Effects of land-use changes on the properties of a Nitisol and hydrological and biogeochemical processes in different forest ecosystems at Munesa, south-eastern Ethiopia

Dissertation zur Erlangung des Doktorgrades an der Fakultät für Biologie, Chemie und Geowissenschaften

der Universität Bayreuth

Vorgelegt von

Yeshanew Ashagrie

Bayreuth, November 2004

Contents

List of tables

List of figures

List of abbreviations

BD	Bulk density
C/N	Carbon to nitrogen ratio
C/S	Carbon to sulphur ratio
CSA	Central Statistical Authority
D	Depth
dbh	Diameter at breast height
EFAP	Ethiopian Forestry Action Program
FAO	Food and Agriculture Organisation
ICP-AES	Inductively Coupled Plasma- Atomic Emission Spectrometry
iPOM	Intra particulate organic matter
LF	Light fraction
MEDaC	Ministry of Economic Development and Cooperation
MoFED	Ministry of Finance and Economic Development
MWD	Mean weight diameter
N/S	Nitrogen to sulphur ratio
NMSA	National Metrological Service Agency
OM	Organic matter
POM	Particulate organic matter
SOC	Soil organic carbon
SOM	Soil organic matter
TF	Throughfall
VWMC	Volume weighted mean concentration
WSA	Water stable aggregates

Organisation of the Thesis

This thesis is based on the following six papers (A–F) which are referred to in the text by their respective capital letters:

A. Yeshanew Ashagrie, Wolfgang Zech and Georg Guggenberger. 2005. Transformation of a *Podocarpus falcatus* dominated natural forest into a monoculture *Eucalyptus* globulus plantation at Munesa, Ethiopia: Soil organic C, N and S dynamics in primary particle and aggregate-size fractions. *Agriculture, Ecosystems & Environment 106, 89-98.*

B. Yeshanew Ashagrie, Wolfgang Zech, Georg Guggenberger and Tekalign Mamo. 2004. Soil aggregation and total and particulate organic matter as affected by conversion of native forests to 26 years continuous cultivation in Ethiopia.

C. Yeshanew Ashagrie, Wolfgang Zech, Georg Guggenberger and Demel Teketay. 2003. Changes in soil organic carbon, nitrogen and sulphur stocks due to the conversion of natural forest into tree plantations (*Pinus patula* and *Eucalyptus globulus*) in the highlands of Ethiopia. *World Resource Review 15, 462-482.*

D. Yeshanew Ashagrie and Wolfgang Zech.Water and nutrient inputs by rainfall into natural and managed forest ecosystems in south-eastern highlands of Ethiopia.

E. Yeshanew Ashagrie and Wolfgang Zech. Dynamics of dissolved nutrients in forest floor leachates: Comparison of a natural forest ecosystem with tree species plantations in south-east Ethiopia.

F. Yeshanew Ashagrie. Geochemistry of inorganic nutrients in water percolating through the mineral soils under two exotic tree species plantations and an adjacent natural forest in south-east Ethiopia.

Contributions of co-authors

This work is part of a DFG funded multidisciplinary research project comprising the following disciplines: Biogeography, Ecophysiology and Soil science. Prof. Zech and Prof. Guggenberger prepared the project proposal for the soil science part and designed the experimental work in the field. Prof. Zech gave me the topic of my thesis; we thoroughly discussed the schedule and the scope of the field and laboratory work and he supervised my work both in the field and laboratory. Interpretation of analytical results was discussed with Prof. Zech and Prof. Guggenberger, finally both read and improved the manuscripts.

Dr. Demel Teketay and Dr. Tekalign Mamo were our Ethiopian counter parts. They contributed in preparing the project proposal, supported me logistically during the field work, and read and improved the manuscripts.

Acknowledgments

First and for most I would like to express my sincere and deepest gratitude to Prof. Dr. Wolfgang Zech for having confidence in me to work with him, and specially for his enthusiasm to help me and share his experiences. I am also very much thankful to Dr. Ludwig Haumaier for the reading and improvement of the manuscript. My appreciations also go to the Amhara National Regional State (ANRS) government and Adet Agricultural Research Centre for giving me leave of absence, and the Ethiopian Agricultural Research Organisation (EARO) management for providing me all the logistical assistance I needed. The financial supports of the Kathlischer Academischer Ausländer-Dienst (KAAD) during my stay in Germany and the German Research Foundation (DFG) for the research through grant number Gu406–8 are highly acknowledged.

SUMMARY

The effects of conversion of natural forest into different exotic tree species plantations and crop cultivation were investigated at Munesa, south-eastern Ethiopia with the objectives of (i) determining changes on soil physical and chemical properties, (ii) quantifying water and nutrient fluxes under the different forest ecosystems, and (iii) assessing nutrient dynamics in water flowing through the soil under the different forest ecosystems. Soil samples were taken from the organic layer and at 0−20, 20−40, 40−70, 70−100 cm depths from the mineral soil. Rainfall and throughfall were collected using plastic funnels mounted 1 m above the ground. Soil solutions were collected with zero-tension (organic layer) and tension (mineral soil at the depth of 20, 50 and 100 cm) lysimeters. After 26 years of cultivation, surface (20 cm depth) soil structure was deteriorated and total soil organic carbon (SOC) and N contents both in bulk soil and water stable aggregates were significantly reduced. Below 21 years old *Eucalyptus* plantation no significant changes on the above mentioned parameters could be identified, but significant reductions in SOC, N and S concentrations associated with the sand and silt separates were evident. There were also significant reductions both in quality and quantity of particulate organic matter (POM) due to cultivation and only in quality of POM due to 21 years *Eucalyptus* plantation. The organic layer mass under 21 years old *Pinus patula*, 21 years old *Eucalyptus globulus* and third rotation *Eucalyptus globulus* (established 42 yr ago) decreased by 43%, 57% and 15%, respectively, relative to the natural forest. There were also significant reductions in the organic layer C and N stocks (9 to 60% and 25 to 68%, respectively), being highest under *Pinus* and lowest under third rotation *Eucalyptus*. In the mineral soil, to 1 m depth, there was a significant ($P<0.05$) reduction (16 to 20%) in SOC stock after conversion of natural forest into forest plantations. The N stocks under the 21 years old *Pinus* and third rotation *Eucalyptus* plantations were significantly reduced amounting 27 and 20% respectively, whereas 21 years old *Eucalyptus* had nearly an equivalent amount of N as that of the natural forest, probably due to a dense forest floor

vegetation, fixing N. The changes in the organic layer and mineral soil S stocks after plantation establishment were not significant.

Of the total annual rainfall (1190 mm) recorded during the monitoring period (October 2001 to September 2002), about 47% and 18% were intercepted by the canopies of *Cupressus* and the natural forest, and *Eucalyptus*, respectively. Total annual nutrients (Ca, Cl, K, Mg, Na, NH_4–N, NO_3–N, PO_4–P, SO_4–S) deposition by rainfall was 12 kg $ha^{-1}yr^{-1}$. Throughfall K, Mg, Ca and Cl fluxes were enriched relative to rainfall, whereas Na, NO_3–N, NH_4–N, PO_4–P and SO_4–S were depleted. Total annual throughfall nutrient inputs (Ca, Cl, K, Mg, Na, NH_4–N, NO_3–N, PO_4–P, SO_4–S) were 14 kg $ha^{-1}yr^{-1}$ under *Cupressus*, 21 kg $ha^{-1}yr^{-1}$ under the natural forest and 24 kg $ha^{-1}yr^{-1}$ under *Eucalyptus*. Water passing through the different forest floors differed only in K, Mg and NO_3–N concentrations, the latter two being higher under the natural forest and *Eucalyptus* plantation than *Cupressus*. Potassium was greater under *Eucalyptus* than the natural forest and *Cupressus*. Except for NH_4–N in the natural forest, forest floor leachate nutrient concentrations were enriched in all forest types in relation to throughfall. Most nutrient fluxes to the mineral soil decreased in relation to throughfall fluxes, whereas NO_3–N fluxes increased by over 50% in all forest types. At all soil depths, the concentrations of most nutrients in the mineral soil solution decreased relative to the concentrations in the forest floor leachate, but Mg, Na and NO_3–N at all depths in *Cupressus* plantation and SO_4–S and Na at some soil depths in the natural forest and *Eucalyptus* plantation had increased. The vertical trends in soil solution nutrient concentrations showed a decreasing trend with depth increments for most of the nutrients, but the concentrations of Cl and Na in all forest types and Ca, Mg and NO_3–N in *Cupressus* increased with increasing soil depth. At 1 m soil depth, the concentrations of Ca, Mg and NO_3–N in *Cupressus*, respectively, were 8, 7 and 23 times higher than in the natural forest and 3, 4 and 81 times higher than in *Eucalyptus* indicating losses by leaching. Generally, the results of this study emphasize the importance of forest type, species composition and management in affecting carbon and nutrient storage, water and nutrient fluxes and dynamics.

ZUSAMMENFASSUNG

Im Munesa-Wald, Südostäthiopien, wurden die Auswirkungen der Umwandlung von Naturwald in Pflanzungen mit unterschiedlichen ausländischen Baumarten bzw. in Ackerland untersucht. Die Zielsetzung war, (i) die Änderungen in bodenphysikalischen und -chemischen Eigenschaften zu ermitteln, (ii) die Wasser- und Nährstoffflüsse in den unterschiedlichen Waldökosystemen zu quantifizieren und (iii) die Nährstoffdynamik im Bodenwasser der unterschiedlichen Waldökosysteme zu beurteilen. Bodenproben wurden von der organischen Auflage und vom Mineralboden in 0–20, 20–40, 40–70 und 70–100 cm Tiefe genommen. Freiland- und Bestandesniederschlag wurden mit Kunststofftrichtern gesammelt, die 1 m über dem Boden angebracht waren. Bodenlösungen wurden mit freidränenden (organische Auflage) bzw. Unterdruck-Lysimetern (Mineralboden in 20, 50 und 100 cm Tiefe) gewonnen. In 26 Jahren Ackerbau verschlechterte sich die Struktur des Oberbodens (0–20 cm) und die Gehalte an organischem Kohlenstoff (SOC) und Stickstoff in Gesamtboden und wasserstabilen Aggregaten nahmen beträchtlich ab. Unter einer 21-jährigen *Eucalyptus*-Pflanzung konnten keine signifikanten Änderungen dieser Parameter festgestellt werden, aber signifikante Abnahmen von organischem Kohlenstoff, Stickstoff und Schwefel traten in der Sand- und Schluffffraktion auf. Auch zeigten sich signifikante Minderungen in Qualität und Quantität der partikulären organischen Substanz (POM) infolge von Ackerbau bzw. nur in der Qualität der POM in der 21-jährigen *Eucalyptus*-Pflanzung. Die Masse der organischen Auflage unter einer 21-jährigen *Pinus patula*-Pflanzung, einer 21-jährigen *Eucalyptus globulus*-Pflanzung und unter *Eucalyptus globulus* in der dritten Rotation (angelegt vor 42 Jahren) nahm gegenüber dem Naturwald um 43%, 57% bzw. 15% ab. Auch die Vorräte an organischem Kohlenstoff und Stickstoff in der Auflage zeigten signifikante Abnahmen (9–60% bzw. 25–68%), am meisten unter *Pinus* und am wenigsten unter *Eucalyptus* in der dritten Rotation. Im Mineralboden bis 1 m Tiefe gab es eine signifikante Abnahme von 16–20%

(P<0,05) im SOC-Vorrat nach der Umwandlung des Naturwalds in Pflanzungen. Die N-Vorräte unter der 21-jährigen *Pinus*-Pflanzung und der *Eucalyptus*-Pflanzung in der dritten Rotation waren signifikant um 27 bzw. 20% reduziert, während die 21-jährige *Eucalyptus*-Pflanzung nahezu den gleichen N-Vorrat aufwies wie der Naturwald, wahrscheinlich aufgrund des dichten, N-fixierenden Unterwuchses. Die Veränderungen der Schwefel-Vorräte in organischer Auflage und Mineralboden nach dem Anlegen der Pflanzungen waren nicht signifikant.

Vom gesamten Jahresniederschlag während der Messperiode (1190 mm von Oktober 2001 bis September 2002) wurden etwa 47% durch das Kronendach von *Cupressus* und Naturwald bzw. 18% von *Eucalyptus* zurückgehalten. Die gesamte jährliche Deposition von Nährstoffen (Ca, Cl, K, Mg, Na, NH_4–N, NO_3–N, PO_4–P, SO_4–S) im Niederschlag betrug 12 kg ha^{-1} $Jahr^{-1}$. Die Flüsse von K, Mg, Ca, und Cl im Bestandesniederschlag waren höher als im Freilandniederschlag, die von Na, NO_3–N, NH_4–N, PO_4–P und SO_4–S dagegen niedriger. Die gesamten jährlichen Nährstoff-Einträge (Ca, Cl, K, Mg, Na, NH_4–N, NO_3–N, PO_4–P, SO_4–S) mit dem Bestandesniederschlag betrugen 14 kg ha^{-1} $Jahr^{-1}$ unter *Cupressus*, 21 kg ha^{-1} $Jahr^{-1}$ unter dem Naturwald und 24 kg ha^{-1} $Jahr^{-1}$ unter *Eucalyptus*. Das Sickerwasser aus den verschiedenen Auflagen unterschied sich nur in den Konzentrationen von K, Mg und NO_3–N, wobei die beiden letzteren unter Naturwald und *Eucalyptus* höher waren als unter *Cupressus*. Kalium war unter *Eucalyptus* höher als unter Naturwald und *Cupressus*. Die Nährstoff-Konzentrationen im Auflagen-Sickerwasser waren im Vergleich zum Bestandesniederschlag in allen Waldtypen erhöht mit Ausnahme von NH_4–N im Naturwald. Die Flüsse in den Mineralboden waren für die meisten Nährstoffen niedriger als die Flüsse mit dem Bestandesniederschlag, während die von NO_3–N in allen Waldtypen um über 50% höher waren. Die Konzentrationen der meisten Nährstoffen waren in den Mineralbodenlösungen aller Tiefen gegenüber dem Auflagen-Sickerwasser vermindert, die von Mg, Na und NO_3–N aber in allen Tiefen unter *Cupressus* und die von SO_4–S und Na in einigen Bodentiefen unter Naturwald und *Eucalyptus* erhöht. Der vertikale Verlauf der Nährstoffkonzentrationen in den Bodenlösungen zeigte eine Abnahme mit den Tiefenstufen für der meisten Nährstoffen. In allen Waldtypen nahmen aber die Konzentrationen von Cl und Na mit der Tiefe zu, in der *Cupressus*-Pflanzung auch die von Ca, Mg und NO_3–N. In 1 m Bodentiefe unter *Cupressus* waren die Konzentrationen von Ca, Mg und NO_3–N um den Faktor 8 bzw. 7 bzw. 23 höher als unter Naturwald und um den Faktor 3 bzw. 4 bzw. 81 höher als unter *Eucalyptus* und wiesen somit auf Auswaschungsverluste hin. Insgesamt unterstreichen die Ergebnisse dieser Studie die Bedeutung von Waldtyp, Artenzusammensetzung und Wirtschaftsweise für die Kohlenstoff- und Nährstoff-Speicherung, die Wasser- und Elementflüsse sowie die Nährstoff-Dynamik.

1. GENERAL INTRODUCTION

1.1 Socio-economic setup

Ethiopia is located between 3°N and 15°N, and 33°E and 48°E and covers an area of about 1130 000 km^2 (FAO, 2003). It has diverse topographic features with high mountains, deep gorges, flat-topped plateaus, and rolling plains. The altitude ranges from the highest peak at Ras Dejen (4620 m) down to the Dallol depression (110 m below sea level). The physical conditions and variations in altitude have resulted in a great diversity of climate, soil and vegetation (Asrat Abebe, 1992). Ethiopia's population is estimated at 67 million (MoFED, 2002) with an annual growth rate of 3 percent (MEDaC, 2001). The Ethiopian economy is highly dependent on agriculture, which accounts for 50 percent of the gross national product and contributes to more than 88 percent of exports and 85 percent of employment (CSA, 1999). The agricultural sector is dominated by the subsistent smallholder farmers, which contributes 95 percent of the agricultural production, and pastorals with a nomadic form of production. The country also has the largest livestock population in Africa (Mengiftu, 2002). About 88 percent of the human population and 70 percent of the total cattle population live in the highlands (above 1500 m) which make up 44% of the total land area (Hurni, 1988; Asrat Abebe, 1992, EFAP, 1993), making it the most densely populated agricultural areas in Africa (Anonymous, 2004). This has placed high pressure and a greater burden on the vulnerable land, forest and soil resources.

1.2. Rationale and research problem

In historic times, Ethiopia was believed to be extensively covered with dense forests. Over the last few hundred years, however, human actions have caused the country's forest cover to shrink significantly (von Breitenbach, 1962; EFAP, 1993). Documented evidences on the original extent of forest prior to human impact are scarce, but scientists estimate the losses by looking at remnant scattered trees as well as by using knowledge of the soil, elevation, and climatic conditions required by forests where forest could potentially exist if it were not for human actions. Comparing this "potential" forest area with the existing forest cover, Evans (1982) has estimated historical forest losses to be 36% since 1850. The major cause for the disappearance of forests is rapid population growth leading to extensive forest clearing for cultivation and grazing, exploitation of forests for fuel wood and construction material (EFAP, 1993, 1994). The destruction of forests has widespread implications for all mankind and has wider implications of global importance (Redhead and Hall, 1992), but is clearly of most immediate importance to rural populations living in and near the forest areas. The consequences are very severe; the cumulative results are shortage of wood and ecological imbalance, manifestations of which are noticed in recurrent droughts, reduced water resources, extinction of flora and fauna and heavy soil erosion. It is estimated that the country is loosing over 2 billion tons of fertile top soil every year, most of it from the highlands, as a result of soil erosion by water (FAO, 1986). This has resulted in a massive environmental degradation and serious threat to sustainable agriculture and forestry.

In the last few decades, large areas of forest plantations (*ca*. 200,000 ha), predominantly exotic species (*Eucalyptus* spp., *Cupressus lusitanica* and *Pinus* spp.) have been established to satisfy the growing wood demands of the population and to rehabilitate degraded lands (Pohjonen, 1989; EFAP, 1994; FAO, 2003). Also the fast growing nature of exotic species and favourable economic returns from tree plantations have encouraged the conversion of slow-growing and low-productive secondary natural forests into plantations. Recent estimates of the distribution of forest and woodland areas made by FAO (2001) indicated that about 4.2% of the land is covered by forests and the areas under planted forests are small (about 0.2%) compared with the size and needs of the population. The remaining natural forests are, therefore, under constant pressure from rising population in the wake of expansion of agricultural land and widening gap between demand and supply of forest products (EFAP, 1994). The current rate of deforestation is estimated to be 0.8% per year while the current expansion of planted forests is about 0.18% per year (FAO, 2001) which does not compensate for the loss of natural forests. There is no prospect of an early end to the pressures causing the clearing of the scarce forest resources to agricultural use, and cutting for fuelwood will continue. The challenge is not to prevent these activities but to manage them. The aim must be to ensure that wood and other forest products are harvested sustainably and that the subsequent land uses are productive and sustainable. Management of fast-growing and high-yielding short rotation plantations, with long-term stability of soil fertility and nutrient balance, to sustain high biomass production and quality of the environment is an important challenge.

The future of Ethiopia is linked with the judicious and efficient management of its natural resources and restoration of its environment. Although intensive management of exotic tree species may provide rapid growth and a higher economic return than would native tree species, little is known about the environmental impacts of this practice, such as on hydrology, soil quality and long-term productivity. The conversion of natural forest ecosystems into cultivation and monoculture plantations can change the nutrient cycling processes through changes in plant cover and species composition owing to differential patterns among plant species in litter production and turnover and nutrient accumulation (Gosz, 1981; Brown and Lugo, 1990; Lugo, 1992). Frequent harvesting of forest plantations result in long-term decline in soil organic carbon (SOC) and nutrient content due to disruption of the flow of carbon and nutrients through litter, removal of large amounts of nutrients from the soil through biomass and also losses by erosion and leaching (Zech and Drechsel, 1998). Human-induced land-use changes are known also to affect the spatial and temporal patterns of landscape water fluxes (Bosch and Hewlett, 1982) because forest stands of different tree species differ in their aboveground vegetation surface area, stand structure and morphology, and can have a differential impact on rain water interception and evapotranspiration losses, hence, on soil water regimes (Pritchett, 1979; Cape et al., 1991). For example Swank and Douglass (1974) in the United States found that streamflow was reduced by 20% by converting a deciduous hardwood stand to a *Pinus strobus* L. plantation.

Previous investigations on the effects of plantations on soil properties in Ethiopia have focused on changes to solid phase soil properties (Michelsen et al., 1993; Betre et al., 2000; Lemenih et al., 2004). These studies generally indicate that the changes in soil properties after plantation establishment are species specific. Moreover, to date, studies on the hydrology of forest ecosystems in Ethiopia have not been conducted. Nutrient cycling within ecosystems forms the major source of nutrients for plant use and nutrient inputs from the atmosphere are important to the long-term development of soils and ecosystems (Binkley, 1986). The input of nutrients from the atmosphere and the dynamics of nutrients in soil solution, which are an important aspect in studying nutrient cycling in forest ecosystems, are only beginning to be investigated in Ethiopia. In contrast to bulk soil properties, which are typically slow to respond to a change in land-use, soil solution chemistry is often a sensitive indicator of biogeochemical processes in forests responding quickly to various changes and may provide an early indication of the long-term changes in soils associated with land-use changes (Ranger et al., 2001; McDowell et al., 2004). Studies of solute concentrations and fluxes through forest ecosystems have been conducted mainly in North America (Likens et al., 1977) and Europe (Ulrich, 1983; Gundersen et al., 1998; De Vries et al., 2003) with greater risk of air pollution (Krupa, 2002). However, even in the absence of air pollution risks, such studies are also of critical importance because of the potential ecological significance of atmospheric depositions in forest ecosystems nutrient cycling and the need for such information to make reliable forest management decisions.

2. OBJECTIVES

The overall objective of the study was to determine the effect of land-use changes on soil properties and understand ecosystem specific hydrological and biogeochemical processes under the different forest ecosystems. The specific objectives were (i) to assess the effect of natural forest conversion on soil physical and chemical properties, (ii) to quantify water and element fluxes under the different forest ecosystems, and (iii) to assess nutrient dynamics in water flowing through the forest floor and mineral soil under the different forest ecosystems.

3. MATERIALS AND METHODS

3.1. Location and general description of the study area

The Munesa Shashemene forest (7°34´N and 38°53´E; 240 km south east of Addis Ababa) is located in the eastern escarpments of the central Ethiopian rift valley within the Bale/Arsi highlands massif (Fig. 1). The Munesa Shashemene forest consists of three branches, namely Degaga, Gambo and Sole. The forest cover at Degaga, where this study was conducted, comprises 8527 ha of disturbed natural forest and 2518 ha of forest plantations. The altitude ranges from 1500 m in the foothills to 3500 m at the peak. The climate is sub-humid with a long-term mean annual rainfall of 1250 mm and mean annual temperature of 19°C (Solomon et al., 2002). The distribution of rainfall is bimodal, most of it falling during the main rainy season (June to September) with peaks in July and August, and small rains from February to May. Generally, mean annual rainfall increases and mean annual temperature decreases with increasing altitude. Geologically, the area lies on tertiary volcanic deposits and the soils developed from these rocks are principally Nitisols (Anonymous, 2004). The topography and vegetation change rapidly with increasing altitude. Generally, vegetation varies from savannah and open woodland in the foothills at 1500 m to some disturbed forests and alpine vegetation closer to the peak (Müller-Hohenstein and Abate, 2004). The vegetation of the study area is described in detail by Abate (2004).

