed.). *Agroecosystem Sustainability: Developing Practical Strategies.* CRS Press. pp.3-14.

Gomez, A.A., Swete Kelly, D.E., Syers, J.K. and Coughlan, K.J. 1996. Measuring sustainability of agricultural systems at the farm level. IN: Doran, J.W. Jones, A.J. (eds.), Methods for Assessing Soil Quality. *Soil Sci., Soc. Am*., Madison, WI. Pp. 401-410.

Grace, P.R., Oades, J.M., Keith, H. and Hancock, T.W. 1995. Trends in wheat yields and soil organic carbon in the Permanent Rotation Trial at the Waite Agricultural Research Institute, South Australia. *Australian Journal of Experimental Agriculture* 35: 857-864.

Granström, A. 1986. *Seed Banks In Forest Soils and Their Roles in Vegetation Succession After Disturbance.* PhD Dissertation, Swedish University of Agricultural Sciences. Dept. of Forest Site Research, Umeå.

Greenland, D. 1975. Bringing the green revolution to the shifting cultivator: better seeds, fertilizers, zero or minimum tillage, and mixed cropping are necessary. *Science* 190: /4217), 845.

Greenland, D. and Swift, L.W. 1991. Climate variability and ecosystem response: opportunities for LTER network. *Bulletin of Ecological Society of America* 72: 188-226.

Greenland, D.J. and Szabolcs, I. 1994. *Soil Resilience and Sustainable Land Use*. CABI, Wallingford.

Grove, T.S. O'Connell, A.M. Mendham, D.S. Barrow, N.J. and Rance, S.J. 2001. Sustaining the productivity of tree crops on agricultural land in south-western Australia, Publication No.10/09. Rural Industries Research and Development Corporation (RIRDC), Canberra.

Guariguata, M.R., Rheingans, R. and Montagnini E. 1995. Early woody invasion under tree plantations in Costa Rica: implications for forest restoration. *Restoration Ecology* 3: 252-260.

Hailu, Z. 2002. *Ecological Impact Evaluation Of Eucalyptus Plantations in Comparison With Agricultural and Grazing Land-Use Types in the Highlands of Ethiopia*. Ph.D.

dissertation, Institute of Forest Ecology, Vienna University of Agricultural Sciences, Vienna.
Hall, T.B., Rosillo-Calle, F., Williams, R.H. and Ewoods, J. 1993. Biomass for energy: supply prospect. In: Hall, T.B. Kelly, H., Reddy, A.K.N., Williams, R.H. (eds). *Renewable Energy: Sources for Fuels and Electricity*. Island Press, Washington D.C, pp. 593-651.
Hamilton, A. C. 1982. *Environmental History of East Africa*. Academic press.
Harrington, C.A. 1999. Forests planted for ecosystem restoration or conservation. *New Forets* 17: 175-190.
Harrington, R.A. and Ewel, J.J. 1997. Invasibility of tree plantations by native and non-indigenous plant species in Hawaii. *Forest Ecolology and Management* 99: 153-162.
Hartemink, A.E. 1997. Soil fertility decline in some Major Soil Groupings under permanent cropping in Tanga Region, Tanzania. *Geoderma* 75: 215-229.
Hartemink, E.A., 1998. Soil chemical and physical properties as indicator of sustainable land management under sugar cane in Papua New Guinea. *Geoderma* 85: 283-306.
Hawando, T. 1997. *Desertification in Ethiopian Highlands*. RALA report No. 200. Norwegian Church AID, Addis Ababa, Ethiopia.
Hayes, F.E. and Samad, I. 1998. Diversity, abundance and seasonality of birds in a Caribbean pine plantation and native-broadleaved forest at Trinidad, West Indies. *Bird Conservation Internationa l* 8: 67–89.
Herrick, J.E. 2000. Soil quality: an indicator of sustainable land management? *Applied Soil Ecology* 15: 75-83.
Herweg, K. and Stillhardt, B. 1999. *The Variability of Soil Erosion in the Highlands of Ethiopian and Eritrea.* Research Report No. 42. Center for Development and Environment. University of Berne: Berne.
Hill, M.O. 1979. The development of a flora in even-aged plantations. In: Ford, E.D., Malcolm, D.C. and Atterson, J. (eds). *The Ecology of Even-aged Forest Plantations*. Institute of Terrestrial Ecology Edinburgh, United Kingdom. pp. 175–192
Hill, M.O. 1987. Opportunities for vegetation management in plantation forests. In: Good, J.E.G. (ed). *Environmental Aspects of Plantation Forestry in Wales*. Institute of Terrestrial Ecology, Edinburgh, United Kingdom. pp. 64–69.
Hill, M.O. and Wallace, H.L. 1989. Vegetation and environment in afforested sand dunes at Newborough, Anglesey. *For*estry 62: 249–267.
Hills, C.S. and Morris, M.D. 1992. The function of seed banks in northern forest ecosystems: a literature review. Ontario Ministry of Natural Resources, Forest Research Information Paper No. 107.
Högberg, P. 1990. Forests losing large quantities of nitrogen have elevated $^{15}N/^{14}N$ ratios. *Oecologia* 84: 229-231.
Honnay, O. Hermy, M. and Coppin, P. 1999. Impacts of habitat quality on forest plant species colonization. *Forest Ecology and Management* 115: 157-170
Hurni, H. 1988. Land degradation and conservation of soil resources in the Ethiopian highlands. *Mountain Research and Development* 8: 123-130.
Hurni, H. 1993. Land degradation, famines and resource scenarios in Ethiopia. In: Pimentel D. (ed). *World Soil Erosion and Conservation*. Cambridge University Press: Cambridge. pp. 27–62.
IMF, 2002. *Ethiopia: Statistical Appendix*. Washington, D.C: IMF Country Report No. 02/214.
Islam, K.R. and Weil, R.R. 2000. Land use effects on soil quality in a tropical forest ecosystem of Bangladesh. *Agriculture, Ecosystem & Environment* 79: 9-16.
Jaiyeoba I. A. 2001. Soil rehabilitation through afforestation: evaluation of the performance of Eucalyptus and pine plantations in a Nigerian Savanna environment. *Land degradation & development* 12: 183-194.
Janssen, M. and Willkens, A. 1994. *Interpretation of Crop Response Data, Ethiopia*. Consultancy Report, Project GCPF /ETH/039/ITA, FAO, Rome.
Jenkins, R.K.B., Roettcher, K. and Corti, G. 2003. The influence of stand age on wildlife habitat use in exotic Teak tree *Tectona grandis* plantations. *Biodiversity and Conservation* 12: 975-990.