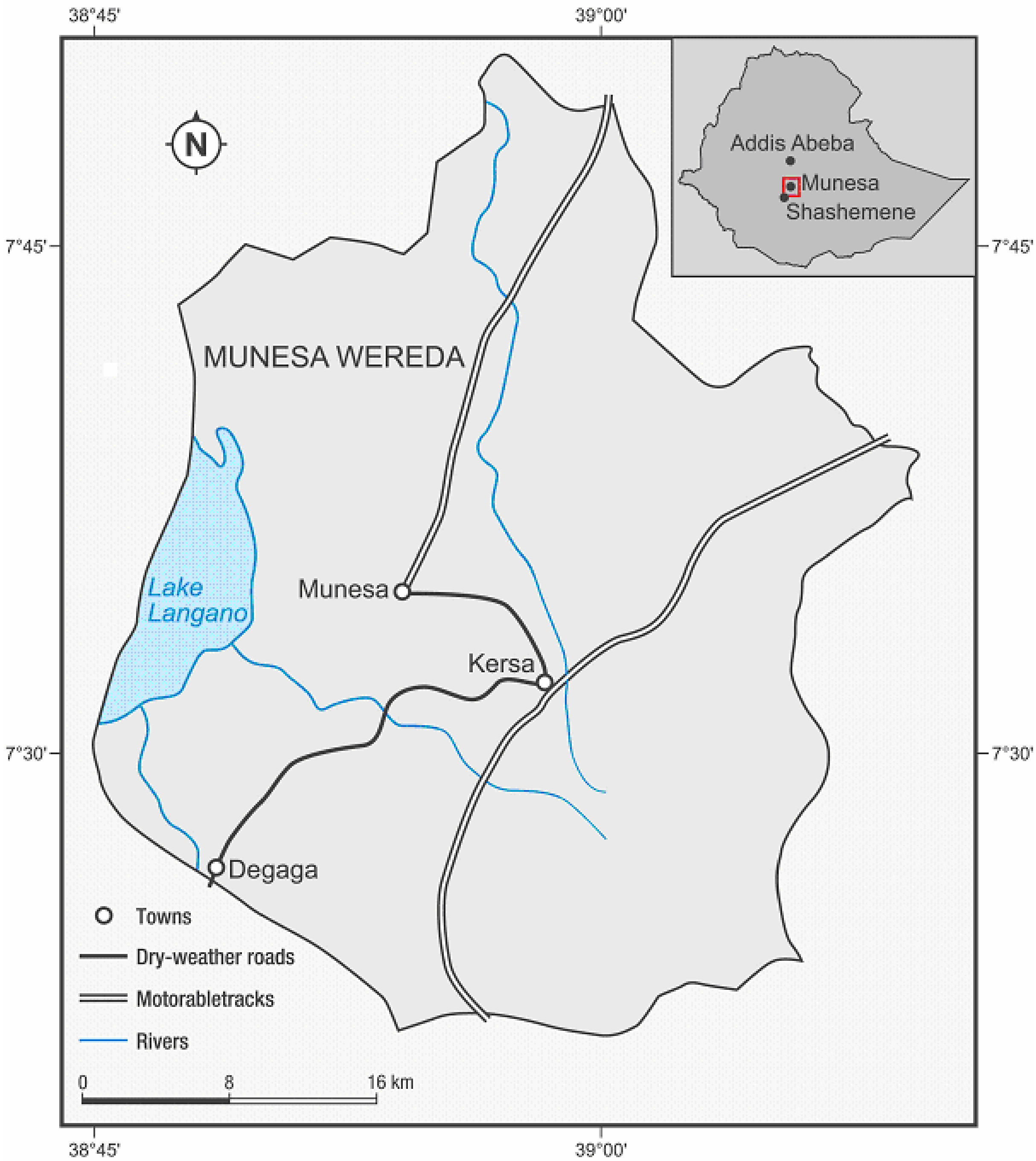


Figure 1. Map of the study area.

3.2. The studied forests and experimental design

Two monoculture exotic tree species plantations (*Cupressus lusitanica* and *Eucalyptus globulus*) and an adjacent natural forest were selected to undertake multidisciplinary (Ecophysiology, Geobotany and Soil Science) field investigations. The natural forest is dominated by old growth *Podocarpus falcatus* trees and other common medium sized canopy tree species include *Croton macrostachys*, *Olea hochstetterii* and *Scheffelera abyssinica*. The *Eucalyptus* plantation is sparsely stocked (595 trees ha^{-1}) relative to the *Cupressus* plantation (672 trees ha^{-1}) and has a native understorey canopy tree (*Croton* m*acrostachys*) and shrubs notably *Acanthopale pubescens*, *Achyrospermum schimperi*, *Bothriocline schimperi*, *Carex spicato-paniculata*, *Hypoestes forskaolli*. The forest floor in the natural forest and *Eucalyptus* plantation consists of dense grass and broad-leaved herbaceous species. The mean height of *Eucalyptus* is 30–40 m and the mean diameter at breast height (dbh) is 19–39 cm. The *Cupressus* plantation has almost no ground vegetation. The mean height of *Cupressus* is 18–20 m and dbh is 29 cm. In addition, two plantations (*Pinus patula*) and third rotation (*Eucalyptus globulus*) and an adjacent crop field were included to compare some soil related parameters with those in forests selected by the multidisciplinary research team. All of the plantations and the crop field were established after clearing of part of the existing natural forest at different time scales. The third rotation *Eucalyptus* was established in 1960 while all the other plantations were established in 1980. The crop field was established in 1975. The natural forest is approximately 3 to 4 thousand years old (Zech pers. communication). In each forest type and the crop field three 0.04–0.06 ha permanent plots were randomly located. In the two plantations (*Cupressus* and *Eucalyptus*), which were selected by the multidisciplinary research team, and the natural forest, about 20–25 m^2 of the area was fenced at the centre of each plot for the installation of field equipment. In addition a soil pit was excavated to the depth of 1.2 m within in each plot. Soil properties under the plantations and the crop field prior to their establishment were assumed to have been similar to those under the natural forest.

3.3. Equipment

An automatic weather datalogger was placed in a big opening between the natural forest and the plantations. To monitor water and nutrient dynamics, rain water collectors were placed at three locations (three collectors per location) close to the automatic weather data logger in the open area. Within the fenced areas of the permanent experimental plots of each forest, throughfall collectors (five per plot) were placed around the sample tree at a distance of 0.8 to 1 m from the trunk. Rainfall and throughfall were collected using plastic funnels of 12 cm diameter and 2 l capacity mounted 1 m above the ground. Table tennis balls were put inside each collector to prevent loss of water by evaporation. In addition, tension and zero-tension lysimeters and tensiometers (each of them three per plot) were installed. The zero-tension lysimeters made of plastic boxes (0.15 x 0.15 m) were placed horizontally in the contact zone between the forest floor and the mineral soil. The boxes were connected to a 2 l bottle placed in a soil pit. To avoid any solid material entering the boxes and bottles, a fine wire mesh (0.5 mm) was attached to the upper part of each plate. Tension lysimeters and tensiometers were installed at three depths (0.2, 0.5 and 1 m below soil surface). The three suction cups per depth and per plot were connected to one collecting bottle. Tensiometers were placed approximately 0.5 m away from the suction lysimeters. All equipments were installed in May 2001.

3.4. Sampling and sample preparation

Soil samples were taken at 0–20, 20–40, 40–70 and 70–100 cm depths from the three sides of the pit. In addition, two 1 m^2 plots were marked randomly within each plot and samples were taken by auger at three points within the 1 m^2 area and mixed for the above mentioned depth classes. Soil samples were put in individual polyethylene bags, air-dried and passed through a 2-mm sieve. Samples for the mineral soil bulk density determination were taken by 100 cm^3

Aluminium cylinder at seven points for each soil depth. Sampling of the organic layer (3 samples per plot) was done by pressing a 30 x 30 cm steel sheet sampling frame into the organic layer. The surrounding organic matter was removed leaving a block of the organic layer in which the litter (L) and fermentation (Of) horizons were identified and the thickness of the different horizons was measured with a ruler. The materials (excluding woody debris > 2 cm) from the different horizons were put in separate paper bags. The organic layer samples were dried in an oven at 65 °C and weighed. After drying, the three samples of each plot were mixed and the final number of samples was reduced to three.

Rainfall and throughfall water and litter leachates were sampled from October 2001 to September 2002. Mineral soil solutions were sampled only during the main rainy season (June to September). Samples retrieved during June to September 2001 were discarded to allow ions on the exchange surfaces of the ceramic to equilibrate with the soil solution. Samples collected during the main rainy season of 2002 were used for chemical analysis. Soil solution samples were taken by applying vacuum produced by vacuum pumps based on the tensiometer readings at each soil depth. Sampling was done on a weekly basis and during sample collection the volume of water was registered. After each collection, the collectors were washed with deionized water or with a portion of the sample water. On each sampling day, water samples were transported to the storage facility and kept frozen. All samples were transported in cool boxes to Germany for chemical analysis. Solution samples were filtered through 0.45 μm glass fibre filters (Schleicher & Schuell). After filtration, samples from the rainfall and throughfall collectors and zero-tension lysimeters in one plot were proportionally bulked per source per plot prior to chemical analysis, yielding one sample per sampling day. The dried samples of the organic layers and mineral soil horizons were finely ground with a rotary ball mill for chemical analysis.

3.5. Laboratory analysis

3.5.1. Soil particle and aggregate-size fractionation

Air-dried and sieved (2 mm mesh) 30 g samples were put in a centrifuge tube and dispersed ultrasonically at a soil: water ratio of 1:5 (w/v), with an energy input of 60 J ml^{-1}using a probe type sonicator (Branson Sonifier W−450). Coarse sand fraction (250−2000 µm) was separated by wet sieving, and the remaining material in the <250 µm fraction was further sonicated at a soil: water ratio of 1:10 (w/v), with an energy input of 440 J ml^{-1}. The clay-size separates (< 2 µm) were isolated from the silt (2−20 µm) and fine sand (20−250 µm) by repeated centrifugation, while the silt-size separates were isolated from the fine sand by wet sieving. After fractionation, the different particle-size fractions were dried at 50 ^{o}C.

The size distribution of aggregates was measured by wet sieving through a series of sieves (2, 1, 0.5, 0.25 and 0.053 mm) following the procedures of Cambardella and Elliott (1993). A 70–80 g sample of air-dried soil passed through a 5 mm sieve was spread on the top of a 2 mm sieve submerged in a bucket of deionized water. The water level was adjusted so that the aggregates on the sieve were just submerged. Soils were left immersed in the water for 10 min and then sieved by moving the sieves 3 cm vertically 50 times during a period of 2 min. During the sieving process, floatable materials >2 mm were removed and discarded. According to Six et al. (1998) materials > 2mm are not considered an integral part of SOM. The material remaining on the 2 mm sieve was transferred to a glass pan. Soil plus water that passed through the sieve were poured onto the next finer sieve and the processes were repeated, but floatable materials were not removed and discarded. The different aggregate sizes were dried in the oven at 50 °C overnight for chemical analysis.

3.5.2. Separation of particulate organic matter (POM)

The separation of POM followed the procedure of Six et al. (1998). Prior to POM separation, the fractions in the >0.25 mm size aggregates were bulked as macroaggregates and the 0.053–0.25 mm size as microaggregates. After the aggregates were dried (105 °C) in the oven overnight and cooled in a desiccator to room temperature, about 10 g of each aggregate fraction was taken in a conical centrifuge tube and suspended in 35 ml sodium polytungstate (adjusted to a density of 1.8 g cm^{-3}) by hand shaking. The suspension was allowed to stand for 20 min before centrifugation at 1250 rpm for 60 min. After centrifugation, the floating material was collected on filters and rinsed thoroughly with deionized water to remove sodium polytungstate, this material is referred to as free light fraction (LF). The heavy fraction remaining in the tube was washed twice with 50 ml deionized water and dispersed in 50 ml of 5% sodium hexametaphosphate by shaking in a reciprocal shaker for 18 hours. The dispersed heavy fraction was rinsed through a 0.053 mm sieve with deionized water. The material remaining on the sieve is intra-particulate organic matter (iPOM) + sand. Both the free LF and iPOM were dried in the oven at 50 °C overnight. The dried subsamples from each aggregate size class, particle size fraction, the free light fraction, and iPOM were finely ground in a rotary ball mill for chemical analysis.

3.5.3. Chemical analysis

Organic C, N and S concentrations in bulk soil, size fractions and POM were determined using a CHNS–analyzer (Vario EL, Elementar Analysensysteme GmbH, Hanau, Germany). The pH_{KCl} (soil:solution ratio 1:2.5) of the soil was determined with a standard pH electrode (Orion U402–S7). Bulk density was determined after drying a defined volume of soil in an oven at 105°C. Solutions were analysed for pH, total content of Ca^{2+}, K^+, Mg^{2+}, Na^+ (plasma emission spectroscopy, ICP-AES, Integra XMP), and Cl^-, NO_3^-, NH_4^+, PO_4^{3-}, SO_4^{2-} (ion chromatography, Dionex 2000i-SP). Detection limits (mg l^{-1}) were: 0.025 for NH_4^+, 0.2 for

Ca^{2+}, Na^{+} and Mg^{2+}, 0.25 for K^{+}, 0.27 for Cl^{-}, 0.34 for NO_3^{-}, 0.28 for PO_4^{3-} and 0.32 for SO_4^{2-}.

3.6. Calculations and data analysis

Element stocks (kg m^{-2}) were calculated as a product of bulk density, depth of sampling and element's concentration per unit of soil samples (Guo and Gifford, 2002).

$$C = BD \times C_c \times D/10 \quad (1)$$

where BD is the soil bulk density (g cm^{-3}), Cc (%) the soil element concentration, and D is the soil sampling depth (cm).

The mean weight diameter (MWD) of water stable aggregates was determined as the sum of the percentage of soil on each sieve multiplied by the mean intersieve diameter of adjacent sieves (Haynes, 1999).

$$MWD = \sum \text{(percent of sample on sieve x mean intersieve size)} \quad (2)$$

All calculations for a particular parameter in rainfall, throughfall and litter leachate of each season, i.e. dry season (October–January), small rainy season (February–May) and main rainy season (June–September) were based on mean values of three plots per forest type. Volume weighted concentrations (VWMC) and fluxes of elements in rainfall, throughfall and litter leachate for a given season were estimated from the paired measurements of element concentration and rainfall, throughfall and litter leachate volume in each plot (Tobon et al., 2004).

$$VWMC_i = \sum^{n}_{j=1} C_{ij} \,.\, TF_j / \sum^{n}_{j=1} TF_i \quad (3)$$

where C_{ij} is the i-element concentration in throughfall on the j-collection day, TF is the total throughfall water volume and n is the total number of sampling dates. The same procedure was used for rainfall and litter lecheate element concentrations. Using rainfall, throughfall and litter leachate water volume, concentrations were converted into gram quantities of various nutrients for each season and summed to yield annual inputs. Canopy exchange (i.e. canopy leaching and canopy uptake) was calculated as the difference between throughfall flux of a particular element and its atmospheric deposition to the rain collectors.

Data for each parameter in rainfall, and throughfall and litter lecheate of the different treatments were assessed using MSTAT–C version 2.10 statistical package. Differences between and among treatment means were considered significant at $P \leq 0.05$. Correlation analysis was conducted between pairs of elements in rainfall, throughfall, litter leachate and soil solution, and rainfall, throughfall and litter leachate volume and element concentrations.

4. RESULTS AND DISCUSSION

4.1. Soil physical and chemical properties

4.1.1. Soil aggregation

Clearing of the natural forest and reforestation with *Eucalyptus* did not significantly affect the distribution of water-stable aggregates (WSA), but after 26 years of continuous crop cultivation, the amount of water-stable macroaggregates was significantly reduced from > 70% in the natural forest soil to 50% in the cultivated soil, indicating that cultivation resulted in the structural degradation of this soil (Table B1 & Table A4). In the two forest types, 87–90% of the total soil mass remained as water-stable aggregates with >74% as macroaggregates (> 0.25 mm), and 14−17% as microaggregates (0.05−0.25 mm). In contrast, in the cultivated soil, significantly large proportion of the soil was retained as microaggregates and small macroaggregates (0.25–0.5 mm). This could be attributed mainly to the breakdown of aggregates by tillage and differences between the two land use types in annual organic matter input which gives cementing agents. These results confirm earlier observations that macroaggregates are dynamic in nature and the size distribution of macroaggregates is affected by the change in land use and management (Dormaar, 1983;

Elliott, 1986; Beare et al., 1994; Puget et al., 1995; Spaccini et al., 2001). The effect of cultivation was much more evident in the larger macroaggregates (>1mm) than the smaller macroaggregate size classes (Table B1). The >2 mm and >1 mm classes of the natural forest soil were 13 and 4 times, respectively, larger than in the cultivated soil.

The relatively higher reduction in larger macroaggregates compared to the smaller aggregates upon cultivation could be mainly due to the fact that the former are largely dependent on live and decaying plant roots and fungal hyphae and probably casts of earthworms and termites which are rapidly destroyed by tillage (Tisdall and Oades, 1982). A greater shift in water-stable aggregates from large macroaggregates to smaller macroaggregates and microaggregates upon cultivation had also led to a significant reduction of MWD from 0.92 mm in the natural forest soil to 0.36 mm in the cultivated soil (Table B1). Spaccini et al. (2001) reported MWD reductions of 37 to 76% for cultivated Ethiopian Vertisols, Alfisols, Entisols, and Andisols relative to the forest soil, being highest in Vertisols and lowest in Andisols.

4.1.2. Total SOC, N and S concentrations in particle- and aggregate-size fractions

Conversion of the natural forest into a monoculture *Eucalyptus* plantation 21 years ago resulted in the depletion of mean SOC concentrations of sand and silt fractions, and N and S concentrations of the sand fraction (Table A2). The coarse sand fraction showed the highest losses of all three elements, suggesting that organic matter associated with the coarser fractions is more labile and the first to be affected by changes in land-use and soil management (Christensen, 1996; Solomon et al., 2002; Zinn et al., 2002). The loss of OC was larger than the losses of N and S. Mean C/N and C/S ratios of all the particle-size fractions and N/S ratio of the clay fraction were also significantly narrowed after conversion of the natural forest into *Eucalyptus* plantation (Table A3). In both forest types, the C/N and C/S ratios of the coarse and fine sand and silt fractions were higher than in the bulk soil, whereas those of clay were lower (Tables A1&A3). This might be due to the more aliphatic and humified nature of the clay-associated organic matter (OM) in comparison to the OM in the bulk soil and coarser fractions (Buyanovsky et al., 1994; Mahieu et al., 1999).

In the two forest types, C, N and S concentrations were not significantly different among the different aggregate size fractions (Fig. A1). In contrast, in the cultivated soil, the OC and N concentrations were significantly different among the different size classes, and appeared to decrease as size increases from 0.053 to 2 mm diameter (Table B3). This could be attributed partly to the redistribution and / or transfer of organic matter from the large aggregates to smaller ones either in the process of biodegradation or by mechanical disruption of the large macroaggregates (Dormaar, 1983; Christensen, 1992). Conversion of the natural forest into *Eucalyptus* plantation did not significantly affect the OC, N and S concentrations associated with each water-stable aggregate size class. However, although the differences generally are not statistically significant, the OC and N concentrations associated with each maroaggregate-size class in the natural forest were 2–3 times higher than the corresponding values in the cultivated soil (Table B3).

The average C/N ratios of the larger aggregates (> 0.5 mm) were significantly wider in the soil under natural forest than in the soil under *Eucalyptus*, whereas C/S and N/S ratios were not different between the two forest types (Fig. A2). In the cultivated soil, C/N ratios of the different aggregate-sizes were not significantly different from the natural forest soil aggregates, but the overall mean C/N ratio of the water-stable aggregates was significantly narrowed from 11 in the natural forest soil to 9 in the cultivated soil. The mean C/N, C/S and N/S ratios of the aggregates in both forest types and C/N ratio in the cultivated soil were nearly the same as those of the corresponding bulk soil (Table A1, Fig. A2 and Tables B2& B3).

4.1.3. Free LF and iPOM C, N and S concentrations associated with soil aggregates

Conversion of the natural forest into a monoculture *Eucalyptus* plantation significantly reduced the free LF C associated with both aggregate-sizes and N associated with macroaggregates (Table A5). The iPOM C, N and S associated with macroaggregates and S associated with microaggregates below *Eucalyptus* were also significantly reduced relative to the natural forest soil (Table A6). Cultivation of the natural forest soil for 26 years also significantly reduced the mean C and N concentrations in both the free LF and iPOM fractions (Table B5). The effect of cultivation was more pronounced on the iPOM C than on the free LF C concentration. Similarly, although the *Eucalyptus* plantation had nearly the same level of soil aggregation (Table A4) as in the natural forest, the losses in iPOM C and N concentrations were more pronounced than losses from the free LF. This could be due to (i) the input of organic material to the LF material from the previous crop and year-round input of litter from the plantation and (ii) gaseous losses of OM inside the aggregates caused by high fire temperatures during clearing and site preparation; otherwise biodegradation is normally nearly three times as fast outside aggregates as within them (Besnard et al., 1996) and in addition deterioration of aggregation in the cultivated soil was another reason. According to Jastrow (1996) and Six et al. (1998), the amount of total occluded POM C and nutrients per unit soil is mainly a function of aggregation, whereas the free light POM C i.e., LF C is mostly affected by residue input. Buschiazzo et al. (2001) linked the large decrease of OC after cultivation of a forest soil to the occurrence of natural fire before cultivation.

The effect of changes in land use was more drastic on macroaggregate-associated POM C, N and S than on POM associated with microaggrates (Tables A5&A6 and Table B5). This confirms the conclusions of several authors (Elliott, 1986; Gupta and Germida, 1988; Besnard et al., 1996) that organic matter associated with macroaggregates is more labile than organic matter associated with microaggregates. Jastrow (1996) found relatively higher proportions of

POM C inside the macroaggregates of a virgin prairie soil compared to corn field and restored prairie soil. Six et al. (1998) also reported higher iPOM levels in water-stable macroaggregates sampled from native sod soil than those from cultivated soil. Overall, the results showed that the effect of conversion of the natural forest into tree plantation and cultivation was more pronounced on the POM C and N than those observed in the whole soil and in water-stable aggregates, indicating that POM constitutes soil organic matter fraction more sensitive to the effects of land-use change and soil management.

4.1.4 Dry mass accumulation, and SOC, N and S storage

The effect of clearing and reforestation of the natural forest soil with different plantation species significantly influenced the accumulation of dry mass in the organic layer (Table C3). Organic layer mass was highest under the natural forest followed by third rotation *Eucalyptus* and lowest under *Pinus*. The reductions in average litter mass after clearing and replacement of the natural forest ranged from a low of 6.4 t ha^{-1} (–14%) under third rotation *Eucalyptus* to a maximum of 24.2 t ha^{-1} (–57%) under *Pinus* (Table C3). Such variations in the organic layer mass accumulation may be due to differences in rate of litter production, litter quality, age and species composition. The greatest mass under third rotation *Eucalyptus* compared to the other two plantations is due to the accumulation of litter after each harvest and differences in time since establishment. Zinn et al. (2002) reported an increase in litter mass after conversion of native *Cerrado* to *Pinus* and a decrease after conversion to *Eucalyptus* for sub humid site conditions in Central Brazil.

Clearing and replacement of the natural forest by tree plantations significantly affected the organic layer C concentration, being greater under third rotation *Eucalyptus* compared to the natural forest and 21 years *Eucalyptus*, but differences between *Pinus* and the other forest types were not significant. Total N and S concentrations were higher under 21 years *Eucalyptus* in comparison to the other two plantation species but were not different from the natural forest (Table C2). The C/N ratios under *Pinus* and third rotation *Eucalyptus* organic layers were significantly ($P<0.01$) higher than under the natural forest, but 21 years *Eucalyptus* had an equivalent C/N ratio as the natural forest probably due to the presence of N-fixing plants in the understorey vegetation. From the ecological point of view, litter with high nutrient contents or low C/N ratio play an important role in plantation forestry because rather than immobilising nutrients it releases them for rapid recycling (Lugo et al., 1990).

The mean C, N and S stocks of the organic layers under the different forest types ranged from 6.5–16.4 t ha^{-1}, 0.3–0.7 t ha^{-1} and 0.03–0.1 t ha^{-1}, respectively, (Table C3). Wilcke et al. (2002) reported 103, 5.53 and 0.77 t ha^{-1} C, N and S stocks, respectively, in the tropical montane rainforest of Ecuador. Higher litter mass accumulation in the natural forest and third rotation *Eucalyptus* resulted in a significantly ($P<0.01$) higher C and nutrient storage in comparison to the other two plantation treatments. C stock under the natural forest (16.4 t ha^{-1}) was found to be significantly reduced by 8.2 t ha^{-1} (–50%) and 9.9 t ha^{-1} (–60%) after conversion to 21 years *Eucalyptus* and *Pinus* plantations, respectively, while third rotation *Eucalyptus* had an equivalent amount of C (15 t ha^{-1}) as the natural forest probably due to the greater amount of litter accumulated after each harvest. Like that of C, the reductions in N and S stocks under *Pinus* were much higher followed by 21 years *Eucalyptus* (Table C3).

There were no considerable variations in the mineral soil mean bulk densities to the depth of 1 m among the different forest types (Table C4). Mean SOC concentration of the mineral soil under the natural forest was significantly higher than under the 21 years *Eucalyptus* and *Pinus* stands. The natural forest and 21 years *Eucalyptus* had greater N concentration compared to third rotation *Eucalyptus* and *Pinus* stands, but there were no considerable differences between the former and the latter two. In the surface 20 cm soil layer, the natural forest and

21 years *Eucalyptus* had higher SOC, N and S concentrations compared to third rotation *Eucalyptus* and *Pinus* stands, but differences between the former two were not significant (Table C4). Below the 20 cm soil depth, except OC and N in the 20–40 cm layer, all forest types had nearly the same OC, N and S concentrations. Mean C/N ratio to the depth of 1 m under 21 years *Eucalyptus* (9) was significantly ($P<0.01$) lower than the C/N ratios under third rotation *Eucalyptus* (11), and under the natural forest and *Pinus* (12). In all forest types, with the exception of 21 years *Eucalyptus*, C/N ratio tended to decrease with increasing depth (Fig. C1) probably due to leaching of N-rich materials from the upper soil layers.