Jiru, D. 1989. Evaluation of crop yield under *Acacia albida* shade and alley cropping in semiarid farms. In: *Proceedings of IAR/ICRAF National Agroforestry Workshop,* 14-18 Nov. 1988 Awassa, Ethiopia. Institute of Agricultural Research, Addis Ababa. Pp. 33-38

Jonsson, K. Ståhl, L. and Högberg, P. 1996. Tree fallows: A comparison between five tropical species. *Biolgical Fertilty of Soils* 23: 50-56.

Jordan, C.G. 1985. *Nutrient Cycling in Tropical Forest Ecosystems*. John Wiley & Sons, New York.

Kapkiyai, J.J., Karanja, N.K., Woomer, P. and Qureshi, J.N. 1998. Soil organic carbon fractions in a long-term experiment and the potential for their use as a diagnostic assay in highland farming systems of central Kenya. *African Crop Science Journal* 6: 19-28.

Kardell, L., Steen, E. and Fabiao, A. 1986. Eucalytpus in Portugal- a threat or a promise. *Ambio* 165: 7-13.

Karlen, D.L. Eash, N.S. and Unger, P.W. 1992. Soil and crop management effects on soil quality indicators. *American Journal of Alternative Agriculture* 7: 48-55.

Kassa, H. B. 2003. *Livestock and Livelihood Security in the Harar Highlands of Ethiopia: Implications for Research and Development*. Doctoral thesis Swedish university of agricultural Sciences, Department of Rural Development Studies, Uppsala.

Keenan, R., Lamb, D., Woldring, O., Irvine, T. and Jensen, R. 1997. Restoration of plant biodiversity beneath tropical tree plantations in northern Australia. *Forest Ecology and Management* 99: 171-131.

Keersmaeker, D.L., Martens, L., Verheyen, K., Hermy, M., Schrijver, D.A. and Lust, N. 2004. Impact of soil fertility and insolation on diversity of herbaceous woodland species colonizing afforestations in Muizen forest (Belgium*). Forest Ecology and Management* 188: 291-304.

Kelbessa, E., Sebsebe, D., Zerhun, W. and Edwards, S. 1992. Some threatened endemic plants of Ethiopia. *NAPRECA Monograph* 2: 35-55.

Kellomäki, S., Kouki, J., Neimelä, P. and Peltola, H. 2001. Timber industry. *Encyclopedia of Biodiversity* 5: 655-666.

Kelly, E. Chadwick, O.A. and Hilinski, T.E. 1998. The effect of plants on mineral weathering. *Biogeochemistry* 42: 21-53.

Khurana, E. And Singh, J.S. 2001. Ecology of tree seed and seedlings: Implications for tropical forest conservation and restoration. *Current Science* 80: 748-757.

Kidanu, S. 2004. Using Eucalyptus for soil & water conservation on the highland vertisols of Ethiopia. Tropical Resource Management Papers, No. 52 (2004). Wageningen University, The Netherlands.

King, D.L. 1990. Sustainable soil fertility practices. In: Francis, A.C., Flora, B. C. and King, D.L. (eds). *Sustainable Agriculture in Temperate Zones*. John-Wiley & Sons, Inc. USA.pp. 144-173.

Krebs, C.J. 1985. Ecology: the Experimental Analysis of distribution and abundance. Harper & Row, New York.

Krebs, J.E. 1975. A comparison of soils under agriculture and forests in San Carlos, Costa Rica. In: Golley, F.B. and Medina, E. (eds), ibid. pp. 381-390.

Krebs, J.E., Jan, K.H., and Golley, F.B. 1974. A comparative study on chemical characteristics of tropical soils from volcanic material under forest and agriculture. *Communication in Soil Science and Plant Analysis* 5: 579-596.

Kumar, H. D. (ed.). 1999. *Biodiversity and Sustainable Conservation*. Science Publishers, Inc. USA.

Kuru, A. 1990. Roots of deforestation and problems in Ethiopia. In: Palo, M. and Mery, G. (eds). *Deforestation or Development in the Third World*? Vol. II. Finnish Forest Research Institute, Helsinki. pp. 71-79.

Lal, R. 1976. Soil erosion problems on Alfisols in Western Nigeria and their control, IITA Monograph No-1, ITTA, Ibadan.

Lal, R. 1996. Deforestation and land use effects on soil degradation and rehabilitation in western Nigeria. II. Soil chemical properties. *Land Degradation & Development* 7: 87-98.

Lamb, D. 1998. Large-scale ecological restoration of degraded tropical forest lands: The potential role of timber plantations. *Restoration Ecology* 6: 271-279.

Lamb, D. and Tomlinson, M. 1994. Forest rehabilitation in the Asia-Pacific region: past lessons and present uncertainties. *Journal of Tropical Forest Science* 7: 157-170.

Landon, J.R. 1991. *Booker Tropical Soil Manual. A Handbook for Soil Survey and Agricultural Land Evaluation in the Tropics*. Longman Inc. New York.