Average SOC stocks in the mineral soil horizons in this study ranging from 26.2–32.7 kg m^{-2} to 1 m depth (Table C5) were higher than the world average (11.7 kg m^{-2} to 1 m depth) based on the data of Eswaran et al. (1993) and several other authors (Brown and Lugo, 1982; Lugo et al., 1986; Brown and Lugo, 1990; Zinn et al., 2002). Differences between our study and others could be due to differences in soil forming factors, including climate, parent material, topography, vegetation, and human impact. Soil OC under the different plantations varied from 26.2–27.5 kg m^{-2} representing 80–84% of the SOC stock under the natural forest (32.7 kg m^{-2}) (Table C5). Since there is about three times as much C in the world's soils as in the atmosphere (Follett, 2001), the observed changes (–16 to –20%) in this pool can have considerable feed-back effects on the amount of CO_2 in the atmosphere and thereby on global warming.

Mean N stocks to 1 m soil depth (Table C5) differed among forest types ($P<0.01$), being highest under the natural forest and 21 years *Eucalyptus* plantation compared to third rotation *Eucalyptus* and *Pinus* stands, but both the former and the later two were not significantly different from each other. The reductions in N stocks relative to the natural forest varied from a maximum of 0.78 kg m^{-2} (–27%) under *Pinus* to a low of 0.04 kg m^{-2} (–1.4%) under third rotation *Eucalyptus* (Table C5). The changes in S stocks due to the transformation of the natural forest into different forest plantations were non-significant, being 0.05 kg m^{-2} (–13%) under *Pinus* and 21 years *Eucalyptus*, while there was a net gain of 0.02 kg m^{-2} (+5 %) under third rotation *Eucalyptus* (Table C5).

The distribution of SOC, N and S stocks across the profile (Table C5) tended to follow the general trend in SOC, N and S concentrations (Table C4), decreasing from the surface to the subsoil although bulk density values increased in this direction. Nearly one-third of the total SOC, N and S stocks to 1 m depth in all forest types (Table C5) were found in the surface 0–20 cm layer. This points out the need for proper management as it represents the pool most exposed to management effects that may accelerate its decomposition and release of CO_2 to the atmosphere. Surface soil OC, N and S stocks under the natural forest (Table C5) were not significantly different from the 21 yr *Eucalyptus*, however, the natural forest and 21 years *Eucalyptus* stored greater amounts of OC, N and S in the surface 20 cm depth compared to *Pinus* and third rotation *Eucalyptus*. For the depth 20–100 cm, treatment effects on SOC, N and S stocks were less clear, but the losses range from 2.73–5.89 kg m^{-2} (13–28%) for SOC, 0.21–0.4 kg m^{-2} (11–20%) for N and 0.06 kg m^{-2} (21%) for S (Table C5). This indicates that any conclusion based on surface soil responses to changes in soil OC and nutrients such as N and S that occurred after forest clearing is conservative.

4.2. Water and nutrient fluxes

4.2.1. Water flux

Total rainfall amount during the one year study period amounted 1190 mm (Table D1), lying very close to the past long-term value (1250 mm) from the nearby meteorological station (Solomon et al., 2002). There was a marked variation in the distribution of rainfall among the different seasons because in Ethiopia rainfall is mainly associated to a change in the predominant wind direction (monsoon); northeast winds prevail during the dry season and westerly to southwesterly winds during the rains (NMSA, 1996). Of the total annual rainfall, the highest amount (60%) fell during the main rainy season (June to September) and the minimum (12%) during the dry season (October to January) (Table D1). The monthly maximum and minimum rainfalls, respectively, were 67.4 and 6.2 mm in the dry season, 136.4 and 20.8 mm in the small rainy season and 268.2 and 120 mm in the main rainy season. Daily minimum rainfall was the same in all the three seasons (0.2 mm) while the daily maximum was variable: amounting 8.2 mm in the dry season, 39.2 mm in the small rainy season and 60 mm in the main rainy season. Of the 12 months, monthly rainfall was less than 100 mm from October to February and was above 200 mm only in August.

The proportions of annual incident rainfall that reached the forest floor were 82% under *Eucalyptus* and 53% under *Cupressus* and the natural forest (Table D1). This variation was mainly attributed to the difference in stand density and total canopy area, leaf morphology, branch geometry and hydrophobicity among species. However, the possibility of spatial variations in rainfall intensity within the study area could not be ruled out. In general, interception loss was highest during the dry season (65% in *Cupressus*, 63% in the natural forest and 32% in *Eucalyptus*) (Table D1) not only due to the pronounced sunny days before and after rain events, but also rainfall intensity for most of the rain events was very low (< 5 mm) to produce throughfall. During the monitoring period, throughfall water fluxes under the different forest types were generally less than rainfall (Table D1) which is expected since cloud water is not a factor. Throughfall values ranging from 62–88% have been reported for different montane tropical forests (Veneklaas, 1990, 1991; Cavelier et al., 1997; Schrumpf, 2004). In Brazil, Lilienfein and Wilcke (2004) found that throughfall was 75–85% of incident rainfall (1682 mm) under *Pinus caribaea* plantation. Variability in throughfall amount between different studies can be attributed in part to differences in climatic patterns, meteorological conditions, and stand density and species composition. In the Munesa forest, long sunny periods were common even during the wetter months and so there was usually plenty of time for the canopy to dry out.

4.2.2. Nutrient concentrations and fluxes

The volume weighted mean (VWM) nutrient concentrations in rainfall ranged from 0.09 mg l^{-1} for Mg to 3.29 mg l^{-1} for Na (Table D2). VWM concentration of NH_4–N was 1.78 times higher than that of NO_3–N. Rainfall at Munesa was weakly acidic (mean pH 6.7) with most of the potential acidity being neutralised by Na and Ca. On an equivalent basis, Na was accompanied by Cl and Ca. In all forest types, canopy interactions produced throughfall more alkaline than bulk precipitation (Table D2). The VWM nutrient concentrations in throughfall were dominated by K>Cl>Ca>Na>SO_4–S in all forest types. Throughfall nutrient concentrations were found to be consistently greater for the natural forest than for the two plantations although the differences for some of the nutrients were not significant (Table D2). This might have been caused by differences in dry deposition and canopy interception capacity which is a result of several factors such as stand density, canopy area and roughness, and leaf morphology.

In each forest type, VWM throughfall Ca, K, Mg and Cl concentrations were significantly increased in relation to rainfall. The increases in K and Mg concentrations relative to those of rainfall were highest under the natural forest compared to the two plantations. Throughfall NH_4–N concentration was lower in each forest type and PO_4–P was lower in the two plantations in relation to rainfall. The concentration of NO_3–N in rainfall was significantly lowered after passing through the canopy of *Cupressus* plantation, while under *Eucalyptus* plantation and the natural forest the reverse holds true. Although statistically not significant in *Eucalyptus*, the concentration of SO_4–S in all forest types increased after the passage through the canopy. With few exceptions, nutrient concentrations in rainfall and throughfall of our study site were higher than those summarized for other montane tropical forest sites (Table

D2). The seasonal patterns in nutrient concentrations in throughfall of each forest type (Table D4) were similar, being highest, with few exceptions, during the dry season (October–January) presumably due to wash-off of dry deposition accumulated on the canopy during dry periods by intermittent low-volume rain events. There was no discernible trend with time in rainfall nutrient concentrations except for Na which showed a slight increasing tendency from the dry season to the wet season (Table D4).

The annual total amounts of nutrients (Ca, K, Mg, Na, Cl, NH_4–N, NO_3–N, SO_4–S, PO_4–P) reaching the soil (Table D5) in throughfall were 14 kg $ha^{-1}yr^{-1}$ under *Cupressus*, 24 kg $ha^{-1}yr^{-1}$ under *Eucalyptus* and 21 kg $ha^{-1}yr^{-1}$ under the natural forest. Of these, 12 kg $ha^{-1}yr^{-1}$can be explained by the rainfall while 2, 9, and 12 kg $ha^{-1}yr^{-1}$ under *Cupressus*, natural forest and *Eucalyptus*, respectively, derived from dry deposition and leaching of intracellular solutes from the canopy. In spite of the same amount of throughfall water with that of *Cupressus* and about 30% less than *Eucalyptus*, the observed annual total weight of nutrients in the natural forest suggests that the much rougher surface of the natural forest canopy increased the deposition area and allowed interception of dust carrying winds. Except NH_4–N and PO_4–P in all forest types, Ca and NO_3–N in *Cupressus* and the natural forest, and SO_4–S in *Eucalyptus* the fluxes of all other nutrients in throughfall of each forest type were significantly different from that of rainfall (Table D5). Annual fluxes of nutrients in rainfall and throughfall in the Munesa forest (Table D5) were lower than the values summarised for other montane tropical forests (Table D5). The greatest variability in rainfall and throughfall inputs between our study and others could be due to variability in rainfall amount, species composition and canopy structure, and the availability of nutrients from atmospheric and rock weathering processes and exposure to acid precipitation. Throughfall inputs of Ca, Mg, Na and Cl were significantly different among forest types. Although statistically not significant for some of the nutrients, throughfall in *Cupressus* had the lowest fluxes of each nutrient compared to the natural forest and *Eucalyptus*, NH_4–N was an exception. *Eucalyptus* was found to have relatively the highest throughfall input of Ca, Mg, Na, NO_3–N and SO_4–S compared to the natural forest mainly due to high volume of water reaching at the soil surface under the *Eucalyptus* plantation. The inputs of Cl and PO_4–P were slightly highest under the natural forest compared to *Eucalyptus* mainly resulting from high concentration. Nutrient fluxes varied considerably from season to season and were higher during the wet season (Table D7) because of higher rainfall volume or rainfall intensity, although concentrations of most nutrients tended to be higher in the dry season. This seasonal pattern of variation in fluxes indicated that except for the few relatively high-volume dry season rain events, throughfall in dry season is not likely to provide a major nutrient source via root uptake for overstorey tree species.

The data in net throughfall Na, NH_4–N, NO_3–N, SO_4–S and PO_4–P fluxes (Table D5) indicate absorption by the canopies of all forest types, whereas net throughfall Ca, K, Mg and Cl fluxes indicate canopy leaching. The magnitude of absorption and canopy leaching were both nutrient and tree species specific. Comparison of net throughfall fluxes among seasons indicated clear temporal patterns of canopy leaching and very different chemical speciation associated with biological uptake (Fig. D1). Ammonium–N and PO_4–P were taken up in larger quantity during the small rainy season in all forest types while Na, NO_3–N and SO_4–S were mainly taken up during the main rainy season. Calcium in *Cupressus* and natural forest indicated intermediate behaviour: a tendency towards absorption during the main rainy season and canopy leaching during the dry and small rainy seasons. A similar behaviour was also observed for NO_3–N in *Eucalyptus* and the natural forest.

4.3. Nutrient dynamics in soil solution

Water passing through the forest floor under the different tree species did not differ in mean nutrient concentrations except for K, Mg and NO_3–N (Table E2). Magnesium and NO_3–N concentrations were significantly higher under the natural forest and *Eucalyptus* plantation than under *Cupressus*. Potassium concentration was higher under *Eucalyptus* than under the natural forest and *Cupressus*. The low C/N ratio in the forest floors of *Eucalyptus* and natural forest might have triggered nitrification in comparison to *Cupressus* which had high C/N ratio in the organic layer. Soils with C/N ratios >25-30 and low nutrient concentrations are reported to be poor-nitrifying (Gundersen and Rasmussen, 1990). In each forest type, after K, Ca and Cl were the most abundant nutrients leached from the litter layer (Table E2). PO_4–P was the least of all the nutrients followed by NH_4–N. Except for NH_4–N in the natural forest, forest floor leachate nutrient concentrations were enriched in all forest types in relation to rainfall and throughfall, being more pronounced in NO_3–N, Ca, Mg and PO_4–P concentrations. Leaching of nutrients from decaying vegetation and microbial mineralization of elements within organic matter contribute to the observed enrichment of forest floor leachates.

Nutrient fluxes from the forest floor to the mineral soil were not significantly different among forest types, but were slightly highest under *Eucalyptus* (Table E4). In general, large fluxes were observed for Ca and Cl. Measured nutrient exports from the forest floor to the mineral soil decreased in relation to throughfall fluxes for most of the nutrients indicating that nutrients that are deposited from throughfall as well as those released from decomposition are effectively taken up by plant roots or immobilised. The nutrients that decreased most were NH_4–N≈K>Cl>SO_4–S under the natural forest, SO_4–S >Na> Ca≈ Mg under *Eucalyptus*, and NH_4–N >SO_4–S>Na>Ca under *Cupressus*. Nitrate–N exports from the forest floor exceeded the inputs via throughfall by about 161% for *Cupressus* and 70% for the natural forest and *Eucalyptus*. Calcium and PO_4–P exports by leaching out of the *Cupressus* forest floor were 34% and 33% higher than the corresponding throughfall inputs; below *Eucalyptus* as high as 50% more PO_4–P was exported in comparison to the input by throughfall.

The median mean nutrient concentrations in the soil solutions of the mineral soil were in the order: Na>Cl>Ca>SO_4–S>Mg>NO_3–N>K>NH_4–N below the natural forest, Na>Ca> SO_4–S>Cl>Mg> NO_3–N>K> NH_4–N below *Eucalyptus* and NO_3–N>Ca>Na>Cl> SO_4–S>K> NH_4–N below *Cupressus* (Table F4). The concentration of PO_4–P in the mineral soil solution was generally below the detection limit in the three forest types. Phosphorus is relatively insoluble and readily fixed by soil minerals (Brady and Weil, 1999). Potassium was also often below the detection limit in the mineral soil solution under the natural forest and under *Cupressus* plantation probably due to the high biological demand for this element. The lower NH_4–N concentration relative to NO_3–N in both the forest floor leachate and mineral soil solution was probably a result of nitrification, vegetation uptake, adsorption or assimilation by microbes. With the exceptions of Mg, Na and NO_3–N concentrations at all depths below *Cupressus* plantation and SO_4–S and Na at some soil depths below the natural forest and *Eucalyptus* plantation, all other nutrients decreased relative to the concentrations in the forest floor leachate. Potassium, Mg, NH_4–N and PO_4–P decreased to a great degree compared to the other nutrients. An increase in NO_3–N concentration in the mineral soil solution relative to forest floor leachate below *Pinus* was also reported by Lilienfein et al. (2001) in Brazil. Schrumpf (2004) observed a decrease in Ca, K, Mg, Na and NH_4–N concentrations and an increase in NO_3–N concentration in the mineral soil solution of Andisols in Kilimanjaro in relation to the forest floor leachate. In Congo, Laclau et al. (2003) reported an increase in NH_4–N, NO_3–N and SO_4–S and a decrease in Ca, K, Mg and Na in the mineral soil solution relative to the forest floor leachates below *Eucalyptus* plantation.

The concentrations of K, NH_4–N and SO_4–S in all forest types and Ca, Mg and NO_3–N concentrations below *Eucalyptus* and the natural forest decreased steadily with increasing soil depth, presumably due to adsorption by the soil colloid or to plant and microbial uptake (Table F4). In contrast, below *Cupressus*, the concentrations of Ca, Mg and NO_3–N decreased from 0.2 m depth to 0.5 m depth and then increased at the depth of 1 m. This pattern appears to follow the root distribution and concurrent nutrient uptake as the roots of *Cupressus* are confined to the surface 0.5 m (Ashagrie, pers. observation). Median Ca, Mg and NO_3–N concentrations below *Cupressus*, respectively, were 4, 3.37 and 17 times higher than below the natural forest and 2, 2.41 and 7 times higher than below *Eucalyptus* (Table F4). The higher Ca, Mg and NO_3–N concentrations in the soil solution under *Cupressus* relative to the other two forest types were probably due to the fact that these nutrients were in excess of tree and microbial requirements. Much of the observed differences in median mean nutrient concentrations were attributed to the large differences at the depth of 1 m, being 3 and 8 times, 4 and 7 times and 81 and 23 times more for Ca, Mg and NO_3–N under *Cupressus* than under *Eucalyptus* and the natural forest, respectively (Table F4). In a ^{15}N tracer study made by Fischer (2004) at the same experimental plots, large proportion of the ^{15}N applied at the surface (0 m soil depth) under *Cupressus* was found in the deeper soil layer (0.3–0.6 m) confirming that leaching had occurred below *Cupressus*. Lilienfein et al. (2000, 2001) reported two times higher Ca, K, Mg, Na and NO_3–N concentrations in soil solution under *Pinus* than under *Cerrado* in sub humid Central Brazil.

5. GENERAL CONCLUSIONS

This study showed that conversion of natural forest into crop land had more deleterious effects on soil aggregation, SOC and nutrient contents than conversion into *Eucalyptus* plantation. Direct measurement of short-term SOM losses or gains resulting from variations in land-use may not clearly reflect the effects of land use and soil management because of the generally high background soil C pool (Haynes, 1999). Physical fractionation of soils into aggregate and particle-size fractions enabled separation of SOM into pools of differing composition and biological function and turnover, thus allowing sensitive detection of changes in SOM dynamics and soil fertility resulting from changes in land-use. In general, losses of SOC and nutrients associated with the different size/density fractions resulting from the conversion of the natural forest into *Eucalyptus* plantation and crop cultivation were more pronounced than losses observed in the bulk soil and total water stable aggregates. Plantations of several tree species growing under similar site conditions offer an opportunity to evaluate species´ effects without confounding problems of prior soil differences. The results of the present study on SOC and nutrient stocks as affected by conversion of the natural forest into different exotic tree species plantations emphasise the importance of forest type, stand age and management in affecting the size of C and nutrient stocks. In general, the effect of tree species appear limited largely to the forest floor, with little change in mineral soils. The accumulation of organic detritus and the relative losses or accumulation rates of C and nutrients in the soil depends on the rate of decomposition of the plant material which is influenced by litter quality, acidity, soil moisture and temperature, and the kinds of micro flora and fauna present. Hence, many of the above mentioned factors that affect ecosystem processes and C and nutrient storage, and the relationships between substrate quality and decomposition rates need to be further investigated.

The results of the present study also showed that water and nutrient inputs into and nutrient outputs from the studied ecosystems were affected after conversion of the natural forest into managed forest plantations. Because *Eucalyptus* leaves often are held vertically on the twigs, the leathery nature of the leaves, and the overall low stand density, the amount of rain water intercepted and lost by evaporation was lower in comparison to the natural forest and *Cupressus* plantation. The higher rainfall interception under the natural forest and *Cupressus* means reduced rainfall infiltration and insufficient rainy season replenishment of ground water reserves that may in the longer-term affect plant productivity and dry season stream flows. Canopy characteristics such as leaf area and canopy density, as well as canopy roughness relative to wind and the amount of water reaching the soil affected the input of nutrients by rainfall and throughfall. Rainfall chemistry at Munesa showed no evidence of acid or polluted deposition of anthropogenic origin. However, except for K and Mg, the annual levels of mineral–element accession in rainfall can augment the nutrient stocks in the soil. The input of most nutrients by throughfall under *Cupressus* was lower than under the natural forest and *Eucalyptus*. Ecosystem-specific patterns of vegetation composition and associated demand for nutrients appear to control the dynamics of nutrients in soil solution. In general, forest ecosystems retain nutrients very efficiently. Exceptions to this general pattern are ecosystems with low nutrient uptake by plants and immobilization cannot compensate for reduced plant uptake. Plant uptake is a major sink of available nutrients. Apparently, the movement of Ca, Mg, and NO3–N out of the rooting zone were higher under *Cupressus* than under the other forest types. The fact that NO_3–N and basic cations leached from relatively low or non nitrifying soils such as in *Cupressus* with high C/N ratio in the organic layer may, indicate that plant uptake for Ca, Mg and NO3–N periodically did not match mineralization in the soil or mineralization has exceeded the retention capacity of the system. From the ecological point of view, the presence of basic cations and mineralised nitrogen in subsoil solution under *Cupressus* indicates leaching, and that the ecosystem is not characterised by tight nutrient-cycling.

Generally, considering the high C/N ratio in the organic layer of the studied *Cupressus* stand that will negatively influence the rates of litter and nutrient turnover, loss of basic cations from the rooting zone may, in the short-term, reduce site fertility and contribute to the onset of nutrient deficiencies. However, in the long-term, the positive impacts of annual cycling of nutrients through uptake by roots, fine root turnover, and above-ground litter deposition and atmospheric inputs act to maintain fertility of the soil. Weathering, one of the chief sources of nutrients, also acts to counteract loss of cations from the system. In *Eucalyptus* plantation, the presence of diverse shrub and herbaceous understorey vegetation might have contributed in nutrient retention. These aspects reveal some characteristics that could be important to *Cupressus* plantation management. In light of the poor nutrient retention capacity of *Cupressus*, future monoculture plantations with high tree density should be discouraged. Rather mixed stands formed by several tree species or monocultures with minimum tree density that allow the growth of understorey shrub and herbaceous vegetation should be encouraged so as to maintain the fertility status of the soil for future rotations. Furthermore, such practices may also ensure input of sufficient rain water to the soil and enhance the regeneration of native plant species which is now lacking under *Cupressus*. Unfortunately, it is impossible to estimate the total loss rate due to the inherent methodological problems in quantifying the total water flow rates through the soil, but enlighting the general trends and patterns found in comparison of the different forest ecosystems may be more important than the precise budgetary calculations. Generally, drawing conclusions or making inferences solely based on a one year ecosystem analysis studies that focus on elucidating processes would be difficult, therefore, continuous monitoring of water and nutrient input and output patterns in the studied ecosystems is needed to reach a valid conclusion.

6. REFERENCES

Abate, A., 2004. Biomass and nutrient studies of selected tree species of natural and plantation forests: Implications for a sustainable management of the Munesa-Shashemene forest, Ethiopia. Ph.D. Thesis. University of Bayreuth. pp150.

Anonymous, 2004. Ethiopian montane moorlands. www:/.worldwildlife.org/wilds/terrestrial/at/1008_full.html. downloaded on 26/06/04.

Asrat Abebe., 1992. Assessment of runoff and soil loss under different cover crops and slope lengths. Pp. 45–50. Natural resources management for concservation and development. Proceedings of the second natural resources concservation conference.10–13 may 1990. Addis Ababa, Ethiopia. IAR, Addis Ababa.

Beare, M.H., Cabera, M.L., Hendrix, P.F., Coleman, D.C., 1994. Aggregate protected and unprotected organic matter pools in conventional and no-tillage soils. Soil Science Society of America Journal 58, 787−795.

Besnard, E., Chenu, C., Balesdent, J., Puget, P., Arrouays, D., 1996. Fate of particulate organic matter in soil aggregates during cultivation. European Journal of Soil Science 47, 495−503.

Betre, A., Tekalign, M., Larsoon, R., 2000. Effects of natural forest conversion to plantation and secondary forests on soil chemical properties in Munesa Montane forests, Ethiopia. Ethiopian Journal of Natural Resources 2, 165−187.

Binkley, D., 1986. Forest nutrition management. John Wiley & Sons. Pp. 289.

Bosch, J.M., Hewlett, J.D., 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evaporation. Journal of Hydrology 55, 3–23.

Brady , N.C., Weil, R.R., 1999. The nature and properties of soils, 12th Edition. Prentice Hall.

Brown, S., Lugo, A.E., 1990. Effets of forest clearing and succession on the carbon and nitrogen content of soils in Puerto Rico and US Virgin Islands. Plant and Soil 124, 53−64.

Brown, S., Lugo, A. E., 1982. The storage and production of organic matter in tropical forests and their role in the global carbon cycle. Biotropica 14, 161–176.

Buschiazzo, D.E., Hevia, G.G., Hepper, E.N., Urioste, A., Bono, A.A., Babinec, F., 2001. Organic C, N and P in size fractions of virgin and cultivated soils of the semi-arid pampa of Argentina. Journal of Arid Environments 48, 501−508.

Buyanovsky, G.A., Aslam, M., Wagner, G.H., 1994. Carbon turnover in soil physical fractions . Soil Science Society of America Journal 58, 1167−1173.

Cambardella, C.A., Elliott, E.T., 1993. Carbon and nitrogen distributions in aggregates from cultivated and grassland soils. Soil Science Society of America Journal 57, 1071−1076.

Cape, J.N., Brown, A.H.F., Robertson, S.M.C., Howson, G., Parerson, I.S., 1991. Interspecies comparisons of throughfall and stemflow at three sites in northern Britain. Forest Ecology and Management 46, 165–177.

Cavelier, J., Jaramillo, M., Solis, D., Leon, D., 1997. Water balance and nutrient inputs in bulk precipitation in tropical montane cloud forest in Panama. Journal of Hydrology 193, 83–96.

Christensen, B.T., 1996. Carbon in primary and secondary organo-mineral complexes. In: Carter, M.R., Stewart, B.A. (eds.), Structure and Organic Matter Storage in Agricultural Soils. Advances in Soil Science. CRC. Lewis Publisher, Boca Raton. pp. 58−97.

Christensen, B.T., 1992. Physical fractionation of soil organic matter in primary particles and density separates. Advances in Soil Science 20, 1−76.

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

De Vries, W., Reinds, G.J., Vel, E., 2003. Intensive monitoring of forest ecosystems in Europe 2: Atmospheric deposition and its impacts on soil solution chemistry. Forest Ecology and Management 174, 97−115.

Dormaar, J.F., 1983. Chemical properties of soil and water stable aggregates after sixty-seven years of cropping to spring wheat. Plant and Soil 75, 51−61.

EFAP. 1994. Ethiopian Forestry Action Program. VolumeI-Executive summary. Ministry of Natural Resources Development and Environmental Protection. Addis Ababa, Ethiopia.14p.

EFAP. 1993. Ethiopian Forestry Action Program. Volume II-The chanllenge for development. Ministry of Natural Resources Development and Environmental Protection. Addis Ababa, Ethiopia.

Elliott, E.T., 1986. Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. Soil Science Society of America Journal 50, 627−633.

Eswaran, H., van den Berg, E. and Reich, P. 1993. Organic carbon in soils of the world. Soil Science Society of America Journal 57, 192–194.

Evans, J., 1982. Plantation forestry in the tropics. Oxford university press, New York. 472pp.

FAO. 2003. State of the world´s forets. Rome.

FAO. 2001. Global forest resources assessment 2000: Main report. FAO forestry paper No. 140. Rome.

FAO. 1997. FAO/UNESCO Soil map of the world, revised legend with corrections and updates. World soil resources, Report No. 60, ISRIC, Wageningen, The Netherlands.

FAO. 1986. Ethiopian highlands reclamation study. Final report. Rome.

Fischer, H., 2004. Stickstoff in verschiedenen waldökosystemen des Munessa-waldes, Süd Äthiopien: Stckstoff-pools und ^{15}N-tracerdynamik. Diplomarbeit. Bayreuth University. Pp. 100.

Follet, R. F., 2001. Soil management concepts and carbon sequestration in cropland soils. Soil and Tillage Research 61, 77–92.

Gosz, J.R., 1981. Nitrogen cycling in coniferous ecosystems. Ecol. Bull. (Stockholm), 33, 405−426.

Gundersen, P., Emmet, B.A., Kjonaas, O.J., Koopmans, C.J., Tietema, A., 1998. Impact of nitrogen deposition on nitrogen cycling in forests: a synthesis of NITREX data. Forest Ecology and Management 101, 37−55.

Gundersen, P., Rasmussen, L., 1990. Nitrification in forest soils: effects of nitrogen deposition on soil acidification and aluminium release. Review of Environmental Contamination and Toxicology, 13, 1−45.

Guo, L. B., Gifford, R. M., 2002. Soil carbon stocks and land use change: a meta analysis. Global change biology 8, 345–360.

Gupta, V.V.S.R., Germida, J.J., 1988. Distribution of microbial biomass and its activity in different soil aggregate size classes as affected by cultivation. Soil Biology and Biochemistry 20, 777−786.

Jastrow, J.D., 1996. Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. Soil Science Society of America Journal 28, 665−676.

Haynes, R.J., 1999. Labile organic matter fractions and aggregate stability under short-term, grass-based leys. Soil Biology and Biochemistry 31, 1821−1830.

Hunri, H., 1988. Degradation and cincservation of the resources in the Ethiopian highlands. Mountain Resaerch Development 8(2/3), 123-130.

Krupa, S.V., 2002. Sampling and physico-chemical analysis of precipitation: a review. Environmental Pollution 120, 565–594.

Laclau, J.P., Ranger, J., Bouillet, J.P., Nzila, J.D., Deleporte, P., 2003. Nutrient cycling in a clonal stand of *Eucalyptus* and an adjacent savanna ecosystem in Congo. 1. Chemical composition of rainfall, throughfall and stemflow solutions. Forest Ecology and Management 176, 105–119.

Lemenih, M., Olsson, M., Karltun, E., 2004. Comparison of soil attributes under *Cupressus lusitanica* and *Eucalyptus saligna* established on abandoned farmlands with continuously cropped farmlands and natural forest in Ethiopia. Forest Ecology Management 195, 57−67.

Likens, G.E., Bormann, F.H., Pierce, R.S., Eaton, J.S., Johnson, N.M., 1977. Biogeochemistry of a forested ecosystem, Springer-Verlag, New York, 146 pp.

Lilienfein, J., Wilcke, W., 2004. Water and element input into native, agri- and silvicultural ecosystems of the Brazilian savanna. Biogeochemistry 67, 183–212.

Lilienfein, J., Wilcke, W., Thomas, R., Vilela, L., Lima, S.D.C., Zech, W., 2001. Effects Pinus Caaribaea forests on the C, N, P and S status of Brazilian savanna Oxisols. Forest Ecology and Management 147, 171−182.

Lilienfein, J., Wilcke, W., Ayarza, M.A., Vilela, L., Lima, S.D.C., Zech, W., 2000. Soil acidification in *Pinus caribaea* forests on Brazilian savanna Oxisols. Forest Ecology and Management 128, 145−157.

Lugo, A. E., 1992. Comparison of tropical tree plantations with secondary forests of similar age. Ecological Monograph 62, 1−41.

Lugo, A. E., Cuevas, E. and Sanchez, M. J., 1990. Nutrients and mass in litter and top soil of ten tropical tree plantations. Plant and Soil 125, 263−280.

Lugo, A. E., Sanchez, M. J. and Brown, S., 1986. Land use and organic carbon content of some subtropical soils. Plant and Soil 96, 185−196.

Mahieu, N., Powlson, D.S., Randall, E.W., 1999. Statistical analysis of published carbon-13 CPMAS NMR spectra of soil organic matter. Soil Science Society of America Journal 63, 307−319.

McDowell,W.H., Magill, A.H, Aitkenhead-Peterson, J.A., Aber, J.D., Merriam, J.L., Kaushal, S.S., 2004. Effects of chronic nitrogen amendment on dissolved organic matter and inorganic nitrogen in soil solution. Forest Ecology Management 196, 29–41.

MEDaC 2001. Ethiopia: Development framework and Plan of Action 2001-2020. Presentation of GFDRE on the third UN Conference on the Least of Developed Countries, Brussels, 2001. Ministry of Economic Development and Cooperation, Addis Ababa.

MoFED (Ministry of Finance and Economic Development) 2002. Ethiopia: Sustainable Development and Poverty Reduction Program (SDPRP). Addis Ababa, Ethiopia.

Mengistu, K., 2002. Tropical secondary forest management in Africa: Reality and perspectives. Ethiopia country paper. Presented at the workshop on tropical secondary forest management in Africa: Realities and perspectives. 9−13 December 2002, Nairobi, Kenya.

Michelsen, A., Lisanework, N., Friis, I., 1993. Impacts of tree plantations in the Ethiopian highland on soil fertility, shoot and root growth, nutrient utilization and mycorrhizal colonization. Forest Ecology and Management 61, 299–324.

Müller-Hohenstein, K., Abate, A., 2004. Rain forest margins and their dynamics in south-east Ethiopia. In Gerold, G., Fremerey, M., Guhardja, E., (eds.), Land use, nature concservation and sustainability of rain forest margins in southeast Asia. Springer Verlag Berlin. Pp. 217-238.

NMSA, 1996. Climatic and agroclimatic resources of Ethiopia. Vol. 1, No. 1. National Meteorological Service Agency of Ethiopia, Addis Ababa. 137 pp.

Pohjonen, V., 1989. Establishment of fuelwood plantations in Ethiopia. Silva Carelica 14, 1−388.

Pritchett, W.L. 1979. Properties and management offorest soils. John willey&Sons. New York. 500p.

Puget, P., Chenu, C., Balesdent, J., 1995. Total and young organic matter distributions in aggregates of silty cultivated soils. European Journal of Soil Science 46, 449–459.

Ranger, J., Marques, R., Jussy, J-H., 2001. Forest soil dynamics during stand development assessed by lysimeter and centrifuge solutions. Forest Ecology Management 144, 129−145.

Redhead, J.F., Hall, J.B., 1992. Tropical forestry. Longman Scientific and Technical, UK. 173pp.

Schrumpf, M., 2004. Biogeochemical investigations in old growth and disturbed forest sites at mount Kilimanjaro. Dissertation, PhD., Bayreuth University.

Six, J., Elliott, E.T., Paustian, K., Doran, J.W., 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. Soil Science Society of America Journal 62, 1367−1377.

Spaccini, R., Zena, A., Igwe, C.A., Mbagwu, J.S.C., Piccolo, A., 2001. Carbohydrates in water-stable aggregates and particle size fractions of forested and cultivated soils in two contrasting tropical ecosystems. Biogeochemistry 53, 1−22.

Solomon, D., Fritzsche, F., Tekalign, M., Lehmann, J., Zech, W., 2002. Soil organic matter composition in the sub-humid Ethiopian highlands as influenced by deforestation and agricultural management. Soil Science Society of America Journal 66, 68−82.

Swank, W.T., Douglass, J.E., 1974. Streamflow greatly reduced by converting deciduous hardwood stands to pine. Science 185, 857–859.

Tisdall, J.M., Oades, J.M., 1982. Organic matter and water stable aggregates in soils. Journal of Soil Science 33, 141−163.

Tobon, G., Sevink, J., Verstraten, J.M., 2004. Litterflow chemistry and nutrient uptake from the forest floor in northwest Amazonian forest ecosystems. Biogeochemistry 69, 315–339.

Ulrich, B., 1983. Interactions of forest canopies with atmospheric constituents:SO_2, alkali and earth alkali cations, and chloride. In Ulrich, B., Pankrath, J., (eds.), Effects of Accumulation of Air Pollutants in Forest Ecosystems. D. Reidel Publishing, Dordrecht, The Netherlands. Pp.33–45.

Veneklaas, E.J., 1991. Litterfall, nutrient fluxes in two montane tropical rain forests, Colombia. Journal of Tropical Ecology 7, 319–336.

Veneklaas, E.J., 1990. Nutrient fluxes in bulk precipitation and throughfall in two montane tropical rain forests, Colombia. Journal of Ecology 78, 974–992.

von Breitenbach, F., 1962. National forestry development planning: A feasibility and priority study on the example of Ethiopia. Forestry Review No. 3/4. MoA, Addis Ababa.

Wilcke W, Yasin S, Abramowski U, Valarezo C and Zech W 2002 Nutrient storage and turnover in organic layers under tropical montane rain forest in Ecuador. European Journal of Soil Science 53, 15−27.

Zech, W., Drechsel, P., 1998. Nutrient disorders and nutrient management in fast growing plantations. In Schulte, A., Ruhiyat, D. (eds.), Soils of tropical forest ecosystems. Charactreristics, ecology and management. Springer Verlag, Berlin. Pp. 90-106.

Zinn, Y.L., Resck, D.V.S., Silva, J.E., 2002.Soil organic carbon as affected by afforestation with *Eucalyptus* and *Pinus* in the Cerado region of Brazil, Forest Ecology and Management 166, 285−294 2002.

7. PAPERS (A−F)

A Transformation of a *Podocarpus falcatus* dominated natural forest into a monoculture *Eucalyptus globulus* plantation at Munesa, Ethiopia: Soil organic C, N and S dynamics in primary particle and aggregate-size fractions

Y. Ashagrie[1]*, W. Zech[1] and G. Guggenberger[1,2]

[1]*Institute of Soil Science and Soil Geography, University of Bayreuth, D-95440 Bayreuth, Germany*
[2]*Present address: Institute of Soil Science and Plant Nutrition, Martin Luther University, D-06099 Halle, Germany*
* Correspondence: yeshanew@hotmail.com, Tel: ++49-921-55-21-46 Fax: ++49-921-55-22-46

Abstract

Changes in land-use and management can affect soil structure, soil organic carbon (SOC) and other nutrients reserve (such as N, P, S). We analysed organic carbon (OC), total nitrogen (N), and total sulfur (S) in particle-size, aggregate-size and size/density fractions of soil organic matter (SOM) in order to identify the SOM pools most affected by the conversion of a *Podocurpus falcatus* dominated mixed natural forest into a monoculture *Eucalyptus globulus* plantation 21 years ago on a reddish brown Nitisol at Munesa, Ethiopia. Bulk soil OC, N, and S concentrations and stocks in soil to 20 cm depth were not significantly changed after the conversion of the natural forest into Eucalyptus plantation, but C/N ratio narrowed significantly. Soil organic C, N and S concentrations, and C/N and C/S ratios in sand and silt separates from the plantation samples were significantly reduced, while clay N and S concentrations had slightly increased. The losses of SOC, N and S in the sand fraction were more pronounced than that in the silt. Aggregate stability and total SOC, N and S concentrations of the aggregates were not significantly different in samples from the *Eucalyptus* plantation and the natural forest. In the plantation samples, both the free light fraction (LF) and the intra-particulate organic matter (iPOM) C, N and S concentrations associated with the macroaggregates were significantly reduced. Differences in the total amount of the free LF (on the basis of water-stable aggregates proportion) between the two forest types were not apparent, suggesting that SOM quality is more prone to changes in land-use and soil management strategies than the total amount of SOM. The loss of iPOM was

higher than that of free LF probably due to gaseous losses of organic matter (OM) inside the aggregates caused by high fire temperatures during clearing and site preparation. In both forest types, the LF OM comprised the highest percentage of whole soil OM and the loss of particulate organic matter (POM) accounted for much of the losses of OM. Overall, the results showed that analysis of OC, N and S concentrations in soil particle and aggregate-sizes, and size/density fraction of SOM allowed sensitive detection of changes in SOM dynamics and soil fertility resulting from changes in land-use.

Key words: Land-use change; Soil organic carbon; Nitrogen, Sulfur, Eucalyptus plantation; Particulate organic matter; Nitisol; Ethiopia.

1. Introduction

In Ethiopia, massive deforestation of natural forests and extensive use of agricultural lands have resulted in soil degradation and loss of environmental quality (EFAP, 1994). To reduce land degradation, and to satisfy the demand for timber and timber products of the local population, extensive afforestation with fast-growing exotic tree species has been carried out on degraded agricultural lands (Pohjonen and Pukkala, 1990). Sometimes, degraded secondary forests containing low quality and non-uniform stands of several species were also transformed into forest plantations. Of the total area of 200, 000 ha covered by plantations in 1992 more than 60% is under *Eucalyptus* species (EFAP, 1994). Although intensive management of exotic tree species may provide rapid growth and a higher economic return than would native tree species, little is known about the environmental impacts of this practice, such as on soil quality and productivity. Following refforestation, changes inevitably occur in the quantity, quality, temporal and spatial distribution of soil organic carbon (SOC) inputs, depending on type of forest established (Brown and Lugo, 1990). For example, in Ethiopia, Solomon et al. (2002) reported losses of about 27% of SOC and 13% of N and S after 25 years of conversion of the natural forest into *Cupressus* plantations. Zinn et al. (2002) found about 23 to 48% SOC loss after conversion of Brazilian native wooded savanna to *Eucalyptus* plantations.

Maintenance and improvement of soil organic matter (SOM) content is generally accepted as being an important aim for any sustainable soil fertility management because it is a major reservoir of nutrients such as N, S and P, and influences soil structure, water availability and other important chemical, physical and biological properties of soil (Haynes and Beare, 1996). Carbon is stored in terrestrial ecosystems in diverse organic forms with a wide range of mean residence times (Balesdent and Mariotti, 1996). The organic matter associated with different size fractions of soil, and that of the organo-mineral fractions of specific particle and aggregate sizes, exhibit distinct properties with respect to their composition and turnover (Christensen, 2001). The initial impact of land-use or management change occurs disproportionately in pools with short residence times (Cambardella and Elliott, 1992), whereas the effect on stable SOC pools occurs slowly over a much longer time period.

Direct measurement of short-term SOM losses or gains resulting from variations in land-use may not clearly show the effect of land use and soil management because of the generally high background soil C pool (Haynes, 1999). Therefore, approaches based on characterization of active SOM with comparatively rapid turnover rates have been suggested as a more sensitive indicators of soil fertility that allow early detection of changes in soil fertility before soil degradation becomes apparent (Cadisch et al., 1996; Haynes and Beare, 1996). Physical fractionation of soil into aggregate and particle-size fractions in studies of SOM has received increased attention because it enables separation into pools of differing composition and biological function (Christensen, 1992, 2001).

Among the different labile SOM pools, those associated with the sand fraction (Christensen,

2001) and particulate organic matter (POM), a pool that is functionally similar to light fraction (free LF) organic matter (Cambardella and Elliott, 1994), closely reflect early changes in SOM resulting from changes in land use and soil management. Similarly, the OM that binds microaggregates to macroaggregates is labile and responds more sensitively to changes in land use than the organic matter that binds microaggregates (Elliott, 1986; Gupta and Germida 1988; Cadisch et al. 1996; Christensen, 1996). Most of the labile organic matter within macroaggregates could be free light-fraction POM of relatively low-density, mineral-associated OM (Cambardella and Elliott, 1993). Several authors (Guggenberger et al., 1994; Solomon et al., 2002; Zinn et al., 2002) found differences in the quality and amount of SOM associated with mineral particles of different sizes. They also reported relatively greater losses of OC in the coarser particle-size separates than in the finer particle-size separates as a result of changes in land use from native vegetation to plantation.

The presence of a monoculture *Eucalyptus* plantation side by side with the natural forest from which it was established 21 years ago provided the opportunity to determine if soil structure and the quantity, as well as the quality, of organic matter in the mineral soil had changed as result of land-use change in the highlands of Ethiopia.

2. Materials and methods

2.1. Site description

The study was conducted at the Munesa/Shashemenie forest enterprise site (7°34´N and 38°53´E) located about 240 km south east of Addis Ababa at an altitude of 2400 m.a.s.l. Rainfall is bimodal with mean annual precipitation of 1250 mm most of it falling in July and August, and mean annual temperature is 19 °C with little seasonal variation. The soils are clayey and very deep with reddish brown colour, and are moderately acidic at or near the surface and slightly acidic at depth. The principal parent materials are of volcanic origin from which Rhodic Nitisols were derived (FAO, 1997). A *Podocarpus falcatus* dominated mixed natural forest (ca. 3 to 4 thousand years) and an adjacent 21 years *Eucalyptus* plantation were selected for this study. The natural forest is one of the few remaining natural forest reserves in the country. Eucalyptus plantation in the study area covers about ca. 1, 620 ha comprising different species, and was established after clearing and burning of part of the natural forest. Clearing was done manually and the surface biomass was burned on site. Tree density in the studied plantation compartment was about 595 tree ha^{-1} and tree diameter at breast height (dbh) ranged from 19 to 39 cm with a height of 30 to 40 m. The studied plantation is open to light penetration with dense understorey grass and broad-leaved herbaceous, and different species of shrub vegetation, and is occasionally grazed by free grazing cattle.

2.2. Sampling

In each forest type, three 0.06 ha plots ca. 100 m apart from each other were located randomly and a pit was excavated to the depth of 1.2 m at the centre of each plot. In addition, four 1 m^2 sub plots were marked randomly at 10 m radius from the centre of each plot. Soil samples ca. 500 g were taken from the three sides of the pit by a shovel, and at three points within each of the 1 m^2 sub plots by an auger to the depth of 0–20 cm. All the auger and pit samples in the 0.06 ha plot were mixed and the final number of samples were reduced to three per land use. After air drying, a sub sample was sieved through 5 mm sieve size for aggregate fractionation, and the remaining was sieved through 2 mm sieve size for bulk soil C, N and S analysis, and particle size fractionation. Soil samples for bulk density determination were taken from the wall of the three pits by a 100 cm^3 metal cylinder; totally seven per land use.

2.3. Soil particle size fractionation

Air-dried and sieved (2 mm mesh) 30 g samples were put in a centrifuge tube and dispersed ultrasonically at a soil: water ratio of 1:5 (w/v), with an energy input of 60 J ml^{-1}using a probe type sonicator (Branson Sonifier W–450). Coarse sand fraction (250–2000 µm) was separated by wet sieving, and the remaining material in the <250 µm fraction was further sonicated at a soil: water ratio of 1:10 (w/v), with an energy input of 440 J ml^{-1}. The clay-size separates (< 2 µm) were isolated from the silt (2–20 µm) and fine sand (20–250 µm) by repeated centrifugation, while the silt-size separates were isolated from the fine sand by wet sieving. After fractionation, the different particle-size fractions were dried at 50 ^{o}C.

2.4. Soil aggregate size fractionation and separation of particulate organic matter

The size distribution of aggregates was measured by a wet sieving through a series of sieves (2, 1, 0.5, 0.25 and 0.053 mm) following the procedures of Cambardella and Elliott (1993). A 70–80 g sample of air-dried soil passed through a 5 mm sieve size was spread on the top of a 2 mm sieve submerged in a bucket of deionized water. The water level was adjusted so that the aggregates on the sieve were just submerged. Soils were left immersed in the water for 10 min and then sieved by moving the sieves 3 cm vertically 50 times during a period of 2 min. During the sieving process, floatable materials >2 mm were removed and discarded. According to Six et al. (1998) materials > 2mm are not considered an integral part of SOM. The material remaining on the 2 mm sieve was transferred to a glass pan. Soil plus water that passed through the sieve were poured onto the next finer sieve and the processes repeated, but floatable materials were not removed and discarded. The different aggregate sizes were dried in the oven at 50 °C overnight for chemical analysis.

The separation of POM followed the procedure of Six et al. (1998). Prior to POM separation, the fractions in the >0.25 mm size aggregates were bulked as macroaggregates and the 0.053–0.25 mm size as microaggregates. After the aggregates were dried (105 °C) in the oven overnight and cooled in a desiccator to room temperature, about 10 g of each aggregate fraction was taken in a conical centrifuge tube and suspended in 35 ml sodium polytungstate (adjusted to a density of 1.8 g cm^{-3}) by hand shaking. The suspension was allowed to stand for 20 min before centrifugation at 1250 rpm for 60 min. After centrifugation, the floating material was collected on filters and rinsed thoroughly with deionized water to remove sodium polytungstate, this material is referred to as free LF. The heavy fraction remaining in the tube was washed twice with 50 ml deionized water and dispersed in 50 ml of 5% sodium hexametaphosphate by shaking in a reciprocal shaker for 18 hours. The dispersed heavy fraction was rinsed through a 0.053 mm sieve with deionized water. The material remaining on the sieve is intra-particulate organic matter (iPOM) + sand. Both the free LF and iPOM were dried in the oven at 50 °C overnight. The dried subsamples from each aggregate size class, particle size fraction, and the free light fraction and iPOM were finely ground in a rotary ball mill for chemical analysis.

2.5. Soil analysis

Organic C, N and S concentrations in bulk soil, size fractions and POM were determined using a CHNS–analyzer (Vario EL, Elementar Analysensysteme, GmbH, Hanau, Germany). Element stocks (kg m^{-2}) were calculated as a product of bulk density, depth of sampling and element´s concentration per unit of soil samples. The pH_{KCl} (soil:solution ratio 1:2.5) of the soil was determined with a standard pH electrode (Orion U402–S7). Bulk density was determined after drying the soil in an oven at 105°C.

2.6. Statistical analysis

One way analysis of variance (ANOVA–1) was performed to asses the effect of change in land-use on soil aggregate stability, and soil organic C and nutrients associated with the different particle size/density fractions using the MSTATC statistical package. Separation of means were performed using Tukey´s honestly significance difference test with a significance level of P < 0.05.

3. Results and discussion

3.1. Organic C, N, and, S in bulk soil samples

Analysis of variance performed on the data showed that mean SOC, N and S concentrations and the C/S and N/S ratios in bulk soil samples did not differ significantly in the natural forest and *Eucalyptus* plantation (Table A1). The changes in bulk density after the establishment of *Eucalyptus* was also not significant, and varied from 0.86 g cm^{-3} under the natural forest to 0.91 g cm^{-3} under *Eucalyptus*. On an area basis, the two forest types had almost the same level of SOC and S stocks, but there appeared to be a slight and non significant gain of N in

Table A1. Soil organic C, N and S concentrations and stocks, and element ratios and bulk density (Bd) under the different land use types, results refer to the 0–20 cm soil depth.

	C	N	S	C/N	C/S	N/S	Bd	C	N	S
	------g kg^{-1}------						g cm^{-3}	-----kg m^{-2}------		
Natural forest	72	6	0.72	12**a**	100	8	0.86	12.4	1.0	0.12
	(7.4)	(0.6)	(0.07)	(0.6)	(3.5)	(0.3)	(0.02)	(0.9)	(0.1)	(0.01)
Eucalyptus plantation	61	7	0.75	9**b**	81	9	0.91	11.1	1.3	0.14
	(4.9)	(0.5)	(0.03)	(0.9)	(3.7)	(0.3)	(0.02)	(1.3)	(0.1)	(0.01)

Means followed by different lower case letters in a column are significantly different from each other at P<0.05. Numbers in parentheses are standard errors (n=7 for bulk density and n=3 for other parameters).

the *Eucalyptus* plantation (Table A1). This relative gain could perhaps be attributed to the recycling of N via excreta of free grazing cattle. In contrast to our results, Michelsen et al. (1993) reported significantly lower OC and nutrient concentrations under a 40 years *Eucalyptus* plantation than under an adjacent natural forest on a reddish brown soil in

Ethiopia. There was a significant decrease in the C/N ratio from 12 in the natural forest to 9 in the *Eucalyptus* plantation, indicating that changes in organic matter quality took place (Table A1).