Larson, W.E. and Pierce, F.J. 1994. The dynamics of soil quality as a measure of sustainable management. In: Doran, J.W., Coleman, D.C. Bezdicek, D.F. and Stewat, B.A. (eds). *Defining Soil Quality for a Sustainable Environment*. Soil Science Society of America. Special publication No.35, Madison, WI. pp. 37-51.

Laurance, W.F., Delamonica, P., Laurance, S.G. Vasconcelos, H.L. and Lovejoy, T.E. 2000. Rainforest kills big trees. *Nature* 404: 836.

Laurance, W.F., Ferreira, L., Rankin-De Morena, J., and Laurance, S.G. 1998. Rain forest fragmentation and the dynamics of Amazonian tree communities. *Ecology* 79: 2032-2040.

Laurance, W.F., Lovejoy, T.E., Vasconcelos, H., Bruna, E., Didhiam, R. Stouffer, P., Gascon, C. Bierregaad, R.O., Laurance, S.G. and sampiao, E. 2002. Ecosystem decay of Amazonian forest fragments: a 22-year investigation. *Conservation Biology* 16: 605-616.

Laurance, W.F., Rankin-De Merona, J., Andrade, A. Laurance, S.G. D'Angelo S., Lovejoy, T.E, and Vasconcelos, H.L. 2003. Rain-forest fragmentation and the phenology of Amazonian tree communities. *Journal of Tropical Ecology* 19: 343-347.

Laycock, W.A. 1995. New perspectives on ecological condition of rangelands: can state and transition or other models better define condition and diversity? In: Montes, L, Olivia, GE. (eds). *Proceedings of the International Workshop on Plant Genetic Resources, Desertification and Sustainability*, INTA-EEA, Rio Gallegos, Argentina.

Leck, M.A., Parker, V.T., and Simpson, R.L. (eds.) 1989. *Ecology of Soil Seed Banks*. Academic Press, San Diego.

Legesse, B. 2003. *Risk Management Strategies Of Smallholder Farmers in the Eastern Highlands of Ethiopia*. Doctoral thesis, Swedish University of Agricultural Sciences, Department of Rural Development Studies, Uppsala, Sweden.

Lemenih, M. and Teketay, D. 2004. Constraints and opportunities associated with restoration of native forest flora in the degraded highlands of Ethiopia: a review. *SINET: Ethippian Jpournal of Science* (in press).

Lemenih, M. and Bekele, T. 2004. Effect of age on calorific value and some mechanical properties of three *Eucalyptus* species grown in Ethiopia. *Biomass & Bioenergy* (In press).

Lemmon, P. E. 1956. A spherical densiometer for estimating forest overstory density. *Forest Science* 2: 314-320.

Lilienfein, J., Wilcke, W., Ayarza, M.A., Vilela, L., Lima, S.d.C., Zech, W., 2000. Chemical fractionation of Phosphorus, Sulpher, and Molybdenum in Brazilian savannah Oxisols under different land use. *Geoderma* 96: 31-46.

Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Crime, J.P., Hector, A., Hooper, D.U. Huston, M.A., Raffaelli, D., Schmid, B., Tilman, D. and Wardle, D.A. 2001. Biodiversity and Ecosystem functioning: current knowledge and future challenges. *Ecology* 294: 804-808.

Loumeto, J.J. and Huttel, C. 1997. Understorey vegetation in fast-growing tree plantations on savanna soils in Congo. *Forest Ecolology and Management* 99: 65-81.

Lourzada, J.N.C., Schoereder, J.H., and Marco, P.D., 1997. Litter decomposition in semi-decidous forest and Eucalyptus spp. Crop in Brazil: a comparison. *Forest Ecolology and Management* 94: 31-36.

Lugo, A.E. 1992. Tree plantations for rehabilitating damaged lands in the tropics. In: Wali, M.K. (ed). *Environmental Rehabilitation*, Vol. 2, SPB Academic Publishing. The Hague. pp. 247-255.

Lugo, A.E. 1997. The apparent paradox of re-establishing species richness on degraded lands with tree monocultures. *Forest Ecolology and Management* 99: 9-19.

Lugo, A.E., Parrotta, J.A. and Brown, S. 1993. Loss of species caused by tropical deforestation and their recovery through management. *Ambio* 22: 106-109.

Lundgren, B. 1971. Soil studies in a montane forest in Ethiopia. Research Notes Nr11. Department of Forest ecology and Forest Soils, Royal College of Forestry, Stockholm.

Lundgren, B., 1978. *Soil Conditions and Nutrient Cycling Under Natural and Plantation Forests in Tanzanian Highlands*. 31. 1978. Uppsala, Sweden, Department of Forest Soils, Swedish University of Agricultural Sciences. Reports No. 31. Uppsala.

Lundgren, L. and Taylor, G. 1993. *From Soil Conservation to Land Husbandry: Guidelines Based on Sida's Experience*. Sida. 58 pp.

Lyaruu, H.V. 1998. *Seed dynamics and the Ecological Restoration of Hill Slopes of Kondoa Irangi, Central Tanzania*. PhD thesis, Faculty of Science and Technology 383, Uppsala, Sweden.

MacDonald, W. D., Moorhouse, P. T. and Enck, W.J. 2002. The ecological context: a species population perspective. In: Perrow, M.R. and Davy A. J. (eds). *Handbook of Ecological Restoration. Vol. 1. Principles of Restoration*. Cambridge University Press, Cambridge. pp. 155 –183.

MacMahon, J.A. 1997. Ecological restoration. In: Meffe, G.H. and Carroll, C.R. (eds.), *Principles of Conservation Biology,* 2dn ed. Sinauer Associates, Inc. Publishers, Sunderland, Massachusetts, USA. pp. 479-512.

Makin, M.J., Kingham, J.J., Waddams, A. E., Brichall, C.J. and Tamene T., 1975. *Land resources Study 21. Development Prospects in the southern Rift Valley, Ethiopia*. Land Resources Division, Great Britain Ministry of Overseas Development No. 21. pp. 270+maps

Marques, R. and Ranger, J. 1997. Nutrient dynamics in a chronosequence of Douglas-fir (*Pseudotsuga menziesii* (Mirb) Franco) stands on the Beaujolais Mounts (France). 1. Qualitative approach*. Forest Ecolology and Management* 91: 255-277.

Marris, H. R. 2002. Manipulating the chemical environment of the soils. In: Perrow, M.R. and Davy A. J. (eds). *Handbook of Ecological Restoration. Vol. 1. Principles of Restoration*. Cambridge University Press, Cambridge. pp. 155 –183.

Marry, M.V. and Sankaran, K.V. 1991. Ex-situ decomposition of leaf litters of *Tectona grandis, Eucalyptus tereticornis* and *albizia falcataria*. KFRI Research Report 71. pp.41.

Martin, A., Mariotti, A., Balesdent, J., Lavelle, P. and Vuattoux R. 1990. Estimate of organic-matter turnover rate in a savanna soil by C-13 natural abundance measurements. *Soil Biology and Biochemistry* 22: 517-523.

McCann, J. C. 1999. *Green Land, Brown Land, Black Land: An Environmental History of Africa, 1800-1990.* James Currey Ltd. USA.

McDonagh, J.F., Thomsen, T.B. and Magid, J. 2001. Soil organic matter decline and compositional change associated with cereal cropping in southern Tanzania. *Land degradation & Developmen*t 12: 13-26.

McInvar, J. and Starr, L. 2001. Restoration of degraded lands in the interior Columbia River basin: passive vs. active approaches. *Forest Ecolology and Management* 153: 15-28.

Mendham, D.S., O'Connell, A.M. and Grove, T. S. 2003. Change in soil carbon after land clearing or afforestation in highly weathered lateritic and sandy soils of south-western Australia. *Agriculture, Ecosystem & Environment* 95:143-156.

Miller, K., Allegretti, M.H., Johnson, N. and Jonsson, B. 1995. Measures for conservation of biodiversity and sustainable use of its components. In: Heywwod, V.H. and Watson, R.T.(eds). *Global Biodiversity Assessment*. The United Nations Environment Programme and Cambridge University Press, Cambridge. pp. 915-1061,

Mizota, C. and van Reeuwijk, L.P., 1989. *Clay mineralogy and Chemistry Of Soils Formed Under Volcanic Material in Diverse Climate Regions*. Soil Monogr. 2 ISRIC, Wageningen.

MoA (Ministry of Agriculture), 2002. Rural Development Strategy. MOA, Addis Ababa, Ethiopia.

MoFED (Ministry of Finance and Economic Development) 2002. Ethiopia: Sustainable Development and Poverty Reduction Program (SDPRP). Addis Ababa, Ethiopia. Also available on web at http://www.worldbank.org/afr/et/prsp.htm.

Montagnini, F. 2001. Strategies for the recovery of degraded ecosystems: experiences from Latin America. *Interciencia* 26: 498-503

Mooney, H. F. 1954 *Report On Forestry In Ethiopia With Special Reference to the Forests of Arssi and Sidamo*. Submitted to the Ministry of Agriculture. Imperial Forestry Institute of Oxford University.

Motzkin, G., Foster, D., Allen, A., Harrod, J. and Boone, R. 1996. Controlling site to evaluate history: vegetation patterns of a New England sand plain. *Ecological Monographs* 66: 345–366.

Mugabe, J. 1998. Biodiversity and sustainable development in Africa. In: Mugabe, J. and Clark, N. (eds). *Managing Biodiversity. National Systems of Conservation and Innovation in Africa*. African Center for Technology Studies (ACTS), Nairobi, Kenya. pp. 5-30.

Murage, E. W., Karanja, N.K., Smithson, P.C. and Woomer, P.L. 2000. Diagnostic indicators of soil quality in productive and non-productive stallholders' fields of Kenya's Central Highlands. *Agriculture, Ecosystem & Environment* 79: 1-8.

Murphy, P. G. and Lugo, A.E., 1986. Ecology of tropical dry forest. *Ann. Rev. Ecol. Syst.* 17: 67-88.

Nambiar, K.K.M., Gupta, A.P., Fu, Q., Li, S., 2001. Biophysical, chemical and socio-economic indicators for assessing agricultural sustainability in the Chinese coastal zone. *Agriculture, Ecosystem & Environment* 87: 209-214.

Nason, J.D. and Hamrick, J.L. 1997. Reproductive and genetic consequences of forest fragmentation: two case studies of neotropical canopy trees. *The Journal of Heredity* 88: 264-276.

NCSS (National Conservation Strategy Secretariat), 1993. National Conservation Strategy. Vol.1. National Policy on the Resources Base, its Utilization and Planning for Sustainability. Addis Ababa, Ethiopia, 131p.

Nepstad, D. C., C. Uhl, C. A. Pereira, and J. M. C. da Silva. 1996. A comparative study of tree establishment in abandoned pasture and mature forest of eastern Amazonia. *Oikos* 76: 25–39.

Noss, R.F. 1990. Indicators for monitoring biodiversity: a hierachical approach. *Conservatio Biology* 4: 355-364.

Nyberg G. and Högberg P. 1995. Effects of young agroforestry trees on soils in on-farm situations in Western Kenya. *Agroforestry systems* 32: 45-52.

Nye, P.H. and Greenland, D.J. 1960. The Soil Under Shifting Cultivation. Tech. Commun. 51. Commonwealth Bureau of Soils, Harpenden, UK.

Olsen, S.R., Cole, C.V., Watanabe, F.S., Dean L.A., 1954. *Estimation of Available Phosphorous in Soils by Extraction with Sodium Bicarbonate*. U.S Dep. Agri. Cir. 939. USDA, Washington D.C.

Olsson, M. 2001. Do trees improve soil? Swedish University of Agricultural Sciences. *Currents* 25/26: 31-34.