3.2. Particle size distribution, and concentrations of OC, N, and S in particle size fractions

The proportional distribution of the different primary particles in the different size classes were similar in soils under the two land use types (Table A2) suggesting that the textural composition of the soils under the two land use types were comparable which further confirms similar origin of the two soils. Table A2 shows that, with the exception of S which was significantly higher in the clay fraction than the other fractions, OC and N did not differ significantly in the different particle-size fractions in the natural forest. In soil under *Eucalyptus* forest, however, OC, N and S concentrations were highest in the clay fraction; this indicates a preferential shift of the organic matter to the finer fractions during the decomposition process. The redistribution of sand-sized OM to clay-complexed OM during decomposition has already been shown by other authors (Anderson et al., 1981; Zinn et al., 2002).

Element ratios (C/N, C/S, and N/S) differed significantly among some of the size fractions at both sites, and tended to decrease in the order sand>silt>clay (Table A3). This could be attributed mainly to the accumulation of newly added and less decomposed organic matter in the coarser fractions (Guggenberger et al., 1994; Gerzabek et al., 2001). In both forest types, the C/N and C/S ratios of the coarse and fine sand, and silt fractions were higher than in the bulk soil, where as that of clay was lower. This might be due to the more aliphatic and humified nature of the clay-sized OM in comparison to the OM in the bulk soil and coarser fractions (Buyanovsky et al., 1994; Mahieu et al., 1999). The proportion of whole soil OC, N and S associated with the different particle size fractions calculated by multiplying the quantity of each particle size by the element concentrations showed that most of the whole soil OC, N and S in both land use types were associated with the finer particle sizes (<20 µm), being highest in the clay fraction (data not shown). This is in agreement with the observations of Desjardins et al. (1994) and Solomon et al. (2002) for tropical soils.

Table A2. Particle size distribution (%), and organic C, N and S concentrations (g kg^{-1} size fraction) in soil under natural forest (NF) and in soil 21 yr after conversion of natural forest into *Eucalyptus* plantation (EP), results refer to the 0–20 cm soil depth.

Particle Size	Size distribution		C		N		S	
	NF	EP	NF	EP	NF	EP	NF	EP
Cs	0.09	0.08	67**A** (9.1)	22**bB** (0.7)	4.0**A** (0.9)	2.0**bB** (0.3)	0.47**bA** (0.1)	0.23**bB** (0.03)
Fs	0.09	0.09	90**A** (1.6)	30**bB** (5.4)	4.5**A** (0.8)	2.3**bB** (0.2)	0.43**bA** (0.1)	0.23**bB** (0.03)
Si	0.28	0.30	62 **A** (0.3)	32**bB** (1.8)	4.1**A** (0.3)	3.3**bA** (0.1)	0.47**bA** (0.03)	0.37**bA** (0.03)
Cl	0.50	0.51	56 **A** (0.5)	53**a A** (2.4)	5.9**A** (0.5)	7.2**aA** (0.5)	0.77**aA** (0.1)	1.00**aA** (0.1)

Different lower case letters in a column indicate significant differences ($P<0.05$) between means under each land use according to Tukey´s HSD mean separation test. Different upper case letters in a row indicate significant differences between means at $P<0.05$.

Cs: Coarse sand; Fs: Fine sand; Si: Silt; Cl: Clay. Numbers in parentheses are standard errors (n=3).

Table A3. Element ratios of particle-size fractions as influenced by conversion of the natural forest (NF) into a *Eucalyptus* plantation (EP); results refer to 0–20 cm soil depth.

Particle size	C/N		C/S		N/S	
	NF	EP	NF	EP	NF	EP
CS	18**aA** (2.7)	11**abB** (0.9)	156**bA** (4)	96**abB** (10)	9**abA** (1.1)	9**abA** (0.2)
FS	20**aA** (0.5)	13**aB** (1.3)	245**aA** (3)	131**aB** (21)	13**aA** (0.6)	12**aA** (1)
Si	15**abA** (0.7)	10**abB** (0.5)	149**bA** (10)	90**abB** (10)	10**abA** (0.4)	9**abA** (0.6)
Cl	9**bA** (0.2)	7**bB** (0.2)	76**cA** (0.2)	54**bB** (3)	8**bA** (0.2)	7**bB** (0.2)

Different lower case letters in a column indicate significant differences between means at $P < 0.05$ according to Tukey´s HSD mean separation test. Different upper case letters in a row indicate significant differences between means at $P<0.05$. Cs: Coarse sand; Fs: Fine sand; Si: Silt; Cl: Clay. Numbers in parentheses are standard errors (n=3).

Mean OC of sand and silt fractions and, N and S of sand fraction concentrations, and C/N and C/S ratios of all the particle size fractions and N/S of clay fraction declined significantly after conversion of the natural forest to 21 years *Eucalyptus* plantation (Tables A2 & A3). The coarse sand fraction showed the highest losses of all three elements (Table A2), suggesting that organic matter associated with the coarser fractions is more labile and the first to be affected by changes in land use and soil management (Christensen, 1996; Solomon et al., 2002; Zinn et al., 2002). The degree of OC loss was larger than the losses of N and S. The changes in the clay-associated OC, N and S were not significantly affected by the change in land use, suggesting that the OM pool attached to clay is more stable. In tropical soils, clay associated SOM may contain the most stable OC, while in temperate soils OM in silt appears more stable than clay (Christensen, 1996). Results on the calculated enrichment factors (g kg $^{-1}$ separate)/ (g kg $^{-1}$ whole soil), which take account of the effects of different SOM levels in whole soils (Christensen, 1992) indicated that clearing of the natural forest and replacing it by the *Eucalyptus* plantation resulted in the depletion of OC, N and S from the sand-sized fractions and enrichment of OC, N and S in the clay-sized fraction (data not shown).

3.3. Aggregate distribution, and OC, N, and S concentrations

Clearing and reforestation of the natural forest with *Eucalyptus* did not significantly affect the distribution of WSA (Table A4). In both forest types, the distribution of WSA among the different size classes was significantly different, with > 85% of the total soil mass, remaining as water stable aggregates, >73% as macroaggregates (> 0.25 mm), and 14–17% as microaggregates (0.05–0.25 mm). Except N in microaggregates, the mean OC, N and S concentrations of the different aggregate sizes did not differ significantly between the natural and *Eucalyptus* forests (Fig A1a, b, c). This is not surprising since both land use types had almost the same level of soil aggregation (Table A4). The average C/N ratios of the larger aggregates (> 0.5 mm) were significantly wider in the natural forest than *Eucalyptus* plantation (Fig. A2a), where as C/S and N/S were not influenced by land use (fig. A2b, c). The mean C/N, C/S and N/S ratios of the aggregates in both soil types were nearly the same as those of the corresponding bulk soil.

3.4. Free light fraction OC, N and S

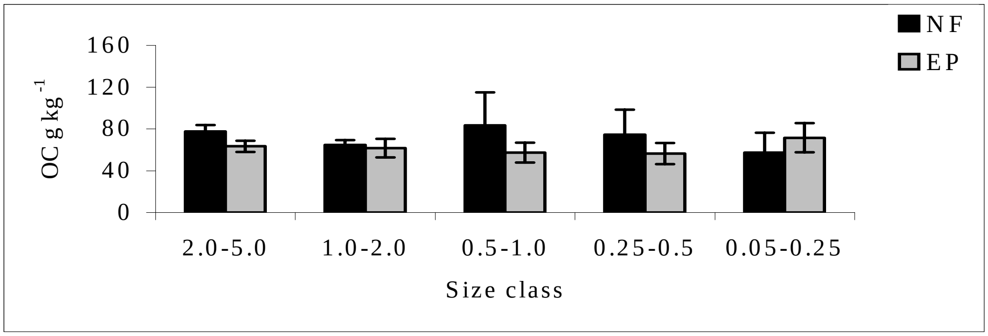
The data on the free LF mean OC and N concentrations (Table A5) demonstrate that there were significant reductions in these elements concentration after conversion of the natural forest into the *Eucalyptus* plantation. However, differences in the total amount of the above parameters (on the basis of water-stable aggregates proportion) between the two forest types were not apparent, suggesting that SOM quality is more prone to changes in land use and soil management strategies than the total amount of SOM. This situation is further illustrated by the significantly narrower mean C/N (11) and C/S (94) ratios in the *Eucalyptus* forest than those of 16 and 146, respectively, in the natural forest. The magnitude of reductions in both OC and N was much higher than the magnitude observed in the whole soil and total water stable aggregates. Cadisch et al. (1996) found 10 times more light fraction C (>100 μm) in the surface (0–2 cm) soil of a Brazilian rain forest than in a papaya plantation, and three-to-five times more than a pasture soil.

Table A4. Distribution of water-stable aggregates (WSA) (%) among different aggregate-size classes to 0–20 cm soil depth as influenced by replacement of natural forest with the *Eucalyptus* plantation 21 years ago.

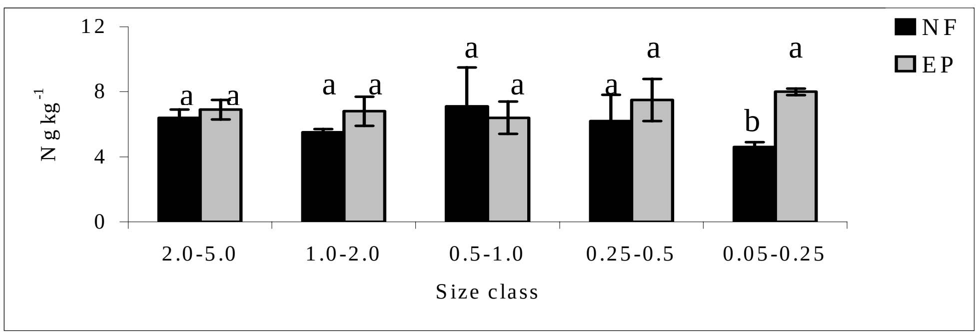
Size class (mm)	Natural forest	*Eucalyptus* plantation
2–5	10**b** (0.2)	8**c** (1.6)
1–2	21**a** (1.4)	18**ab** (0.3)
0.5–1	21**a** (1.3)	23**ab** (1.9)
0.25–0.5	22**a** (1.6)	25**a** (1.8)
0.053–0.25	14**b** (3.1)	17**b** (0.5)
Total	87	90

In a column, means followed by the same lower case letter are not significantly different from each other at P<0.05 according to Tukey´s HSD mean separation test. Numbers in parentheses are standard errors (n=3).

(a)



(b)



(c)

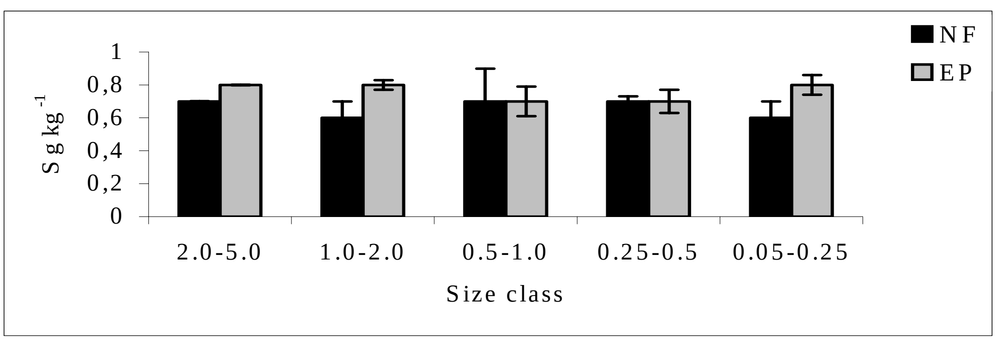
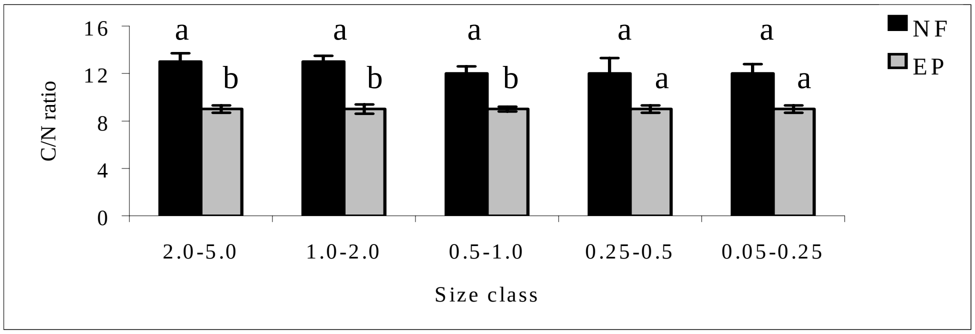
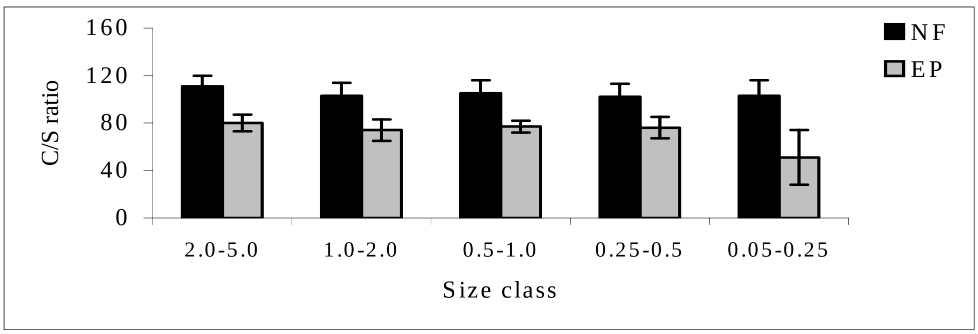


Fig. A1. Organic C (a), N (b) and S (c) concentrations (g kg^{-1} aggregate) of aggregate-size classes in soil at 0–20 cm depth under the natural forest (NF) and *Eucalyptus* plantation (EP). Vertical lines are standard errors (n=3).

(a)



(b)



(c)

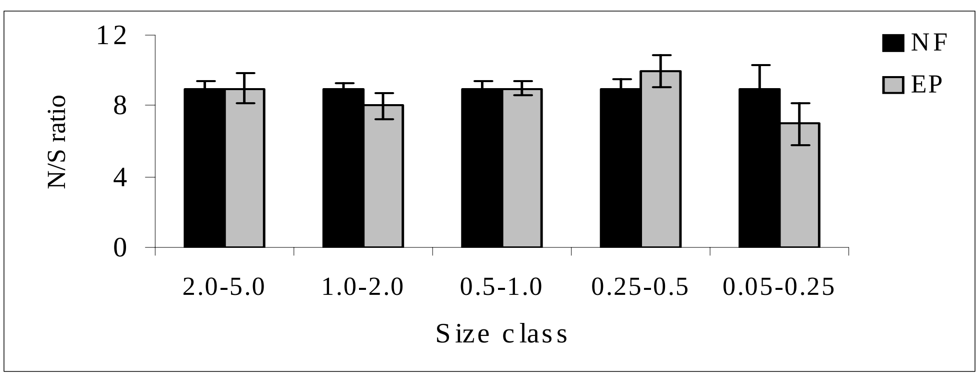


Fig. A2. C/N (a), C/S (b) and N/S (c) ratios of soil aggregate-size classes as affected by conversion of the natural forest (NF) into *Eucalyptus* plantation (EP), results refer to 0–20 cm depth. Vertical lines are standard errors (n=3).

The effect of changes in soil management on soil quality rather than on total SOM was also reported by Janzen et al. (1992) and Biederbeck et al. (1994). Mean S concentration and N/S ratio were not significantly different between the two land use types (Table A5). In the natural forest, the macroaggregate-associated LF had significantly larger OC, N and S concentration than the microaggregate-associated LF, while in the *Eucalyptus* plantation, the difference between the two size fractions was not significant. The mean OC and N concentrations, and C/N and C/S ratios of the macroaggregates in the natural forest were significantly higher than both the macro and micro aggregates in the *Eucalyptus* plantation (Table A5). Differences in microaggregate element concentrations and element ratios between the two land use types were not apparent with the exception of C/S ratio which was significantly higher in the natural forest than in the Eucalyptus plantation.

Table A5. Characteristics of the free light organic matter fractions to the soil depth of 0–20 cm.

	C	N	S			
	(mg g^{-1} fraction)			C/N	C/S	N/S
Natural forest						
Macroaggregates	138**a***	9.9**a***	1.0**a**	14**a***	144**a***	10**a**
	(6)	(0.4)	(0.07)	(0.07)	(3.4)	(0.3)
Microaggregates	84**b***	5.2**b**	0.6**b**	17**a**	148**a***	9**a**
	(12)	(1.4)	(0.07)	(2.2)	(4.4)	(1.3)
***Eucalyptus* plantation**						
Macroaggregates	56**a**	5.9**a**	0. 7**a**	9**a**	87**a**	9**a**
	(8)	(0.9)	(0.2)	(0.5)	(9)	(0.8)
Microaggregates	57**a**	4.4**a**	0.6**a**	13**a**	101**a**	8**a**
	(0.2)	(0.4)	(0.03)	(1.4)	(7)	(0.6)

Values followed by different lower case letters within a land use are significantly different (P<0.05). Values followed by * in the natural forest are significantly higher (P<0.05) than the corresponding values in the *Eucalyptus* plantation. Numbers in parentheses are standard errors (n=3).

The proportion of whole soil OC, N and S contained as free LF calculated by multiplying the quantity of the fraction recovered by concentration of each element were as much as 55, 47, and

39%, respectively, in the natural forest, and 57% of the whole soil OC, 34% of whole soil N and 36% of whole soil S in the *Eucalyptus* plantation. The C concentration of the free LF we report here is half of that reported by Besnard et al. (1996) for a French forest soil, whereas the N concentration is comparable; their values averaged about 250 mg OC and 9 mg N g^{-1} light fraction. This site difference could have resulted from variations in climate and vegetation type, and partly from the removal of large and recognisable plant materials during the sieving of our samples.

3.5. Intra-particulate OC, N, and S

After the conversion of the natural forest into the *Eucalyptus* plantation 21 years ago, total iPOM C, N and S concentrations in the 0–20 cm soil depth were significantly reduced (Table A6). The mean C/N and C/S ratios of the iPOM were also significantly narrowed from 17 and 108 in the natural forest to 9 and 50 in the *Eucalyptus* plantation, respectively, suggesting that soil organic matter under *Eucalyptus* plantation has undergone more decomposition. However, the mean N/S ratio was not significantly different between the two land use types. According to Jastrow (1996) and Six et al. (1998), the amount of total occluded POM C and nutrients per unit soil is mainly a function of aggregation., whereas the free light POM C i.e., LF C is mostly affected by residue input. In this study, although the *Eucalyptus* plantation had a slightly higher iPOM dry weight (data not shown) and nearly the same level of soil aggregation (Table A4) as in the natural forest, the losses in aggregate-protected OC and N (Table A6) were more pronounced than losses from the free LF (Table A5). This could be due to (i) a year-round input of organic material to the LF material after refforestation and (ii) gaseous losses of OM inside the aggregates caused by high fire temperatures during clearing and site preparation; otherwise biodegradation is normally nearly three times as fast outside aggregates as within them (Besnard et al., 1996). Buschiazzo et al. (2001) linked the large decrease of OC after cultivation of a forest soil to the occurrence of natural fire before cultivation. In our study, in contrast to the observations of Besnard et al. (1996) in French forest soils, the LF OC, N, and S concentrations in both forest types were always larger than the iPOM C, N, and S values. In the natural forest, the amounts of iPOM C, N, and S comprised 8, 6 and 4%, respectively, of the whole soil; while in the plantation these values were low, amounting 4% for OC, 2% for N, and 3% for S.

Table A6. Characteristics of the intra-particulate organic matter fractions to the soil depth of 0–20 cm.

	C	N	S			
		(mg g^{-1} fraction)		C/N	C/S	N/S
Natural forest						
Macroaggregates	44**a*** (17)	2.5**a*** (0.8)	0.3**a*** (0.07)	17**a*** (0.9)	155**a*** (21)	9**a** (0.7)
Microaggregates	14**b** (3)	0.9**b** (0.2)	0.3**a*** (0.1)	16**a*** (2)	61**b** (25)	4**b** (1.6)
***Eucalyptus* plantation**						
Macroaggregates	8**a** (2)	0.9**a** (0.1)	0.13**a** (0.03)	9**a** (0.8)	62**a** (5)	7**a** (0.7)
Microaggregates	5**a** (0.6)	0.6**a** (0.0)	0.13**a** (0.03)	8**a** (1)	38**b** (5)	5**a** (1)

Values followed by different lower case letters within a land use are significantly different (P<0.05). Values followed by * in the natural forest are significantly different compared with the corresponding values in the *Eucalyptus* plantation. Numbers in parentheses are standard errors (n=3).

In the natural forest, significantly larger amounts of iPOM C and N were contained in macroaggregates than in the microaggregates, which contained only 36% of the C and 43% of the

N contained in macroaggregates. However, the concentrations of iPOM S in the natural forest and iPOM C, N and S in the Eucalyptus plantation were similar in both aggregate size fractions (Table A6). In both land use types, no significant differences were detected in iPOM element ratios between the macro and microaggregates (Table A6). In a study by Cadish et al. (1996) and Six et al. (1998) macroaggregate iPOMC and N concentrations were found to be higher than microaggregate iPOMC and N concentrations. Macroaggregate iPOM C, N and S concentrations in the natural forest were significantly higher than either macro or microaggregate iPOM C, N and S concentrations in the *Eucalyptus* plantation (Table A6), where as the changes in microaggregate associated element concentrations and element ratios, with the exception of S and C/N ratio, were not significant. This confirms the conclusions of Elliott (1986), Gupta and Germida (1988) and Besnard et al. (1996) that organic matter associated with macroaggregates is more labile than organic matter associated with microaggregates. Because floatable and easily recognizable materials were removed during aggregate size fractionation, the C/N ratios of the LF and iPOM fractions did not differ from one another in either forest types. In a similar study dealing with French forest soils, Besnard et al. (1996) even with out removing the floatable and easily recognisable materials also found that C/N ratios of LF and iPOM did not differ significantly from one another.

4. Conclusions

Bulk soil OC, N and S concentrations did not show significant change as a result of changes in land-use and management. However, physical fractionation of the soils into size and size/density fractions clearly showed the effect of land use and management on the quantity and quality of SOM. Our results showed that organic matter associated with the sand and silt fractions appeared to be more sensitive to changes in land use and management compared with that in clay.

Similarly, both free LF and iPOM associated with macroaggregates were found to be much more affected by changes in land use and management than that associated with microaggregates. Although the level of soil aggregation was the same between the two land-use types, loss of aggregate protected POM was higher than that of free LF after conversion to *Eucalyptus* plantation. Our results suggest that the amount of aggregate protected OM is not only influenced by aggregation but also by soil management. We conclude that high fire temperature used for site clearing and preparation resulted in gaseous losses of aggregate protected POM in the *Eucalyptus* plantation. As a result the observed change in POM resulting from changes in land-use was more evident on the quality than on the total amount. In general, losses of OM associated with the different size and size/density fractions resulting from the conversion of the natural forest to the *Eucalyptus* plantation were more pronounced than losses observed in the bulk soil.

Acknowledgements

We thank the German Research Foundation (DFG) for funding this work. We are grateful to Katholischer Akademischer Ausländer-Dienst (KAAD) for a postgraduate scholarship to the first author.

References

Anderson, D.W., Saggar, S., Bettany, J.R., Stewart, J. W.B., 1981. Particle size fractionation and their use in studies of soil organic matter: I. The nature and distribution of forms of carbon, nitrogen and sulfur. Soil Sci. Soc. Am. J. 45, 767–772.

Balesdent, J., Mariotti, A., 1996. Measurement of soil organic matter turnover using ^{13}C natural abudance. In: Boutton, T.W., and Yamasaki, S.I. (Eds.), Mass Spectrometry of Soils. Marcel Dekker, New York, pp. 83–111.

Besnard, E., Chenu, C., Balesdent, J., Puget, P., Arrouays, D., 1996. Fate of particulate organic matter in soil aggregates during cultivation. Euro. J. Soil Sci. 47, 495−503.

Biederbeck, V.O., Janzen, H.H., Campbell, C.A., Zentner, R.P., 1994. Labile soil organic matter as influenced by cropping practices in an arid environment. Soil Biol. Biochem. 26, 1647−1656.

Brown, S., Lugo, A.E., 1990. Effets of forest clearing and succession on the carbon and nitrogen content of soils in Puerto Rico and US Virgin Islands. Plant Soil 124, 53−64.

Buschiazzo, D.E., Hevia, G.G., Hepper, E.N., Urioste, A., Bono, A.A., Babinec, F., 2001. Organic C, N and P in size fractions of virgin and cultivated soils of the semi-arid pampa of Argentina. J. Arid Environ**.** 48, 501−508.

Buyanovsky, G.A., Aslam, M., Wagner, G.H., 1994. Carbon turnover in soil physical fractions . Soil Sci. Soc. Am. J. 58, 1167−1173.