Omiti, J.M., Parton, K.A., Sinden, J.A., Ehui, S.K. 1999. Monitoring changes in land-use practices following agrarian de-collectivisation in Ethiopia. *Agriculture, Ecosystem & Environment* 72: 111-118.

Otsamo R. 1998. Effect of nurse tree species on early growth of *Anisotepa marginata* Korth. (Diptercarpaceae) on an *Imperata cylindrical* (L.) Beauv. grassland site in South Kalimantan, Indonesia. *Forest Ecolology and Management*105: 303-311.

Otsamo, R. 2000. *Integration of I Indigenous Tree Species into Fast-growing Forest Plantations on Imperata Grasslands in Indonesia. Silvicultural Solutions and Their Ecological and Practical Implications***.** PhD Dissertation. Department of Forest Ecology, University of Helsinki, Helsinki.

Ouedraogo, E. 2004. Soil quality improvement for crop production in semi-arid West Africa. Tropical Resource Management Papers No. 51 (2004). Wageningen University.

Parfitt, R. L, Theng, B. K. G., Whitton, J.S. and Shepherd, T.G. 1997. Effects of clay minerals and land use on organic matter pools. *Geoderma* 75:1-12.

Parrotta J.A. 1999. Productivity, nutrient cycling, and succession in single- and mixed-species plantations of *Casuarina equisetifolia*, *Eucalyptus robusta*, and *Leucaena leucocephala* in Puerto Rico. *Forest Ecolology and Management* 124: 45-77.

Parrotta, J.A. 1992. The role of plantation forests in rehabilitating degraded tropical ecosystems. *Agriculture. Ecosystem & Environment* 41: 115-133.

Parrotta, J.A. 1993. Secondary forest regeneration on degraded tropical lands: the role of plantations as “foster ecosystems”. In Lieth, H. and Lohmann, M. (eds). *Restoration of Tropical Forest Ecosystems*. Kluwer Academic Publishers, The Hague, The Netherlands. Pp. 63–73.

Parrotta, J.A. 1995. Influence of overstory composition on understory colonization by native species implantations on a degraded tropical site. *Journal of Vegetation Science* 6: 627 – 636.

Parrotta, J.A., Turnbull, W.J. and Jones, N. 1997. Catalysing native forest regeneration on degraded tropical lands. *Forest Ecolology and Management* 99: 1-7.

Paul K.L., Polglase P.J., Richards G.P. 2003. Predicted change in soil carbon following afforestation or reforestation, and analysis of controlling factors by linking a C accounting model (CAMFor) to models of forest growth (3PG), litter decomposition (GENDEC) and soil C turnover (RothC). *Forest Ecolology and Management* 177: 485-501.

Paul, E.A. and Clark, F.E. 1996. *Soil Microbiology and Biochemistry*. Academic Press. California, USA.

Paul, K.I., Polglase, P.J., Nyakuengama, J.G. and Khanna, P.K. 2002. Change in soil carbon following afforestation. *Forest Ecolology and Management* 168: 241-257.

Penuelas, J. Filella, I. and Terradas, J. 1999. Variability of plant nitrogen and water use in a 100-m transect of a subdesertic depression of the Ebro valley (Spain) characterized by leaf $\delta^{13}C$ and $\delta^{15}N$. *Acta Oecologica* 20: 119-123.

Perrow, M.R. and Davy A. J. (eds). 2002. *Handbook of Ecological Restoration*. *Vol. 1. Principles of Restoration*. Cambridge University Press, Cambridge.

Pohjonen, V. 1989. Establishment of fuelwood plantations in Ethiopia. Siva Carelica 14: 1-388.

Popenoe, H. 1960. Effects of shifting cultivation on natural soil constituents in central America. In; *Dissertification Abstract* 20: 4473-4474.

Post M.W., Kwon, C.K. 2000. Soil carbon sequestration and land-use change: processes and potential. *Global Change Biology* 6: 317-327.

Pretty, J.N. 1995. Participatory learning for sustainable agriculture. *World Development* 23:, 1247-1263.

Quimby, P. C., King, L.R. and Grey, W.E. 2002. Biological control as a means of enhancing the sustainability of crop/land management systems. *Agriculture, Ecosystem & Environment* 88: 147-152.

Raulund-Rasmussen, K., Borggaard, O.K. Hansen, H.C.B. and Olsson, M. 1998. Effect of natural organic soil solutes on weathering rates of soil minerals. *European Journal of Soil Science* 49: 397-406.

Reicosky, D.C. and Lindstrom, M.J., 1993. Fall tillage methods: Effect on short-term CO_2 flux from soil. *Agronomy Journal* 85: 1237-1243.

Reusing, M. 1998. *Monitoring of Forest Resources in Ethiopia*. Ministry of Agriculture, Addis Abeba.

Rhoades C. and Binkley, D. 1996. Factors influencing decline in soil pH in Hawaiian Eucalyptus and Albizia plantations. *Forest Ecolology and Management* 80: 47-56.

Rocheleau, D., Webber, F. and Field, J.A. 1988. *Agroforestry in Dryland Africa.* ICRAF. Nairobi. Kenya. 311 pp.

Roming, D.E., Garlynd, M.J., Harris, R.F. and McSweeney, K. 1995. How farmers assess soil health and quality. *Journal of Soil & Water Conservation* 50: 229-236

Rowe, R., Sharma, N. and Browder, J. 1994. Deforestation: problems, causes and concerns. In: Norén, S. (Ed.) Compendium for the 20th Inter Nordic course on forests and forestry in developing countries. Swedish University of Agricultural Sciences/International Rural Development Centre. Vol. 2.

Russ. G.W. 1946 *Reports on Ethiopian Forests.* Complied by Wolde Micheal Kelecha (1979), Forestry and Wildlife Authority. Addis Ababa, Ethiopia.