Cadisch, G., Imhof, H., Urquiaga, S., Boddey, R.M., Giller, K.E., 1996. Carbon turnover ($\delta^{13}C$) and nitrogen mineralization potential of particulate light soil organic matter after rainforest clearing. Soil Biol. Biochem. 28, 1555−1567.

Cambardella, C.A., Elliott, E.T., 1992. Particulate organic matter changes across a grassland cultivation sequence. Soil Sci. Soc. Am. J. 56, 777−783.

Cambardella, C.A., Elliott, E.T., 1993. Carbon and nitrogen distribution in aggregates of cultivated and native grassland soils. Soil Sci. Soc. Am. J. 57, 1071−1076.

Cambardella, C.A., Elliott, E.T., 1994. Carbon and nitrogen dynamics of soil organic matter fractions from cultivated grassland soils. Soil Sci. Soc. Am. J. 58, 123−130.

Christensen, B.T., 1992. Physical fractionation of soil and organic matter in primary particle size and density separates. Adv. Soil Sci. 20, 1−76.

Christensen, B.T., 1996. Carbon in primary and secondary organo-mineral complexes. In: Carter, M.R., Stewart, B.A., (Eds.), Structure and Organic Matter Storage in Agricultural Soils. Advances in Soil Science. CRC. Lewis Publisher, Boca Raton, FL, pp. 58‒97.

Christensen, B.T., 2001. Physical fractionation of soil and structural and functional complexity in organic matter turnover. Euro. J. Soil Sci. 52, 345‒353.

Desjardins, T.F., Andreux, F., Volkoff, B., Cerri, C.C., 1994. Organic carbon and 13-C contents in soils and soil size fractions, and their changes due to deforestation and pasture installation in eastern Amazonia. Geoderma 61, 103‒118.

EFAP(Ethiopian Forestry Action Plan).1994. Ethiopian Forestry Action Plan. MoNRDEP, Addis Ababa, Ethiopia. Volume I.14p. and volume II. pp. 21‒34

Elliott, E.T., 1986. Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. Soil Sci Soc. Am. J. 50, 627‒633.

FAO.(1997) FAO/UNESCO Soil map of the world, revised legend with corrections and updates. World soil resources, Report No. 60, ISRIC, Wageningen, The Netherlands.

Gerzabek, M.H., Haberhauer, G., Kirchmann, H., 2001. Soil organic matter pools and carbon-13 natural abundances in particle-size fractions of a long-term agricultural field experiment receiving organic amendments. Soil Sci. Soc. Am. J. 65, 352‒358.

Guggenberger, G., Zech, W., Haumaier, L., Christensen, B.T., 1994. Land use effects on the composition of organic matter in soil particle size separates of soil: I. Lignin and carbohydrate signature. Euro. J. Soil Sci. 45, 449‒458

Gupta, V.V.S.R., Germida, J.J., 1988. Distribution of microbial biomass and its activity in different soil aggregate size classes as affected by cultivation. Soil Biol. Biochem. 20, 777‒786.

Haynes, R.J., 1999. Labile organic matter fractions and aggregate stability under short-term, grass-based leys. Soil Biol. Biochem. 31, 1821‒1830.

Haynes, R.J., Beare, M.H., 1996. Aggregation and organic matter storage in meso-thermal humid soils. In: Carter, M.R., Stewart, B.A., (Eds.), Structure and Organic Matter Storage in Agricultural Soils. Advances in Soil Science.CRC. Lewis publisher, Boca Raton, FL pp. 194‒213.

Janzen, H.H., Campbell, C.A., Brandt, S.A., Lafond, G.P., Townley-Smith, L., 1992. Light fraction organic matter in soils from long term crop rotations. Soil Sci. Soc. Am. J. 56, 1799‒1806.

Jastrow, J.D., 1996. Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. Soil Biol. Biochem. 28, 665‒676.

Mahieu, N., Powlson, D.S., Randall, E.W., 1999. Statistical analysis of published carbon-13 CPMAS NMR spectra of soil organic matter. Soil Sci. Soc. Am. J. 63, 307‒319.

Michelsen, A., Lisanework, N., Friis, I., 1993. Impacts of tree plantations in the Ethiopian highland on soil fertility, shoot and root growth, nutrient utilization and mycorrhizal colonization. For. Ecol. Manage. 61, 299‒324.

Pohjonen, V., Pukkala, T., 1990. *Eucalyptus globulus* in Ethiopian forestry. For. Ecol. Manage. 36, 19‒31.

Six, J., Elliott, E.T., Paustian, K., Doran, J.W., 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. Soil Sci. Soc. Am. J. 62, 1367‒1377.

Solomon, D., Fritzsche, F., Tekalign, M., Lehmann, J., Zech, W., 2002. Soil organic matter composition in the sub-humid Ethiopian highlands as influenced by deforestation and agricultural management. Soil Sci. Soci. Am. J. 66, 68–82.

Zinn, Y.L., Resck, D.V.S., Silva, J.E., 2002. Soil organic carbon as affected by afforestation with *Eucalyptus* and *Pinus* in the *Cerrado* region of Brazil. For. Ecol. Manage. 166, 285‒294.

B Soil aggregation, and total and particulate organic matter as affected by conversion of native forests to 26 years of continuous cultivation in Ethiopia

Y. Ashagrie[1*], W. Zech[1], G. Guggenberger[1,3] and T. Mamo[2]

[1]*Institute of Soil Science and Soil Geography, University of Bayreuth, D-95440 Bayreuth, Germany*
[2]*Winrock International, Addis Ababa, Ethiopia*
[3]*Present address: Institute of Soil Science and Plant Nutrition, Martin Luther University, D-06099 Halle, Germany*
* Correspondence: yeshanew@hotmail.com, Tel: ++49-921-55-21-46 Fax: ++49-921-55-22-46

Abstract

Conversion of native forests to cultivation is usually accompanied by a decline in soil organic carbon (SOC) and nutrients, and deterioration of soil structure. The effect of 26 years of continuous cultivation was studied on soil aggregation, and total and particulate organic matter in a Rhodic Nitisol at Munesa, south eastern Ethiopia. The objectives of this study were (i) to assess the effect of cultivation on aggregate stability, (ii) to evaluate the hierarchical model of soil aggregation and the effect of soil aggregation on soil organic matter (SOM) protection, and (iii) to determine the effect of cultivation on the quantity and quality of particulate organic matter (POM). Samples were collected from a cropland cultivated for 26 years and an adjacent natural forest. After cultivation, the proportion of water-stable macroaggregates was significantly reduced from >70% in the natural forest soil to 50% in the cultivated soil, being more pronounced in the >1 mm size aggregates. Cultivation also induced significant losses of OC and N both in bulk soil and water-stable aggregates. The OC and N associated with the larger aggregates were more affected by cultivation than the smaller aggregates. The amounts of free light fraction (free LF) C and N were more affected by cultivation than the amounts of intraparticulate organic matter (iPOM) C and N. POM C and N associated with the macroaggregates were highest compared to those of microaggregates and the effect of cultivation was more pronounced on macroaggregates associated POM relative to the microaggregates. The effect of cultivation on POM C and N was more pronounced than on total aggregate and whole soil OC and N suggesting that POM constitutes

a more sensitive soil organic matter (SOM) fraction to the effects of cultivation. The data show that after 26 yr continuous cultivation both the physical and chemical properties of the soil are deteriorated.

Key words: Aggregate stability, POM, Cultivation, Nitisol, Ethiopia.

1. Introduction

Maintenance of SOM is important for sustainable use of soil resources due to the multiple effects of SOM on soil nutrient status and soil structural stability. Conversion of native forests to cultivation is usually accompanied by a decline in SOC and nutrients, and deterioration of soil structure (Dormaar, 1983, Detwiler, 1986; Brown and Lugo, 1990; Balesdent et al., 1998; Spaccini et al., 2001; Solomon et al., 2002). Cultivation effects on SOM are caused by complex interactions of the physical, chemical and biological soil processes including reduced inputs of plant residues and increased soil disturbance, but the exact nature of the changes induced by cultivation depend on the particular agronomic practices adopted and on the properties of the virgin soil (Christensen, 1992). Identification of the magnitude of such management induced changes in SOM and associated nutrients is needed to select appropriate management options.

Tillage of a soil breaks soil aggregates and exposes the previously protected OM within aggregates to microbial decomposition. According to the conceptual models of soil aggregation (Oades, 1984; Oades and Waters, 1991), aggregates of different sizes have different strength, and are stabilised by different agents. On the basis of their temporal persistence, Tisdall and Oades (1982) classified the organic binding agents of soil aggregates into: (i) transient or temporary such as polysaccharides, roots, fungal hyphae, bacterial cells, and algae which are responsible for stabilising macroaggregates (0.25−2 mm) and (ii) persistent aromatic humic materials associated with polyvalent metal cations and polymers strongly sorbed to clays mainly responsible for the integrity of the microaggregates (0.05 to

0.25 mm). The organic binding agents keep the aggregates intact and protect them against deformation from heavy rain drop impacts especially in the tropics (Spaccini et al., 2001).

Due to the labile nature of their binding agents, the effect of soil management on macroaggregates and the OM retained in it are greater than those on microaggregates with its stable and more humified organic binding agents (Oades, 1984; Elliott, 1986; Gupta and Germida, 1988; Miller and Jastrow, 1990; Cambardella and Elliott, 1993; Puget et al., 1995; Dalal and Bridge, 1996). Several authors have demonstrated that macroaggregates contain more OC and N than microaggregates (Elliott, 1986; Gupta and Germida, 1988; Beare et al., 1994a, 1994b; Cambardella and Elliott, 1993; Puget et al., 1995; Spaccini et al., 2001). The decline in total SOM during cultivation of native grassland and forest soils has been largely attributed to losses of POM (Besnard et al., 1996; Buyanovsky et al., 1994; Cambardella and Elliott, 1992, 1994; Janzen et al., 1992; Lehmann et al., 2001; Six et al., 1998). Large stocks of POM in a soil have been associated with pronounced mineralization of organically-bound nutrients such as N and are therefore intimately linked to higher soil fertility and productivity (Yakovchenko et al., 1998). POM can also serve as a sensitive indicator of changes in SOM because of its responsiveness to management (Janzen et al., 1992; Dalal and Mayer, 1987).

Except a few (Elliott et al., 1991; Lehmann et al., 2001, Spaccini et al., 2001), studies dealing with sensitive OM fractions have focused on few soil groups of temperate ecosystems only. But soils developed under different vegetation types and climate may have different modes of SOM stabilisation (Elliott, 1986). The importance of aggregate formation and stabilisation in regulating the accumulation or loss of SOM and nutrients in differently managed tropical soils is not well understood. Understanding these relationships may be of particular importance for evaluating the applicability of the hierarchical model of aggregate formation (Tisdall and Oades, 1982) to a broader range of soils and management conditions, and for developing management options for sustainable crop production systems. The objectives of this study were (i) to assess the effect of cultivation on aggregate stability, (ii) to evaluate the hierarchical model of aggregation and its effect on soil organic matter (SOM) protection, and (iii) to determine the effect of cultivation on the quantity and quality of particulate organic matter (POM).

2. Materials and methods

2.1. The study site

The study was conducted at the Munesa/Shashemenie forest enterprise site (7°34´N and 38°53´E) located about 240 km south east of Addis Ababa at an altitude of 2400 m. The area has a sub-humid tropical climate with a bimodal rainfall pattern most of it falling in July and August. Mean annual rainfall is about 1250 mm, and mean annual temperature is 19 °C with little seasonal variation. The soils are clayey and very deep with reddish brown colour, and are moderately acidic at or near the surface and slightly acidic at depth. The principal parent materials are of volcanic origin from which Rhodic Nitisols were derived (FAO, 1997). A *Podocarpus falcatus* dominated mixed natural forest and an adjacent cropland situated side by side were selected for this study. The cropland was established after clearing of part of the natural forest some 26 years ago and has been used continuously for annual crops such as maize (*Zea mays*), bread-wheat (*Triticum aestivum*), faba bean (*Vicia faba*) and sorghum (*Sorghum bicolor*) and fertilised with di-ammonium phosphate annually depending on the crops' fertiliser requirement.

2.2. Sampling

We chose three 0.06 ha plots in the natural forest and three 0.04 ha plots in the cropland randomly, and a pit was excavated to the depth of 1.2 m at the centre of each plot. In addition, four 1 m^2 sub plots were marked randomly at 6−10 m radius from the centre of each plot. Soil samples ca. 500 g were taken from the three sides of the pit by a shovel and, at three points within each of the 1 m^2 sub plots by an auger to the depth of 0−20 cm. All the auger and pit samples in each plot were mixed and the final number of samples was reduced to three per land use. After air drying, a sub sample was sieved through 5 mm sieve size for aggregate fractionation, and the remaining was sieved through 2 mm sieve size for bulk soil C and N analysis. Soil samples for bulk density determination were taken from the three sides of each pit with a 100 cm^3 metal ring.

2.3. Soil aggregate size fractionation and separation of POM

The size distribution of soil aggregates was measured by wet sieving technique following the procedures of Cambardella and Elliott (1993). A 70–80 g sample of air dried soil that passed through a 5 mm sieve size was spread on top of five stacked sieves (2, 1, 0.5, 0.25 and 0.053 mm) submerged in a bucket of deionized water. The water level was adjusted so that the aggregates on the upper sieve were just submerged. Soils were left immersed in the water for 10 min and then sieved by moving the sieve 3 cm vertically 50 times during a period of 2 min. During the sieving process, floatable materials >2 mm were removed and discarded. The material remaining on the 2 mm sieve was transferred to a glass pan. Soils plus water that passed through the sieve was poured onto the next finer sieve and the processes repeated, but floatable materials were not removed and discarded. The different aggregate sizes were dried in the oven at 50°C overnight for chemical analysis. The mean weight diameter (MWD) of water stable aggregates was then determined as the sum of the percentage of soil on each sieve multiplied by the mean diameter of adjacent sieves i.e. MWD=∑ (percent of sample on sieve x mean intersieve size).

The separation of POM was done following the procedure of Six et al. (1998). The fractions in the >0.25 mm size aggregates were bulked as macroaggregates, and the 0.053 to 0.25 mm size was taken as microaggregates. After drying (105°C) in the oven overnight and cooling in a decicator to room temperature, about 10 g macro and microaggregates was suspended in 35 ml sodium polytungstate adjusted to a density of 1.8 g cm^{-3} in a conical centrifuge tube and hand-shaken until all the material was suspended. The suspension was allowed to stand for 20 min before centrifugation at 1250 rpm for 60 min. After centrifugation, the floating material was collected on 2 µm pore size filter paper and rinsed thoroughly with deionized water to remove sodium polytungstate. The material in the < 1.8 g cm^{-3} fraction is referred to as free LF. The heavy fraction remaining in the tube was washed twice with 50 ml deionized water and dispersed in 50 ml of 5% sodium hexametaphosphate by shaking in a reciprocal shaker for 18 hours. The dispersed heavy fraction was rinsed through a 0.053 mm sieve with deionized water. The material remained on the sieve is iPOM + sand. Both the free LF and iPOM + sand were dried in the oven at 50°C overnight. The dried sub samples from each aggregate size class, and the free LF and iPOM + sand were finely ground in a rotary ball mill for chemical analysis.

2.4. Laboratory analysis

Bulk density was determined after drying the soil in an oven at 105°C. Organic C and N concentrations in bulk soil, size fractions and POM were determined using CHNS−analyzer (Vario EL, Elementar Analysensysteme GmbH, Hanau, Germany).

2.5. Statistical analysis

The experiment had a split-plot design (three replications) with land use as a main plot and soil aggregate size as a subplot. Analysis of variance (ANOVA) was conducted using MSTAT-C version 2. One way ANOVA was conducted to detect significant differences between land use and, among aggregate size means within a land use. Two way ANOVA was performed to test significant differences in aggregate size means between land use types.

Significant treatment means were separated using Tukey´s honestly significance difference test (HSD) at $P < 0.05$.

3. Results and discussion

3.1. Aggregate size distribution and stability

Table B1 shows that, in the forest soil, after 10 min of slaking, most soil was found in 0.25 to 2 mm size macroaggregates and to a lesser extent in microaggregates (0.053 to 0.25 mm). In contrast, in the cultivated soil, significantly large proportion of the soil was retained as microaggregates and small macroaggregates (0.25 to 0.5 mm).

Table B1. Distribution and MWD of water-stable aggregates after 26 years continuous cultivation of the natural forest soil.

Aggregate size	Distribution (%)		MWD (mm)	
(mm)	Natural forest	Cultivated	Natural forest	Cultivated
2–5	10**bA** (0.22)	0.8**eB** (0.09)	0.35**aA** (0.01)	0.03**cB** (0.003)
1–2	21**aA** (1.4)	**5dB** (0.87)	0.32**aA** (0.02)	0.08**bB** (0.01)
0.5–1	21**aA** (1.3)	17**cA** (0.62)	0.16**bA** (0.01)	0.13**aA** (0.005)
0.25–0.5	21**aA** (1.6)	27**bA** (2.4)	0.08**cA** (0.01)	0.10**abA** (0.01)
0.053–0.25	14**bB** (2.6)	35**aA** (0.84)	0.01**dA** (0.002)	0.02**cA** (0.001)
Total	87	85	0.92	0.36

Means followed by the same lower case letter in a column and by the same upper case letter in a row are not significantly different. Numbers in parentheses are standard errors (n=3).

After 26 years continuous cultivation, the amount of water stable macroaggregates was significantly reduced from > 70% in the natural forest soil to 50% in the cultivated soil, while that of microaggregates increased by a factor of 2.5, indicating that cultivation resulted in the structural degradation of this soil. This could be attributed to the breakdown of aggregates by tillage, differences between the two land use types in annual organic matter input which gives cementing agents and the enmeshing effects of roots and associated mycorrhizal hyphae. These results confirm earlier observations that macroaggregates are dynamic in nature and the size distribution of macroaggregates is affected by the change in land use and management (Dormaar, 1983; Elliott, 1986; Beare et al, 1994b; Puget et al., 1995; Spaccini et al. 2001). The relatively higher proportions of soil in the macroaggregates of the forest soil further suggest that the soil aggregates under forest were not greatly affected by slaking or were more stable than the cultivated soil.

There was a significant land use x aggregate-size interaction on the distribution of water stable aggregates indicating that the effect of cultivation was much more evident in the larger macroaggregates i.e >1–2 mm size than the smaller macroaggregate-size classes (Table B1). As with the findings of Haynes (1999) in pasture soil and Spaccini et al. (2001) in forest soil, in this study, the >2–5 mm and >1–2 mm classes of the forest soil were 13 and 4 times, respectively, larger than in the cultivated soil (Table B1). The relatively higher level of reduction in larger macroaggregates compared to the smaller macroaggregates up on cultivation could be mainly because the former are largely dependent on live and decaying plant roots and fungal hyphae and probably casts of earthworms and termites which are rapidly destroyed by tillage (Tisdall and Oades, 1982). Beare et al. (1994b) also reported a reduction in the >2 mm aggregates of cultivated surface soil and redistribution of particles to smaller size classes in conventional tillage than in no tillage soil. Tisdall and Oades (1979) and Angers (1992) reported that the effects of cropping treatments on soil aggregates were mostly apparent in the >2 mm size fractions. A greater shift in water stable aggregates from large macroaggregates to smaller macroaggregates and microaggregates up on cultivation had also led to a significant reduction of MWD from 0.92 mm in the forest soil to 0.36 mm in the cultivated soil (Table B1). Spaccini et al. (2001) reported MWD reductions of 37 to 76% on cultivated Ethiopian Vertisols, Alfisols, Entisols, and Andisols relative to the forest soil, being highest in Vertisols and lowest in Andisols.

3.2. Whole soil C and N

Conversion of the natural forest into continuous cultivation had resulted in significant reductions of both the concentrations and stocks of OC and N (Table 2). C/N ratio was also significantly narrowed from 12 in the forest soil to 9 in the cultivated soil (Table B2). The substantial losses of organic C and N after 26 years of cultivation were expected since the break-up of soil aggregates and increased aeration caused by tillage both favour decomposition of soil organic matter. In addition, reduced inputs of organic matter because of the removal of large amounts of above-ground biomass at harvest and burning of crop residues during land preparation are also responsible for the lower C and N content of the cultivated soil. Comparable losses of SOC and nutrients due to cultivation of forest soils have been reported in many studies (Brown and Lugo, 1990; Davidson and Akerman, 1993; Buschiazzo et al., 2001; Spaccini et al. 2001; Solomon et al., 2002).

Table B2. Bulk soil chemical and physical properties under the different land-use types, results refer to the 0-20 cm soil depth.

	C ($g\ kg^{-1}$)	N	C/N	Bd ($g\ cm^{-3}$)	C ($kg\ m^{-2}$)	N
Natural forest	72**a**	6**a**	12**a**	0.86**b**	12.4**a**	1.0**a**
	(7.4)	(0.6)	(0.6)	(0.02)	(0.9)	(0.01)
Cultivated	34**b**	3.9**b**	9**b**	0.99**a**	6.7**b**	0.76**b**
	(4.9)	(0.5)	(0.2)	(0.03)	(0.2)	(0.03)

Bd: bulk density

Means followed by different letters in a column are different from each other. Numbers in parentheses are standard errors (n=3).

3.3. Total OC and N of aggregates

Data on the OC and N concentrations ($g\ kg^{-1}$ aggregates) and, C/N ratio of the different aggregate size classes are reported in Table B3. In the soil under natural forest, none of the parameters showed significant differences among aggregate size classes. In contrast, in the cultivated soil, the OC and N concentrations were significantly different among the different size classes, and appeared to decrease as size increases from 0.053 to 2 mm diameter (Table B3). This could be attributed partly to the redistribution and / or transfer of organic matter from the large aggregates to smaller ones either in the process of biodegradation or by mechanical disruption of the large macro-aggregates (Dormaar, 1983; Christensen, 1992). Oades and Waters (1991) suggested that when roots and hyphae that hold the macroaggregates die and disrupted by tillage or fauna, the decomposed fragments probably become the organic core in the 0.02 to 0.25 mm size microaggregates. Alternatively, the extent of OM decomposition under arable use may be different for the different aggregate fractions. Our results in the cultivated soil contrast with the observations of Elliott (1986), Cambardella and Elliott (1993) and Puget et al. (1995), who observed an increase in OC concentration of the cultivated soil with an increase in diameter of the fractions. The OC and

N concentrations associated with each maroaggregate size in the natural forest were two to threefold higher than the corresponding values in the cultivated soil, although the differences generally are not statistically significant. The effect of cultivation was more pronounced on OC than on N. This was further reflected by a significantly narrower mean C/N ratio in the cultivated soil aggregates (9–10) than in the forest soil aggregates (12–13). In both land use types, C/N ratios of the bulk soil and the different water stable aggregates were nearly the same (Tables B2 & B3).

Table B3. Organic C and N concentrations (g kg^{-1} aggregate) and C/N ratios of soil aggregate size classes to the depth of 0−20 cm as affected by 26 years continuous cultivation.

Aggregate size	OC		N		C/N	
(mm)	NF	Cu	NF	Cu	NF	Cu
2-5	78**a** (6.1)	23**c** (2)	6.4**a** (0.5)	2.5**d** (0.1)	12**a** (0.8)	9**a** (0.2)
1-2	65**a** (4.4)	27**bc** (1)	5.5**a** (0.2)	2.9**cd** (0.3)	12**a** (0.5)	10**a** (0.7)
0.5-1	84**a** (32)	33**ab** (4)	7.1**a** (2.4)	3.7**ab** (0.5)	12**a** (0.6)	9**a** (0.03)
0.25-0.5	74**a** (24)	31**b** (1)	6.2**a** (1.6)	3.4**bc** (0.1)	12**a** (1.3)	9**a** (0.1)
0.053-0.25	57**a** (11)	40**a** (1)	4.6**a** (0.3)	4.3**a** (0.03)	13**a** (0.8)	9**a** (0.2)

NF: Natural forest; Cu: cultivated. In a column, means followed by the same lower case letter are not significantly different. Numbers in parentheses are standard errors (n=3).

Mean total amounts of OC and N (g kg^{-1} whole soil) of the different aggregate size classes did not significantly differ between the two land use types (Table B4), but land use x aggregate size interaction was significant. With the exception of N in the 0.25-0.5 mm size class, the amounts of C and N in the different macroaggregate size classes of the cultivated soil were significantly lower than those in the natural forest. However, the amounts of microaggregate associated C and N were significantly higher in the cultivated soil than that of the natural forest mainly due to large proportion of soil in this size class. In the natural forest, although the proportion of water stable aggregates was different among the different size classes, differences in the amounts of C and N (g kg^{-1} whole soil) among the different size classes did not reach significance.

Table B4.Total amounts of OC and N (g kg^{-1}whole soil) associated with each aggregate-size in natural forest and cultivated field soils.