Rydgren, K., Hestmark G. and Okland, R.H. 1998. Revegetation following experimental disturbance in a boreal old-growth *Picea abies* forest. *Journal of Vegetation Science* 9: 763-776.

Saggar S., Tate K.R., Feltham, C.W., Childs, C.W. and Parshotam, A 1994. Carbon turnover, in a range of allophonic soils amended with ^{14}C-labelled glucose. *Soil Biology and Biochemistry* 26: 1263-1271.

Saggar, S., Parshotam, A., Sparling, G.P., Feltham, C.W. and Hart, P.B.S. 1996. ^{14}C-labelled ryegrass turnover and residence times in soils varying in clay content and mineralogy. *Soil Biology & Biochemistry* 28: 1677-1686.

Sahlemedhin S. 1999. *Ethiopia: Integrated Soil Management for Sustainable Agriculture and Food Security in Southern and East Africa*. Proceedings of the expert consultation, 8-12 December, 1997, Harare, agritex, FAO, Rome, pp. 197-210.

Salami, A.T. 1998. Vegetation modification and man-induced environmental change in rural southwestern Nigeria. *Agriculture Ecosystem & Environment* 70: 159-167.

Sanchez, P.A. 1976. *Properties and Management of Soils in the Tropics*. John Wiley & Sons, New York .

Sanchez, P.A. and Garduno, M.A. 1994. Symposium introduction. In : Sanchez P.A. and van Houten H. (eds.). *Alternative to Slash-to-Burn Agriculture*. Symposium ID-6. Vol. 9, Supplement to Transactions. Proceedings of the Transactions of 15th World Congress of soil Science, July 10-16, Acapulco, Mexico. INEGI and CAN (Mexico). pp. 1-2.

Sanchez, P.A. Palm, C.A., Szott, L.T. Cuevas, E. and Lal, R. 1989. Organic input management in tropical agroecosystems. In: Coleman, D.C., Oades, J.M. Uehara, G.D. (eds.). *Dynamics of soil Organic Matter in Tropical Ecosystems*. Univ. of Hawaii Press, Honolulu, Hawaii. pp 125-152.

Sanchez, P.A., Palm, C.A., Davey, C.B., Szott, L.T. and Russell, C.E. 1985. Trees as soil improvers in the humid tropics? P. 327-358. In: Cannell, M.G.R. and. Jackson, J.E (eds.) *Attributes of Trees as Crop Plants.* Institute of Terrestrial Ecology, Huntingdon, England.

Sayer, A.J., Harcourt, S.C. and Collins, M.N. 1992. *The Conservation Atlas of Tropical Forests Africa.* IUCN, Cambridge.

Scholes, M.C., Swift, M.J., Heal, O.W., Sanchez, P.A., Ingram, J.S.I. and Dalal, R.C. 1994. Soil fertility research in response to the demand for sustainability. In: Woomer, P.L., Swift, M.J. (eds.). *The Biological Management of Tropical Soil Fertility*. Wiley, Chichester, pp. 1-14.

SCRP 1984–1991. *Annual Progress Reports, Soil Conservation Research Project (SCRP*), Ministry of Agriculture, University of Berne (Switzerland) and United Nations University (Tokyo), vol. 1–9, Addis Ababa, Ethiopia

Second Citizen Report, 1985. *The State of India's Environment*. New Delhi: Center for Science and Education.

Seifu, K. 1998. *Estimating Land Cover/Land Use Changes in Munessa Forest Area using Remote Sensing Techniques*. MSc thesis Report No. 1998:32, Swedish University of Agricultural Sciences, Skinnskatteberg,

Senbeta, F. and Teketay, D. 2001. Regeneration of indigenous woody species under the canopy of tree plantations in Central Ethiopia. *Tropical Ecology* 42: 175-185.

Senbeta, F., Teketay, D. and Näslund, B.Å. 2002. Native woody species regeneration in exotic tree plantations at Munessa-Shashamane forest, Southern Ethiopia. *New Forests* 24: 131-145.

Simberloff, D. 1999. The role of science in the preservation of forest biodiversity. *Forest Ecolology and Management* 115: 101-111.

Singh, P. 1995. Land degradation –A global menace and its improvement through Agroforestry. In: Signh, P. Pathak, P.S. and Roy, M.M. (eds.). *Agroforestry Systems for Sustainable Land Use*. Science publishers, Inc. USA. Pp. 4-20.

Smaling, E. M.A., Fresco, L. O. and de Jager, A. 1996. Classifying, monitoring and improving soil nutrient stocks and flows in African agriculture. *Ambio* 25: 492-496.

Smith, C. T. 1994. Is plantation forestry good or bad for soils? *New Zealand Forestry* 39: 19-22.

Society of Ecological restoration, 1996. Ecological restoration: Definition. http://www.ser.org.

Soil Survey Staff. 1999. Keys to soil taxonomy. 8th ed. Pocahontas Press, Blacksburg. VA.

Sojka, R.E. and Upchurch, D.R. 1999. Reservations regarding the soil quality concept. *Soil Science Society of America Journal 63*: 1039-1054.

Solomon, D., Fritzsche, F., Lehmann, J., Tekalign, M. and Zech, W. 2002. Soil organic matter dynamics in the subhumid agroecosystems of the Ethiopian highlands: Evidence from natural ^{13}C abundance and particle-size fractionation. *Soil Science Society of America Journal* 66: 969-978.

Stahr, K., 2000. Land use planning in west Africa. Adapted farming in West Africa, 5.0-7.7.2000, IBS. University Hoenheim. Germany (unpublished stencile).

Steer, A. 1998. Making development sustain able. *Advance in Geo-Ecology* 31: 857-865.

Stone, E.L. 1975. Effects of species on nutrient cycles and soil change. Phil, *Transaction in R. Soc. Lond. B*. 271: 149-162.