	Size fractions (mm)				
	2−5	1–2	0.5−1	0.25–0.5	0.053–0.25
Organic C					
Natural forest	8**a** (0.4)	14**a** (1.4)	18**a** (5.7)	16**a** (3.5)	8**b** (1.8)
Cultivated	0.18**bD** (0.02)	1.4**bD** (0.24)	5.6**bC** (0.65)	8.4**bB** (1.0)	14**aA** (0.11)
Total N					
Natural forest	0.64**a** (0.04)	1.2**a** (0.09)	1.5**a** (0.43)	1.3**a** (0.22)	0.64**b** (0.14)
Cultivated	0.02**bD** (0.003)	0.15**bD** (0.03)	0.63**bC** (0.09)	0.93**aB** (0.08)	1.5**aA** (0.03)

Means followed by the same lower case letter in a column and by the same upper case letter in a row are not significantly different. Numbers in parentheses are standard errors (n=3).

In contrast, in the cultivated soil, the amounts of C and N were significantly different among the different aggregate sizes; increasing steadily as the proportion of water stable aggregates and concentrations of C and N increased (Tables B1, B3 &B4). Significant reductions in OC and N concentrations due to cultivation of a native vegetation soil were reported by Buschiazzo et al. (2001) from macroaggregates of Typic Ustipsament. Spaccini et al. (2001)

reported better protection of carbohydrates associated with smaller aggregate size classes for Ethiopian and Nigerian soils when forests are converted to cultivation.

The relationship between WSA and, OC and N concentrations did not show significant coorelations (data not shown) suggesting that other factors such as inorganic soil constituents (Tisdall, 1996) and, the arrangement of the organic compounds other than the absolute organic matter quantity present might be responsible for the aggregation of this tropical soil (Hamblin and Greenland, 1977; Dormaar, 1983). It has been suggested that diverse organic and inorganic constituents participate in the binding of soil particles in to water stable aggregates and the relative importance of each varies in differing situations (Haynes and Beare, 1996). Our results agree with Dormaar (1983) and Lehmann et al. (2001), who observed no relationship between WSA and organic carbon.

3.4. Free LF and iPOM C and N

The mean C and N concentrations (g kg^{-1} fraction) in both the free LF and iPOM fractions were significantly reduced after cultivation with much of the reductions occurring from the macroaggregate associated fractions (Table B5). The effect of cultivation was more pronounced on the iPOM C than on the free LF C. This was further evidenced by a significantly narrower iPOM C/N ratio under cultivation than the natural forest (Table B5) suggesting that much of the readily decomposable components have been lost, leaving old and more humified components. However, the free LF C/N ratio (Table B5) was not significantly changed after cultivation probably due to recent inputs of less decomposed fresh organic residues from roots of the previous crop. The mean C/N ratios of the free light and iPOM fractions in both land use types were relatively wider than the corresponding whole soil C/N ratios (Tables B2&B5). Christensen (1992) suggested that SOM in LF has usually a wider C/N ratio than in whole soils and heavy fractions. In the present study, no significant difference was detected between the C/N ratios of free LF and iPOM in either of the land use types probably due to the removal of free and released floatable easily recognisable fresh plant residues from the samples soon after slaking. However, this is not unusual as Besnard et al., (1996) also found no significant difference in POM C/N ratio of different positions.

Table B5.Characteristics and total amounts of POM associated with each aggregate size in natural forest and cultivated field soils

	C (g kg^{-1} fraction)	N (g kg^{-1} fraction)	C/N	C (g kg^{-1}whole soil)	N (g kg^{-1}whole soil)
Natural forest					
Free LF					
Macroaggregates	138**a*** (0.6)	9.9**a*** (0.04)	14**a** (0.1)	38**a*** (9.3)	2.7**a*** (0.6)
Microaggregates	84**b*** (1.2)	5.2**a** (0.1)	17**a*** (2.2)	1.5**b** (0.8)	0.10**b*** (0.06)
iPOM					
Macroaggregates	44**a*** (1.7)	2.5**a** (0.08)	17**a*** (0.9)	5.6**a** (2.4)	0.31**a** (0.1)
Microaggregates	14**b*** (0.3)	0.9**a** (0.02)	16**a*** (2.1)	0.3**b** (0.01)	0.02**a** (0.08)
Cultivated					
Free LF					
Macroaggregates	59**a** (0.3)	4.6**a** (0.02)	13**a** (0.3)	3.9**a** (0.8)	0.3**a** (0.06)
Microaggregates	38**a** (0.3)	3.1**a** (0.03)	12**a** (0.1)	0.39**b** (0.1)	0.03**a** (0.01)
iPOM					
Macroaggregates	12**a** (0.1)	1.2**a** (0.02)	10**a** (0.5)	1.5**a** (0.2)	0.15**a** (0.02)
Microaggregates	4.3**b** (0.04)	0.43**a** (0.003)	10**a** (1)	0.3**a** (0.07)	0.03**a** (0.01)

Means followed by the same letter within land use for each POM fractions associated with macro and microaggregates are not different. Means followed by * in the natural forest are significantly different from the corresponding values in the cultivated soil. Numbers in parentheses are standard errors (n=3).

The total amounts of the free LF and iPOM C and N (g kg^{-1}whole soil) were significantly larger in the natural forest than in the cultivated soil (Table B5). Cultivation had resulted in the depletion of over 80% of the free LF C and N, and 69% of the iPOM C and 45% of iPOM N. This could be attributed mainly to the reduced inputs of organic matter and, faster biodegradation of POM due to the more favourable environmental conditions for biological activity such as higher temperature and moisture, and aeration created by tillage under the cultivated soil. The relative losses of POM C and N were larger than those observed in the whole soil and in water stable aggregates, indicating that POM constitutes soil organic matter fraction more sensitive to the effects of cultivation. Similar results have been reported for situations dealing with the conversion of native forest to corn cultivation (Besnard et al., 1996) and native grassland soil to cultivation (Cambardella and Elliott, 1992; Six et al., 1998). After 35 years continuous cultivation of a soil under virgin brigalow (*Acacia harpophylla*), Skjemdstad et al. (1986) reported as much as 95% OC loss from the light fraction. Gregorich et al.(1997) also reported losses of substantial amounts of both free and physically protected organic matter after 32 years cultivation of a sod soil. Averaged over aggregate sizes, much of both the free LF and the iPOM C and N were associated with the macroaggregates compared to the microaggregates, with the LF accounted for much of the POM C and N associated with both aggregate size fractions.

A significant land use x aggregate size interaction indicates that the amounts of the free LF C and N associated with both macro and micro aggregates were significantly larger in the forest soil than in the cultivated soil (Table B5). With respect to the amounts of iPOM C and N, however, the effect of cultivation was evident only in the macroaggregate associated fraction due to the reduction of the proportion of macroaggregates by tillage and due to the labaile nature of macroaggregate associated OM to accelerated decomposition induced by cultivation (Skjemstad et al., 1990; Buyanovsky et al., 1994; Jastrow et al., 1996). Jastrow (1996) found relatively higher proportions of POM C inside the macroaggergates of the virgin prairie soil compared to corn field and restored prairie soil. Six et al. (1998) also reported higher iPOM levels in water stable macroaggregates sampled from native sod soil than those from the cultivated soil. In the natural forest, the proportion of whole soil organic C and N found in free LF accounted for 55% and 47%, respectively, and that of iPOM C and N comprised 8% and 6%, respectively. After 26 years cultivation, the free LF C and N represented 13% of whole soil OC and 8% of whole soil N and the corresponding proportions in iPOM C and N were 5%. These results indicate that the free LF accounted for much of the losses of whole soil OC and N due to cultivation, whereas the iPOM C and N appeared to be a relatively constant fraction of the total OC and N pool, as was also observed by Jastrow (1996). In Jastrow´s (1996) study about 12 to 17 % of the total amount of OC in soils under cornfield and restored prairie and about 14% in virgin prairie soil were found to be composed of POM. In a similar study conducted by Cambardella and Elliott (1992), it was found that POM C comprised about 39% of the total SOC in Nebraska native sod.

4. Conclusions

Cultivation had lead to a reduction of the proportion of water stable macroaggregates and an increase in the proportion of microaggregates. The effect of cultivation on the amount of macroaggregates was most evident in the > 1 mm size aggregates. The breakdown of larger aggregates up on slaking to smaller aggregates in both soil types demonstrated the hierarchical concept of aggregate formation in this Nitisol. The result also shows that the OM that binds microaggregates in to macroaggregates was found to be the most prominent sources of OC and N lost due to cultivation. However, there was no correlation between SOC and N, and aggregate stability. The POM C and N were much more affected by cultivation than the whole soil and, total aggregate OC and N, being highest from the macroaggregates. Of the different POM fractions the losses of the free LF C and N due to cultivation were more pronounced than the iPOM fraction, making it a more biodegradable organic matter pool. Generally the results indicate that cultivation of the natural forest soil for 26 years resulted in the reduction of organic matter and deterioration of soil structure.

Acknowledgements

Funding for this research was given by the German Research Foundation (DFG).

References

Angers, D.A., 1992. Changes in soil aggregation and organic carbon under corn and alfalfa. Soil Science Society of America Journal 56, 1244–1249.

Balesdent, J. Besnard, E. Arrouays, D., Chenu, C., 1998. The dynamics of carbon in particle size fractions of soil in a forest cultivated sequence. Plant and Soil 201, 49–57.

Beare, M.H., Hendrix, P.F., Coleman, D.C., 1994a. Water stable aggregates and organic matter fractions in conventional and no-tillage soils. Soil Science Society of America Journal 58, 777–786.

Beare, M.H., Cabera, M.L., Hendrix, P.F., Coleman, D.C., 1994b. Aggregate protected and unprotected organic matter pools in conventional and no-tillage soils. Soil Science Society of America Journal 58, 787–795.

Besnard, E., Chenu, C., Balesdent, J., Puget, P., Arrouays, D., 1996. Fate of particulate organic matter in soil aggregates during cultivation. European Journal of Soil Science 47, 495–503.

Brown, S., Lugo, A.E., 1990. Effects of forest clearing and succession on the carbon and nitrogen content of soils in Puerto Rico and US Virgin Islands. Plant and Soil 124, 53–64.

Buschiazzo, D.E., Hevia, G.G., Hepper, E.N., Urioste, A., Bono, A.A., Babinec, F., 2001.

Organic C, N and P in size fractions of virgin and cultivated soils of the semi-arid pampa of Argentina. Journal of Arid Environments 48, 501−508.

Buyanovsky, G., Aslam, M., Wagner, G., 1994. Carbon turnover in soil physical fractions. Soil Science Society of America Journal 58, 1167−1173.

Cambardella, C.A., Elliott, T.E., 1992. Particulate soil organic matter changes across a grassland cultivation sequence. Soil Science Society of America Journal 56, 777−783.

Cambardella, C.A., Elliott, E.T., 1993. Carbon and nitrogen distributions in aggregates from cultivated and grassland soils. Soil Science Society of America Journal 57, 1071−1076.

Cambardella, C.A., Elliott, E.T., 1994. Carbon and nitrogen dynamics of soil organic matter fractions from cultivated grassland soils. Soil Science Society of America Journal 58, 123−130.

Christensen, B.T., 1992. Physical fractionation of soil organic matter in primary particles and density separates. Advances in Soil Science 20, 1−76.

Dalal, R.C., Mayer, R.J., 1987. Long-term trends in fertility of soils under continuous cultivation and cereal cropping in Southern Queensland. VII. Dynamics of nitrogen mineralization potentials and microbial biomass. Australian Journal of Soil Research 25, 461−472.

Dalal, R.C., Bridge, B.J., 1996. Aggregation and organic matter storage in sub-humid and semi-arid soils. In: Carter, M.R. and Stewart, B.A. (eds.), Structure and Organic Matter Storage in Agricultural Soils. Advances in Soil Science. CRC. Lewis Publisher, Boca Raton. pp. 214–263.

Davidson, E.A., Ackerman, I.L., 1993. Changes in soil organic carbon inventories following cultivation of previously un-tilled soils. Biogeochemistry 20, 161−193.

Detwiler, R.P., 1986. Land use change and the global carbon cycle: the role of tropical soils.

Biogeochemistry 2, 67−93.

Dormaar, J.F., 1983. Chemical properties of soil and water stable aggregates after sixty-seven years of cropping to spring wheat. Plant and Soil 75, 51−61.

Elliott, E.T., 1986. Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. Soil Science Society of America Journal 50, 627−633.

Elliott, E.T., Reuss, D.E., Monz, C.A., Palm, C.A., 1991. Organic matter contained in soil aggregates from a tropical chronosequence: correction for sand and light fraction. Agriculture, Ecosystems and Environment 34, 427−441.

FAO (1997) FAO/UNESCO Soil map of the world, revised legend with corrections and updates. World soil resources, Report No. 60, ISRIC, Wageningen, The Netherlands.

Gregorich, E.G., Drury, C.F., Ellert, B.H., Liang, B.C., 1997. Fertilisation effects on physically protected light fraction organic matter. Soil Science Society of America Journal 61, 482−484.

Gupta, V.V.S.R., Germida, J.J., 1988. Distribution of microbial biomass and its activity in different soil aggregate size classes as affected by cultivation. Soil Biology and Biochemistry 20, 777−786.

Janzen, H.H., Campbell, C.A, Brandt, S.A., Lafond, G.P., Townley-Smith, L., 1992. Light fraction organic matter in soils from long-term crop rotations. Soil Science Society of America Journal 56, 1799−1806.

Jastrow, J.D., 1996. Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. Soil Science Society of America Journal 28, 665−676.

Jastrow, J.D., Boutton, T.W., Miller, R.M., 1996. Carbon dynamics of aggregate-associated organic matter estimated by carbon-13 natural abundance. Soil Science Society of America Journal 60, 801−807.

Hamblin, A.P., Greenland, D.J., 1977. Effect of organic constituents and complexed metal ions on aggregate stability of some east Anglian soils. Journal of Soil Science 28, 410–416.

Haynes, R.J., Beare, M.H., 1996. Aggregation and organic matter storage in meso-thermal, humid soils. In: Carter, M.R., Stewart, B.A. (eds.), Structure and Organic Matter Storage in Agricultural Soils. Advances in Soil Science. CRC. Lewis Publisher, Boca Raton. pp. 194–213.

Haynes, R.J., 1999. Labile organic matter fractions and aggregate stability under short-term, grass-based leys. Soil Biology and Biochemistry 31, 1821−1830.

Lehmann, J., Cravo, M.S., Zech, W., 2001. Organic matter stabilisation in a Xanthic Ferralsol of the central Amazon as affected by single trees: chemical characterisation of density, aggregate, and particle size fractions. Geoderma 99, 147−168.

Miller, R.M., Jastrow, J.D., 1990. Hierarchy of root and mycorrhizal fungal interactions with soil aggregation. Soil Biology and Biochemistry 22, 579−584.

Oades, J.M., 1984. Soil organic matter and structural stability: mechanisms and implications for management. Plant and Soil 76, 319−337.

Oades, J.M., Waters, A.G., 1991. Aggregate hierarchy in soils. Australian Journal of Soil Research 29, 815−828.

Puget, P., Chenu, C., Balesdent, J., 1995. Total and young organic matter distributions in aggregates of silty cultivated soils. European Journal of Soil Science 46, 449–459. Australian Journal of Soil Research 39, 464−489.

Six, J., Elliott, E.T., Paustian, K., Doran, J.W., 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. Soil Science Society of America Journal 62, 1367−1377.

Skjemstad, J.O., Dalal, R.C., Barron, P.F., 1986. Spectroscopic investigations of cultivation effects on organic matter of Vertisols. Soil Science Society of America Journal 50, 354–359.

Skjemstad, J.O., Le Feuvre, R.P., Prebble, R.E., 1990. Turnover of soil organic matter under pasture as determined by ^{13}C natural abundance. Australian Journal of Soil research 28, 267–276.

Spaccini, R., Zena, A., Igwe, C.A., Mbagwu, J.S.C., Piccolo, A., 2001. Carbohydrates in water-stable aggregates and particle size fractions of forested and cultivated soils in two contrasting tropical ecosystems. Biogeochemistry 53, 1–22.

Solomon, D., Fritzsche, F., Tekalign, M., Lehmann, J., Zech, W., 2002. Soil organic matter composition in the sub-humid Ethiopian highlands as influenced by deforestation and agricultural management. Soil Science Society of America Journal 66, 68–82.

Tisdall, J.M., Oades, J.M., 1979. Stabilisation of soil aggregates by the root systems of ryegrass. Australian Journal of Soil Research 17, 429–441.

Tisdall, J.M., Oades, J.M., 1982. Organic matter and water stable aggregates in soils. Journal of Soil Science 33, 141–163.

Tisdall, J.M., 1996. Formation of soil aggregates and accumulation of soil organic matter. In: Carter, M.R., Stewart, B.A. (eds.), Structure and Organic Matter Storage in Agricultural Soils. Advances in Soil Science. CRC. Lewis Publisher, Boca Raton. pp. 16–57.

Yakovchenko, V.P., Sikora, L.J., Millner, P.D., 1998. Carbon and nitrogen mineralisation of added particulate and macro organic matter. Soil Biology and Biochemistry 30, 2139–2146.

C CHANGES IN SOIL ORGANIC CARBON, NITROGEN AND SULPHUR STOCKS DUE TO THE CONVERSION OF NATURAL FOREST INTO TREE PLANTATIONS (*PINUS PATULA AND EUCALYPTUS GLOBULUS*) IN THE HIGHLANDS OF ETHIOPIA

Y. Ashagrie[1*], W. Zech[1], G. Guggenberger[1] and D. Teketay[2]

[1]Institute of Soil Science and Soil Geography, University of Bayreuth, D-95440 Bayreuth, Germany
[2]Ethiopian Agricultural Research Organisation, P. O. Box 2003 Addis Ababa, Ethiopia
* Correspondence: yeshanew@hotmail.com, Tel: ++49-921-55-21-46 Fax: ++49-921-55-22-46

Key words: SOC stock, deforestation, reforestation, Ethiopia.

ABSTRACT

Conversion of natural forests into monoculture forest plantations can affect soil fertility and carbon sequestration, thus CO_2 induced climate change phenomena. We compared SOC, N and S stocks (organic layer and mineral soil) under *Podocarpus falcatus* natural forest and monoculture forest plantations (21 yr *Eucalyptus globulus* and *Pinus patula*, and third rotation *Eucalyptus globulus* established 42 yr ago) at Munesa, South Ethiopia. Results indicate that organic layer mass decreased by 43 and 57% under the 21 yr *Eucalyptus* and *Pinus* plantations, respectively. The third rotation *Eucalyptus*, however, had 85% of the natural forest organic layer mass mainly due to the accumulation of large amounts of poorly decomposed leaves after each harvest. The C/N ratio of the organic layer decreased in the order: 21yr *Pinus* (42), third rotation *Eucalyptus* (38), 21 yr *Eucalyptus* (26), natural forest (25). There were significant reductions (25 to 68%) in the organic layer N stocks under the plantations relative to the natural forest, being highest under *Pinus* and lowest under third rotation *Eucalyptus*. Unlike the 21 yr *Eucalyptus* and *Pinus* organic layers which had 50 and 60% lower OC, respectively, the organic layer under third rotation Eucalyptus contained 91% of the OC stock under the natural forest (16.4 t ha^{-1}). However, this does not imply that the establishment of *Eucalyptus* plantations is sustainable as repeated harvesting removes considerable amounts of nutrients. In the mineral soil, to 1 m depth, there was a significant ($P<0.05$) reduction (16 to 20%) after conversion of natural forest into forest

plantations. The N stocks under the 21 yr *Pinus* and third rotation *Eucalyptus* plantations were significantly reduced amounting 27 and 20% respectively, whereas 21 yr *Eucalyptus* had nearly an equivalent amount of N as that of the natural forest probably due to a dense forest floor vegetation, fixing N. The changes in the S stocks under the plantations were not significant. Mineral soils under plantations had similar C/N ratios, with the exception of the 21 yr *Eucalyptus* which had narrower C/N ratio compared to the natural forest and other plantations. With the exception of the 0–20 cm depth, differences in the distribution of SOC, N and S stocks across the profile were non significant among forest types. Generally, when the data from the organic layers and mineral soils were pooled, the results indicate that clearing of the natural forests and replacement by monoculture tree species plantations resulted in a reduction of large amounts of SOC and N.

INTRODUCTION

Global warming caused by higher atmospheric greenhouse gases (GHGs) concentrations has been a major issue world-wide. Several GHGs are responsible for the warming of the earth's atmosphere, but the single most important GHGs is carbon dioxide (CO_2), which accounts for about 60% of the anthropogenic effect (BMZ, 2002). According to the report of the Intergovernmental Panel on Climate Change (IPCC) (1996a) the concentrations and distributions of naturally occurring gases have been greatly affected by human activities. The main emission sources are the energy and transport sectors, but deforestation of natural forests affects sources and sinks of greenhouse gases through emissions resulting not only from destruction and burning of the vegetation but also from more and more soil organic matter (SOM) oxidation (Tate et al., 1993). Nowadays questions such as the sources and sinks of GHGs, and the role of the soil carbon and nitrogen cycle in the global climate change have been drawing more and more attention (Johnson, 1992; Johnson and Curtis, 2001; Liping and Erda, 2001). In the tropics where soils are poor in inorganic nutrients, concern on soil carbon levels are not only to decrease the effect of CO_2 emissions on global warming but also to ensure nutrient availability as the soils rely on recycling of nutrients from soil organic matter to maintain productivity (Tiessen et al., 1994; IPCC, 1996b). Nitrogen is of particular concern, since availability of this element limits forest growth more frequently than any other nutrient (Tiessen et al., 1994).

The world's forests store large quantities of carbon, with an estimated 330 Gt C in live and dead above- and below ground vegetation, and 660 Gt C in soil (mineral soil plus organic horizons) (IPCC, 1996b). Tropical forests store 46% of the world's living terrestrial carbon pool and 11% of the world's soil carbon pool (Brown and Lugo, 1982). About 90–95% of the N in forest soils is contained in soil organic matter (SOM) (Fisher and Binkley, 2000). Carbon storage in forest soils is a dynamic balance between inputs (primarily litter and dead roots) and outputs (primarily heterotroph respiration) (Simmons et al., 1996). In mature forests, C and nutrient storage in soil approaches a steady-state where outputs nearly equal the inputs. Land use change is often associated with changes in litter quantity or rate of decomposition and an associated change in SOC and nutrient levels (Johnson, 1995; Van Cleve and Powers, 1995). The degree of SOC and nutrient losses/gains after plantation forest establishment depends on previous land use, time, forest species or forest type, management and local climate and soil conditions (Johnson, 1992). For example, harvesting followed by cultivation or intensive site preparation for planting trees may result in large decreases in soil carbon—up to 30% to 50% in the tropics over a period of up to several decades (Fearnside and Barbosa, 1998). Harvesting followed by reforestation, however, in most cases has a limited effect (±10 percent) (Allen, 1985; Turner and Lambert, 2000; Zinn et al., 2002). This effect is particularly prevalent in the tropics (Johnson, 1992). There are also some cases in which soil carbon increases significantly, probably because of the additions of slash and its decomposition and incorporation into the mineral soil (Detwiler, 1986;

Johnson, 1992).

In the highlands of Ethiopia, increases in human and livestock population and the sedentary nature of agriculture have resulted in massive deforestation of natural forests. Consequently, loss of biodiversity, land degradation and shortage of wood and wood products are the major phenomenon. To satisfy the demand for wood and wood products of the population and to rehabilitate degraded lands, large scale forest plantations on degraded lands and sites supporting low productive secondary natural forests have been carried out in 1980`s (Pohjonen, 1989; EFAP, 1994). Most of the plantations are monoculture exotic species such as *Eucalyptus*, *Cupressus* and *Pinus* covering 200,000 ha (EFAP, 1993). The establishment of plantations on degraded lands not only helps to rehabilitate sites and provide wood resources but also to enhance the amount of OC and nutrients in the soil and to mitigate CO_2 emission effects on climate change (Brown and Lugo, 1982). According to Lugo et al. (1988), the net primary productivity of some plantations could be higher than secondary and mature forests, and some plantation species could also accelerate SOC recovery (Lugo et al., 1990a,b). A major concern with tree plantations in the tropics is when they replace natural forests, and the sustainability of multiple rotations of high yielding species because they may place a high nutrient demand on the soil (Cuevas et al., 1991). The effect of tree plantations on SOC and nutrient levels in the highlands of Ethiopia is not well known. Studies that have been undertaken so far (Michelsen et al., 1993; Solomon et al., 2001, 2002) dealt only with the surface soil layers. Therefore, understanding the changes to soil biogeochemistry that influence SOC and nutrient levels to deeper soil layers after clearing and replacement of natural forests with monospecfic tree plantations will be important for implementing effective and sustainable management plans and for predicting the regional/global and ecological consequences of plantations.

MATERIALS AND METHODS

- **The study site**

The study was conducted at Munesa within the Munesa/Shashemenie forest enterprise area about 240 km south east of Addis Ababa at an altitude of 2400 m. Rainfall is bimodal with a mean annual precipitation of 1250 mm most of it falling in July and August, and mean annual temperature is 19°C with little seasonal variation. The soils are classified as Nitisol (FAO, 1997) and characterised by amorphous hydrous aluminium and iron oxides mineralogy. A mixed natural forest and three plantations established on, and adjacent to, it were selected for this study. The plantations were established after clearing of part of the natural forest. Clearing was done manually and the surface biomass was burned on site. The different plantation compartments are situated side by side to each other and to the natural forest with similar history, only separated by forest roads. Selected chemical and physical properties of the natural forest soil from which all plantations were established are presented in Table C1.