Stoorvogel, J.J. and Smaling, E.M.A. 1990. *Assessment of Soil Nutrient Depletion in Sub-Saharan Africa 1983-2000.* Report 28. The Winand Staring Center for integrated Land. Soil and Water Research (SC-DLO), Waeningen, The Netherlands.

Stoorvogel, J.J. Smaling, E.M.A. and Janssen B.H. 1993. Calculating soil nutrient balances in Africa at different Scales. I Supra-national scale. *Fertilizer Research* 35: 227-235.

Sullivan, T.P., Lautenschlager, R.A. and Wagner, R.G. 1999. Clear-cutting and burning of northern spruce-fir forests: implications for small mammal communities. *Journalof Applied Ecology* 36: 327–344.

Swift, M.J. and Woomer, P. 1993. Organic matter and sustainability of agricultural systems: Definition and measurement. In: Mulongoy, K. and Merckx. R. (eds.). *Proceedings of an International Symposium* held between 4-6 November, Leuven, Belgium. Pp. 3-17

Tadesse W., Denich, M., Teketay, D. and Vlek, P.L.G. 2002. Human impact on the *Coffea arabica* genepool in Ethiopia and the need for its in situ conservation. In: Engels, J.M.M., Brown, A.H.D. and Jackson, M.T., (eds,). *Managing Plant Genetic Resources*. CAB International, Wallington. pp. 237-247.

Tadesse W., Teketay, D., Denich, M. and Thomas, B. 2001. Diversity of traditional coffee production systems in Ethiopia and their contribution to the conservation of coffee genetic diversity. In: *Proceedings of the Conference on International Agricultural Research for Development*, Deutscher Tropentag-Bonn, Bonn. pp. 1-11,

Tamrat B., 1993. Vegetation ecology of remnant Afromontane forests on the Central Plateau of Shewa, Ethiopia. *Acta Phytogeographica Suecica* 79, Opulus Press AB, Uppsala, Sweden.

Tate, K. R. 1992. Assessment, based on a climosequence of soils in tussock grasslands, of soil carbon storage and release in response to global warming. *Journal of Soil Science* 43: 697-707.

Tedla, S. and Gebre, M. 1998. Biodiversity management in Ethiopia. In: Mugabe, J. and Clark, N. (eds). *Managing Biodiversity. National Systems of Conservation and Innovation in Africa*. African Centre for Technology Studies (ACTS), Nairobi, Kenya. pp. 65-90

Teketay, D. 1996. *Seed Ecology and Regeneration in Dry Afromontane Forests of Ethiopia.* Doctoral thesis, Swedish University of Agricultural Sciences, Umeå. 36 p. + 5 Appendices.

Teketay, D. 1997a. The impact of clearing and conversion of dry Afromontane forests into arable land on the composition and density of soil seed banks. *Acta Oecologica* 18: 557-573.

Teketay, D. 1997b. Seedling populations and regeneration of woody species in dry Afromontane forests of Ethiopia. *Forest Ecolology and Management* 98: 149-165.

Teketay, D. 1998. Soil seed banks at an abandoned Afromontane arable sites. *Feddes Repertorium* 109: 161-174

Teketay, D. 2000. Vegetation types and forest fire management in Ethiopia. Proceedings: Round table conference on integrated forest fire management in Ethiopia. 19-20 Sept. 2000. Ministry of Agriculture With GTZ and GFMC, Addis Ababa, Ethiopia. 1-35pp.

Tekle, K. 1998. *Ecological Rehabilitation of Degraded Hill Slopes in Southern Wello, Ethiopia*. Doctoral thesis, Acta Universitatis Upsaliensis, 363, Uppsalla Sweden.

Tekle, K. 1999. Land degradation problems and their implications for food shortage in South Wello, Ethiopia. *Environmental Management* 23: 419-427

Tekle, K. and Bekele, T. 2000. The role of soil seed banks in the rehabilitation of degraded hillslopes in Southern Wello, Ethiopia. *Biotropica* 32: 23-32.

Teklu, G. 2003. Expanse of plantation forest in Ethiopia (An outcome of more than half a century's effort). MoA, NRM&RD, Addis Ababa.

Teshome, T. and Petty, J.A. 2000. Site index equation for *Cupressus lusitanica* stands in Munessa Forest, Ethiopia. *Forest Ecolology and Management* 126: 339-347.

Tewolde BGE, 1990. The importance of Ethiopian forests in the conservation of Arabica coffee Gene-pools. In: *Proceedings of the Twelfth Plenary Meeting of AETFAT*. (Ihlenfeldt, H.D. ed.). Mitt. Inst. Allg. Bot. Hamburg. pp 65-72.

Tewolde BGE. 1989. The environmental variables which led to the ecological crisis in Ethiopia. *Coenoses* 4: 61-67.

Thompson, K., Bakker, J.P. and Bekker, R.M. (eds) 1996. *Soil Seed Banks of North-West Europe: Methodology, Density and Longevity*. Cambridge University Press, Cambridge.

Tiessen, H., Cuevas, E. and Chacon, P. 1994. The role of soil organic matter in sustaining soil fertility. *Nature* 371: 783-785.

Tiessen, M. and Shang C. 1996. Organic matter turnover in tropical land-use system: In: *Carbon and Nutrient Dynamics in Natural and Agricultural Tropical Systems*, Bergstrom L., Kirchmann H. (eds). CAB International: Harare. Pp. 1-14.

Tinker, P.P. Ingram, J.S.I. and Struwe, S. 1994. Effects of slash-and-burn agriculture, deforestation on climate vhange. In : Sanchez P.A. and van Houten H. (eds.). *Alternative to Slash-to-Burn Agriculture*. Symposium ID-6. Vol. 9, Supplement to Transactions. Proceedings of the Transactions of 15th World Congress of soil Science, July 10-16, Acapulco, Mexico. INEGI and CAN (Mexico). pp. 1-2.

Tole, L. 2002. Habitat loss and anthropogenic disturbance in Jamaica's Hellshire Hills area. *Biodiversity and Conser*vation 11: 575-598.

Tolera, A. 1996. *Socio-economic Study: Munessa-Shashamane State Forest Development and Utilization Project*. Final Report, Addis Ababa, Ethiopia.

Trouve, C., Mariotti, A., Schwartz, D., Guillet, B., 1994. Soil organic carbon dynamics under *Eucalyptus* and *Pinus* planted on savannas in the Congo. *Soil Biology and Biochemistry* 26: 287-295.

Tsegaye, B. 1997. The significance of biodiversity for sustaining agricultural production and role of women in the traditional sector: the Ethiopian experience. *Agriculture, Ecosystem and Environment* 62: 215-227.

van der Valk, A. G. and Pederson, R.L. 1989. Seed banks and the management and restoration of natural vegetation. In: Leck, M.A., Parker, V.T. and Simpson, R.L. (eds.). *Ecology of Soil Seed Banks.* Academic Press, Inc. San Diego. pp. 329-363,

van Noordwijk, M. Cerri, C. Woomer, P.L. Nugroho, K. and Bernoux, M. 1997. Soil carbon dynamics in the humid tropical forest zone. *Geoderma* 79: 187-225.

Vance, D.E. 2000. Agricultural site productivity: principles derived from long-term experiments and their implications for intensively managed forests. *Forest Ecolology and Management* 138: 369-396.

Veldkamp, E. 1994. Organic carbon turnover in three tropical soils under pasture after deforestation. *Soil Science Society of America.Journal* 58: 178-180.

Vernberg, F.J. 1988. The function of pristine estuarine ecosystem. In: Hook, D.D. (ed.). *The Ecology and Management of Wetla*nds. Timber Press, pp. 15-25.

Vitorello, V. A., Cerri, C. C., Andreux, F., Feller, C. and Victoria, R. L. 1989. Organic matter and natural carbon-13 distribution in deforested and cultivated Oxisols. *Soil Sciety of Society America Journal* 53: 773-778.

Vitousek, P.M. and Sanford, R.J. Jr. 1986. Nutrient cycling in moist tropical forest. *Annual review of Ecology and Systematics* 17: 137-167.

von Breitenbach, F. 1961. Forests and woodlands of Ethiopia, a geobotanical contribution to the knowledge of the principal plant communities of Ethiopia, with special regard to forestry. *Ethiopian Forest Review* 1; 5-16.

von Breitenbach, F. 1963. The indigenous trees of Ethiopia. Ethiopian Forester Association, Addis Ababa.

Wali, M. 1992. Ecology of the rehabilitation process. In : Wali, M. (ed.), Vol. 1. *Ecosystem Rehabilitatio*n. The Hague: SPB Academic publishing. pp. 3-26.
Wang, X. and Gong, Z. 1998. Assessment and analysis of soil quality changes after eleven years of reclamation in subtropical China. *Geoderma* 81: 339-355.
White, F. 1983. The vegetation of Africa. A descriptive memoir to accompany the UNESCO/AETFAT/UNSO vegetation map of Africa. *Natural Resources Research* (Paris) 20: 1-356.
Wielemaker, W.G. and Lansu, A.L.E., 1991. Land –use changes affecting classification of a Costa Rican soil. *Soil Sciety Society of America Journal* 55: 1621-1624.
Wilson, E.O. 1988. The current state of biological diversity. In: Wilson, E.O. (ed.). *Biodiversity*. Washington D.C. National Academy Press.
Woomer, P.L., Martin, A. Albrecht, A, Resck, D.V.S., and Scharpenseel, H.W. 1994. The importance and management of soil organic matter in the tropics. In: Woomer, P.L. and Swift, M.J., (eds.). *The Biological Management of Tropical Soil Fertility*. John Wiley & Sons, Chichester, UK. Pp. 47-80.
World Bank, 2002. The World Bank group, 2002. Country profile Table. Ethiopia Data Profile, WorldBank. URLhttp:/ /devdata. worldbank. Org/external/CPProfile.asp? SelectedCountry=ETH&CCODE=ETH&CNAME=Ethiopia&PTYPE=CP. Web page viewed in September 2003.
Wunderle, J.M. 1997. The role of animal seed dispersal in accelerating native forest regeneration on degraded tropical lands. *Forest Ecolology and Management* 99**:** 223-235.
Yalden, D.W. 1983. The extent of high ground in Ethiopia compared to the rest of Africa. *SINET, Ethiopean Journal of Science* 6: 35-39.
Yirdaw, E. 2001. Diversity of naturally-regenerated native woody species in forest plantations in the Ethiopian highlands. *New Forests* 22: 159-177.
Yirdaw, E. 2002. *Restoration of the Native Woody – Species Diversity, Using Plantation Species as Foster Trees in the Degraded Highlands of Ethiopia.* PhD Dissertation, Viikki Tropical Resources Institute, University of Helsinki, Finland.
Yirdaw, E. and Luukkanen O. 2003. Indigenous woody species diversity in *Eucalyptus globulus* Labill. ssp. globulus plantations in the Ethiopian highlands. *Biodiversity and Conservation* 12: 567-582.
Young, A. 1997. *Agroforestry for Soil Management,* 2nd ed. CAB International, Wallingford.
Young, A., 1991. Soil monitoring: a basic task for soil survey organizations. *Soil Use and Management* 7: 126-130.
Zanne, A.E., Keith, B., Chapman, C.A. and Chapman, L.J. 2001. Protecting terrestrial mammal communities: potential role of pine plantations. *African Journal of Ecology* 39**:** 399–401.
Zunino, H., Borie, F., Aguilera, S., Martine, J.P. and Haider, K. 1982. Decomposition of ^{14}C- labelled glocus, plant and microbial products and phenols in volcanic ash-derived soils of Chile. *Soil Biology & Biochemistry* 14: 37-43.

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