Table C1. Selected chemical and physical properties under the natural forest.

Horizons	Depth (cm)	pH_{KCl}	CEC $cmol_c\ kg^{-1}$ soil (1M NH_4 OAc)	Al_d	Fe_d	Al_0	Fe_0	sand	silt	clay
				g kg^{-1} soil						
A	0–15	5.4	51.1	3.7	42.9	3.4	9.1	200	300	500
AB	15–29	5.3	32.8	4.6	55.3	4.5	7.6	230	230	540
Bt1	29–68	4.7	30.6	4.7	58.5	4.5	7.2	80	180	740
Bt2	68–108+	4.5	29.2	4.1	57.3	3.9	6.2	80	180	740

The plantation forests were (1) *Eucalyptus globulus* established in 1960 and harvested two times, (2) *Eucalyptus globulus* established in 1980 and never harvested and (3) *Pinus patula* established in 1980 and never harvested. The 21 years *Eucalyptus* stand was more open to light penetration and due to the rich understory shrub and herbaceous vegetation it is also used for grazing purposes. Due to the dense forest floor layer the third rotation *Eucalyptus* stand lacked understory vegetation, while the *Pinus* stand had sparse understory vegetation.

- **Sampling**

In each forest type, three 0.06 ha plots were located and a pit was excavated to the depth of 1.2 m at the centre of each plot. Soil samples for chemical analysis were taken from the three sides of the pit at 0–20, 20–40, 40–70 and 70–100 cm depths. In addition, two 1 m^2 plots were marked randomly within each plot and samples were taken by auger at three points within the 1 m^2 area and mixed for the above mentioned depth classes. Soil samples were put in individual polyethylene bags, air-dried and passed through a 2-mm sieve before grinding for analysis. Litter samples were taken within a 1 m^2 square area at three points by pressing a 0.09 m^2 steel sheet sampling frame into the organic layer. The surrounding organic matter was removed leaving a block of the organic layer in which the litter (L) and fermentation (F) horizons were identified. Each horizon was measured for thickness and the organic material was placed in a separate paper bag. Afterwards the samples from the three square subplots were mixed and the number of samples was reduced to three. Samples for the mineral soil bulk density determination were taken by 100 cm^3 core at seven points for each soil depth. Both the litter and soil samples were transported to Bayreuth University, Germany, for chemical analysis.

- **Laboratory analysis**

The litter samples were dried at 65°C and weighed. Mass per unit area of the organic layer was based on the area of the sampling frame and the oven dry weight of the sample. Mineral soil bulk density was calculated after drying of the sample at 105°C to a constant mass and dividing the oven-dry mass by the volume of the core segment. Carbon, nitrogen and sulphur were determined using a CHNS−analyser (Vario EL, Elementar Analysensysteme). The pH_{KCl} (soil:solution ratio 1:2.5) for the mineral soil and pH_{H2O} (1: 20) for the organic layer were determined with a standard pH electrode (Orion U402−S7). Cation exchange capacity (CEC) was determined with 1 M NH_4OAc (pH=7.0) following the procedure of Hendershot et al., (1993). Dithionite−citrate−bicarbonate extractable aluminium and iron (Al_d, Fe_d) and oxalate-extractable aluminium and iron (Al_o, Fe_o) were determined according to Ross and Wang (1993).

- **Data analysis**

Carbon, N and S stocks (kg m^{-2}) to a given depth were calculated as the product of bulk density, concentration, and depth of sampling. Data on measured mean soil properties of the different forest types were compared by using MSTAT−C version 2.10 statistical package. Differences between and among treatment means were considered significant at $P \leq 0.05$.

RESULTS AND DISCUSSION

- **Organic layer**

The thickness of the organic layer was not significantly affected by forest type and forest type by horizon interaction. The thickness of the L, Of1 and Of2 horizons respectively were 0.8, 1.2 and 1.7 cm (total 3.7 cm) under the natural forest, 1.3, 1.0 and 1.7 cm (total 4 cm) under third rotation *Eucalyptus*, 0.5, 1.0 and 1.3 cm (total 2.8 cm) under 21 yr *Eucalyptus* and 0.5, 1.1 and 1.3 cm (total 3.5 cm) under *Pinus*. In a similar study, Wilcke et al. (2002) reported a thicker organic layer (15.7 cm) in the natural forest of Ecuador relative to our study forests. This difference could be attributable to differences in disturbance, vegetation composition and age, local climate, microtopography and soil type and mineralogy (Lugo et al., 1986). The rganic layer bulk density in the four forest types varied significantly ($P < 0.01$), increasing in the order *Pinus*<21 years

Eucalyptus<third rotation *Eucalyptus*<natural forest (Table C2). In each forest type, with few exceptions thickness and bulk density of the organic layer horizons decreased in the order Of2 > Of1 > L (Table C2). There was a significant effect of forest type on the organic layer mean C ($P < 0.1$), N and S ($P < 0.01$) concentrations, but forest type by horizon interaction was not significant. The least significant difference test (LSD) revealed that C concentration was significantly greater under third rotation *Eucalyptus* compared to the natural forest and 21 years *Eucalyptus*, but differences between *Pinus* and the other forests were not significant (Table C2). N and S concentrations were higher under 21 years *Eucalyptus* than the other two plantations but were not different from the natural forest. In each forest type, C concentration decreased while N and S concentrations increased from the L horizon to the Of2 horizon (Table C2). Similar results have been reported by Lugo et al. (1990b). The C/N ratios under *Pinus* and third rotation *Eucalyptus* organic layers (Table C2) were significantly ($P<0.01$) wider than under the natural forest, but 21 years *Eucalyptus* had an equivalent C/N ratio as that of the natural forest probably due to the presence of N-fixing plants in the understorey vegetation.

In each forest type, as expected, C/N ratio dropped considerably from the L layer to the Of2 layer indicating that litter in the Of layer is in advanced state of decomposition. The *Pinus* plantation L horizon had a C/N ratio of twice that of the natural forest (Table C2). In view of the current increase in atmospheric CO_2 level litter with high C/N ratio could serve as a temporary sink of CO_2 because it positively affects SOC accumulation (Swift et al., 1979). From the ecological point of view, however, litter with high nutrient content or low C/N ratio play an important role in plantations because rather than immobilising nutrients it releases for rapid recycling (Lugo et al., 1990b). Mean pH values in the organic layers of the different forest types ranged from 5.7–7

(Table C2). The L layer pH varied in the order *Pinus* (3.8)<*Eucalyptus* (5.0)<natural forest (7.0). Unlike the results of Wilcke et al. (2002), who reported a decreasing trend in the organic layer pH (Oi > Oe > Oa), in all our study forest types, pH tended to increase from the L layer to the Of layer and this difference was more evident under *Pinus* than the other forest types.

Table C2 Chemical and physical properties of the organic layer horizons as influenced by conversion of the natural forest into tree plantations.

	Natural forest				**3rd rot. *Eucalyptus***				**21 yr *Eucalyptus***				**21 yr *Pinus***			
	L	Of1	Of2	Mean	L	Of1	Of2	Mean	L	Of1	Of2	Mean	L	Of1	Of2	Mean
Bulk density (g cm^{-3})	0.11 (0.03)	0.12 (0.02)	0.12 (0.02)	0.12**a** (0.02)	0.03 (0.01)	0.07 (0.02)	0.16 (0.06)	0.09**ab** (0.03)	0.06 (0.02)	0.07 (0.02)	0.10 (0.04)	0.08**ab** (0.03)	0.05 (0.02)	0.03 (0.01)	0.09 (0.02)	0.06**b** (0.02)
C (g kg^{-1})	444 (16)	382 (22)	343 (64)	390**b** (34)	517 (9.1)	438 (4.8)	368 (74)	441**a** (29)	481 (7.4)	358 (11)	322 (20)	387**b** (13)	493 (26)	449 (47)	326 (38)	423**ab** (37)
N (g kg^{-1})	13 (1.1)	17.3 (0.5)	18 (2.2)	16**ab** (1.3)	8.7 (1.2)	16 (3.1)	14.6 (1.7)	13**ab** (2)	115 (3.8)	21.3 (1.6)	18.3 (1.4)	17**a** (2.3)	7.7 (2.8)	12.9 (1.6)	14.7 (0.2)	12**b** (1.5)
S (g kg^{-1})	1.4 (0.1)	1.8 (0.2)	1.8 (0.1)	1.67**a** (0.13)	1.1 (0.1)	1.6 (0.3)	1.5 (0.2)	1.4**b** (0.2)	1.3 (0.4)	1.8 (0.3)	2.0 (0.1)	1.7**a** (0.3)	1.0 (0.2)	1.3 (0.2)	1.6 (0.2)	1.3**b** (0.2)
C/N	34 (3.8)	22 (1.1)	18 (1.2)	25**c** (2.0)	60 (9.2)	28 (5.9)	26 (6.8)	38**b** (7.3)	45 (15.4)	17 (1.1)	15 (1.1)	26**c** (5.4)	70 (26)	35 (7)	22 (2.3)	42**a** (12)
pH	6.4 (0.7)	7.2 (0.5)	7.4 (0.2)	7.0**a** (0.5)	5.0 (0.5)	6.8 (0.1)	6.7 (0.2)	6.2**ab** (0.3)	5.0 (0.5)	6.9 (0.2)	6.9 (0.1)	6.3**a** (0.3)	3.8 (0.5)	6.1 (0.24)	6.0 (0.22)	5.3**b** (0.3)

- Means followed by the same lower case letter in a row are not significantly different from each other at 0.01 probability level. Numbers in parentheses are standard deviations (n=3).

Table C3 Dry mass and C, N and S stocks (t ha^{-1}) in the organic layers of the different forest types.

	Dry mass				C				N				S			
Forest type	L	Of1	Of2	Total	L	Of1	Of2	Total	L	Of1	Of2	Total	L	Of1	Of2	Total
Natural forest	9.1	13.2	20.2	42.5**a**	4.1	5.4	6.9	16.4**a**	0.12	0.24	0.36	0.72**a**	0.01	0.03	0.04	0.08**a**
	(4.1)	(6.6)	(6.9)	(5.9)	(2.0)	(2.4)	(2.6)	(7)	(0.1)	(0.1)	(0.1)	(0.3)	(0.01)	(0.01)	(0.02)	(0.04)
3rd rot. *Eucalyptus*	3.8	7.7	24.6	36.1**ab**	2.1	3.1	9.8	15**a**	0.03	0.11	0.39	0.53**ab**	0.004	0.01	0.04	0.05**ab**
	(1.7)	(1.3)	(4.1)	(2.4)	(1.0)	(0.7)	(3.6)	(5.3)	(0.02)	(0.04)	(0.5)	(0.56)	(0.002)	(0.004)	(0.004)	(0.01)
***Eucalyptus* (21 yrs)**	2.7	7.4	14.1	24.2**bc**	1.4	2.5	4.3	8.2**b**	0.03	0.15	0.24	0.42**bc**	0.004	0.01	0.03	0.04**ab**
	(0.5)	(1.7)	(8.9)	(3.7)	(0.5)	(0.7)	(2.6)	(3.8)	(0.01)	(0.04)	(0.2)	(0.25)	(0.001)	(0.002)	(0.02)	(0.02)
***Pinus* (21 yrs)**	2.6	3.6	12.1	18.3**c**	1.2	1.5	3.8	6.5**b**	0.02	0.04	0.17	0.23**c**	0.003	0.004	0.02	0.03**b**
	(0.12)	(0.42)	(4.3)	(1.6)	(0.1)	(0.3)	(1.3)	(1.7)	(0.01)	(0.002)	(0.1)	(0.11)	(0.001)	(0.00)	(0.01)	(0.01)

- Means followed by the same lower case letter in a row are not significantly different from each other at 0.01 probability level. Numbers in parentheses are standard deviations (n=3).

Table C4 SOC, N and S contents (g kg^{-1}) and bulk density (g cm^{-3}) at different depths under the different forest types.

Soil depth	NF	Eur	Eu	Pi	NF	Eur	Eu	Pi	NF	Eur	Eu	Pi	NF	Eur	Eu	Pi
		C					**N**			**S**			**Bulk density**			
0–20	66.1	49.3	59.3	46.2	5.7	3.5	6.3	3.2	0.7	0.6	0.7	0.4	0.86	0.90	0.91	0.90
	(12)	(5.7)	(7.8)	(8.6)	(0.9)	(0.3)	(1.0)	(0.6)	(0.1)	(0.04)	(0.1)	(0.1)	(0.05)	0.04)	(0.04)	(0.07)
20–40	34.1	35.3	27.7	32.6	3.0	2.7	3.1	2.4	0.4	0.5	0.4	0.4	0.97	0.97	0.98	0.97
	(7.9)	(6.8)	(10)	(7.5)	(0.8)	(0.4)	(1.1)	(0.4)	(0.0)	(0.04)	(0.1)	(0.04)	(0.03)	(0.01)	(0.02)	80.02)
40–70	29.5	20.1	17.0	21.5	2.6	1.8	1.9	1.8	0.4	0.3	0.2	0.4	1.08	1.07	1.11	1.09
	(11)	(6.2)	(4.6)	(7.6)	(0.7)	(0.4)	(0.7)	(0.5)	(0.1)	(0.04)	(0.1)	(0.1)	(0.04)	(0.07)	(0.01)	(0.05)
70–100	15.4	15.7	12.9	15.0	1.6	1.8	1.5	1.5	0.2	0.3	0.2	0.2	1.11	1.12	1.12	1.13
	(5.1)	(8.6)	(4.0)	(7.6)	(0.4)	(0.5)	(0.4)	(0.3)	(0.1)	(0.1)	(0.1)	(0.1)	(0.04)	(0.06)	(0.01)	(0.03)
Mean	36.3**a**	30.1**ab**	29.2**b**	28.9**b**	3.2**a**	2.4**b**	3.2**a**	2.2**b**	0.4	0.4	0.4	0.4	1.01	1.02	1.03	1.02

Means followed by the same lower case letters in a row are not significantly different from each other at 0.01 probability level. Numbers in parentheses are standard deviations (n =5 for C, N and S and n =7 for bulk density). NF-natural forest, Eur-third rotation *Eucalyptus*, Eu-21 years *Eucalyptus*, Pi-21 years *pinus*.

Table C5 Depth wise storage of OC, N and S stocks in kg m^{-2} , 1 m soil depth. Numbers in parenthesis are standard deviations (n=5).

	Soil depth (cm)					
Land use	0–20	20–40	40–70	70–100	0–100	20–100
		OC				
Natural forest	11.37 (2.13)	6.62 (1.54)	9.56 (3.46)	5.13 (1.68)	32.68**a**	21.31
3rd rot. *Eucalyptus*	8.87 (1.02)	6.85 (1.33)	6.45 (1.98)	5.28 (2.90)	27.45**b**	18.58
Eucalyptus (21 years)	10.79 (1.43)	5.43 (2.03)	5.66 (1.52)	4.33 (1.35)	26.21**b**	15.42
Pinus (21 years)	8.32 (1.62)	6.32 (1.38)	7.03 (2.56)	5.08 (2.46)	26.75**b**	18.43
		Total N				
Natural forest	0.98 (0.15)	0.58 (0.16)	0.84 (0.22)	0.53 (0.14)	2.93**a**	1.95
3rd rot. *Eucalyptus*	0.63 (0.05)	0.53 (0.07)	0.58 (0.13)	0.60 (0.17)	2.34**b**	1.71
Eucalyptus (21 years)	1.15 (0.17)	0.61 (0.21)	0.63 (0.24)	0.50 (0.12)	2.89**a**	1.74
Pinus (21 years)	0.58 (0.1)	0.47 (0.67)	0.59 (0.16)	0.51 (0.11)	2.15**b**	1.57
		Total S				
Natural forest	0.12 (0.02)	0.08 (0.00)	0.13 (0.03)	0.07 (0.03)	0.4	0.28
3rd rot. *Eucalyptus*	0.11 (0.01)	0.11 (0.01)	0.10 (0.01)	0.10 (0.03)	0.42	0.30
Eucalyptus (21 years)	0.13 (0.02)	0.08 (0.02)	0.07 (0.03)	0.07 (0.03)	0.35	0.22
Pinus (21 years)	0.07 (0.01)	0.08 (0.01)	0.13 (0.03)	0.07 (0.02)	0.35	0.28

- Means followed by the same lower case letter in a column are not significantly different from each other at 0.05 and 0.01 probability level for OC and N, respectively.

The average organic layer mass by forest type varied from 18.3–42.5 t ha^{-1} with a site average of 30.28 t ha^{-1}(Table C3). These values were much greater compared to the organic layer mass (10.5

t ha^{-1}) under 11 years old *Pinus* plantation and (5 t ha^{-1}) under secondary natural forest in Puerto Rico (Cuevas et al., 1991) and the Brazilian Cerrado vegetation dominated with broad leaved shrubs and trees (6–13. 5 t ha^{-1}), and *Pinus* (37.69 t ha^{-1}) and *Eucalyptus* (7.62−13.9 t ha^{-1}) plantations (Zinn et al., 2002), but were lower than the organic layer mass estimates of Wilcke et al. (2002) for the tropical montane rain forest (247 t ha^{-1}) in Ecuador. The forest floor mass estimates of Vogt et al. (1986) for the tropics and sub tropics ranged from 3.2−5.4 t ha^{-1} for tropical evergreen, 2.08−16.48 t ha^{-1} for tropical broad-leaved deciduous and 1.34−13.12 t ha^{-1} for subtropical broad-leaved deciduous. Turner (1986) reported organic layer mass of 13–21 t ha^{-1} under 5–31 years old *E. grandis* forest. Lugo et al. (1990b) reported 5–27.2 t ha^{-1} organic layer mass under 26 years old different tree plantations.

Organic layer mass was significantly influenced by forest type ($P<0.01$), being highest under the natural forest followed by third rotation *Eucalyptus* and lowest under *Pinus* (Table C3). However, the interaction of forest type by horizon was non significant. In each forest type, litter accumulation was greatest in Of2 horizon and lowest in L horizon (Table C3). The reductions in average litter mass after clearing and replacement of the natural forest ranged from a low of 6.4 t ha^{-1} (–14%) under third rotation *Eucalyptus* to a maximum of 24.2 t ha^{-1} (–57%) under *Pinus* (Table C3). Such variations in the organic layer mass accumulation may be due partly to differences in rate of litter production, litter quality, age and species composition and, partly to increased microbial activity in the post clearing period caused by more favourable moisture and temperature conditions and burning of the organic layer and above ground biomass of the previous forest during site preparation. Zinn et al (2002) reported an increase in litter mass after conversion of native *Cerrado* to *Pinus* and a decrease after conversion to *Eucalyptus* in loamy Oxisols in Brazil. On the sandy Entisol, however, they reported a non significant difference in litter quantity between the native *Cerrado* and *Eucalyptus* plantation. The greatest mass under third rotation *Eucalyptus* compared to the other two plantations is due to the accumulation of litter after each harvest and differences in time since establishment. This demonstrates that residue management after harvesting can have a large effect on organic layer mass. Cuevas et al. (1991) and Grigal and Ohmann (1992) found differences in forest floor mass due to forest type and age. A poorer quality litter (high in lignin/nitrogen and C/N ratios) input with lower decomposition rates could also contribute to a greater forest floor mass accumulation (Vogt et al., 1986).

Our best estimates of the organic layer mean C, N and S stocks for the different forest types ranged from 6.5–16.4 t ha^{-1}, 0.3–0.7 t ha^{-1} and 0.03–0.1 t ha^{-1}, respectively (Table C3). Wilcke et al. (2002) reported 103, 5.53 and 0.77 t ha^{-1}C, N and S stocks, respectively, in the tropical montane rainforest of Ecuador. The estimates of Tiessen et al. (1994) for the natural rainforest organic layer of Venezuela were 69.95 t ha^{-1}C and 3.17 t ha^{-1}N. Values reported by Lilienfein et al. (2001) in Brazil range from 1.2–24.5 t ha^{-1}C, 0.02–2.6 t ha^{-1}N and 2.3–26 kg ha^{-1}S. Smith et al. (2002) in Brazil estimated 3.59–5.42 t ha^{-1}C for organic layers under natural forest, *P. caribaea*, *C.guianensis* and *E.paraensis* stands. Turner and Kelly (1985) reported C stocks of 27 t ha^{-1} under *Eucalyptus* and 21 t ha^{-1} under *Pinus* stands. The forest floor data compiled by Vogt et al. (1986) indicate average values of 7 t C ha^{-1} for tropical forest, 14 t C ha^{-1} for temperate forest and 17 t C $ha^{-1}$for boreal forest, and the data of Vogt et al. (1995) yield averages of 5 t C ha^{-1} for tropical forest, 17 t C ha^{-1} for temperate forest and 33 t C ha^{-1} for boreal forest. Vogt et al. (1986) reported forest floor N content of 0.12 t ha^{-1} under subtropical broadleaf evergreen forest, 0.04 t ha^{-1} under tropical broadleaf semi-deciduous forest and 0.33 t ha^{-1}under tropical broadleaf evergreen forest. The N values reported under 26 years old different tree plantations organic layer by Lugo et al. (1990b) ranged from 0.06–0.19 t ha^{-1}. Variations in C, N and S stocks between our study and others may be due to differences in site characteristics such as geomorphology, disturbance history, climate, soil texture, slope, and composition and age of vegetation (Homann et al., 1995).

Higher litter mass accumulation in the natural forest and third rotation *Eucalyptus* resulted in a significantly ($P<0.01$) higher proportion of C and nutrient storage. The natural forest had always higher C, N and S stocks compared to 21 years *Eucalypus* and *Pinus* stands. C stock under the natural forest (16.4 t ha^{-1}) was found to be significantly reduced by 8.2 t ha^{-1} (–50%) and 9.9 t ha^{-1} (–60%) after conversion to 21 years *Eucalyptus* and *Pinus* plantations, respectively, while third rotation Eucalyptus had an equivalent amount of C (15 t ha^{-1}) as that of the natural forest probably due to the greater amount of litter accumulated after each harvest. Like that of C, the reductions in N and S stocks under *Pinus* were much higher followed by 21 years *Eucalyptus* (Table C3). These could be attributed to differences in net primary productivity, chemical composition/susceptibility of the litter to decomposition and age of the forest (Minderman, 1968; Swift et al., 1979; Lugo et al., 1988; Cuevas et al., 1991; Mtambanengwe and Kirchmann, 1995). The two *Eucalyptus* stands did not significantly differ in their N and S stocks, but C was higher under third rotation *Eucalyptus* suggesting that C storage in the soil increases with time. In a similar study, Grigal and Ohmann (1992) indicated an increase in the organic layer C stock with time. Minimisation of site disturbance during harvest operations and retention of forest litter and debris after silvicultural activities can help retain organic matter (Johnson, 1992). C and N stocks under *Pinus* were significantly lower than under third rotation *Eucalyptus*, but differences between *Pinus* and 21 years *Eucalyptus* were not significant.

- **Mineral soil**

Unlike the organic layer, there was no considerable variation in the overall mean bulk (Table C4) of the mineral soil used to calculate SOC and nutrient stocks among forest types. SOC concentration under the natural forest was significantly higher than under the 21 years *Eucalyptus* and *Pinus* stands. The natural forest and 21 years *Eucalyptus* had greater N concentration compared to third rotation *Eucalyptus* and *Pinus* stands, but there were no considerable differences between the former and the latter two. The interaction of forest type by soil depth was also significant for OC ($P<0.05$), and N and S ($P<0.01$) concentrations. In the surface 20 cm soil layer, the natural forest and 21 years *Eucalyptus* had higher SOC, N and S concentrations compared to third rotation *Eucalyptus* and *Pinus* stands, but differences between the former two were not significant (Table C4). SOC (4.62−6.61%) and N (0.32−0.63%) concentrations in the top 20 cm soil of our study site were higher than those reported by others in the tropics (Brown and Lugo, 1990; Lilienfein et al., 2001; Zinn et al., 2002). Below the 20 cm soil depth, except OC and N in the 20–40 cm layer, all forest types had nearly the same OC, N and S concentrations (Table 4). Mean C/N ratio to the depth of 1 m under 21 years *Eucalyptus* (9) was significantly ($P<0.01$) lower than the C/N ratios under third rotation *Eucalyptus* (11), and under the natural forest and *Pinus* (12). In all forest types, with the exception of 21 years *Eucalyptus*, C/N ratio tended to decrease with increasing depth (Fig. C1) probably due to leaching of N rich materials from the upper soil layers. Lilienfein et al. (2001) reported average C/N ratios of 12.9 under *pinus* plantation and 15.6 under *Cerrado* vegetation, with a decreasing trend under *Pinus* from the surface to the subsoil. Lugo et al. (1986) reported C/N values ranging from 10.3–13.5 for the Puerto Rican forest soils. At each depth, 21 years *Eucalyptus* had consistently lower C/N ratio compared to the natural forest and the other plantations.

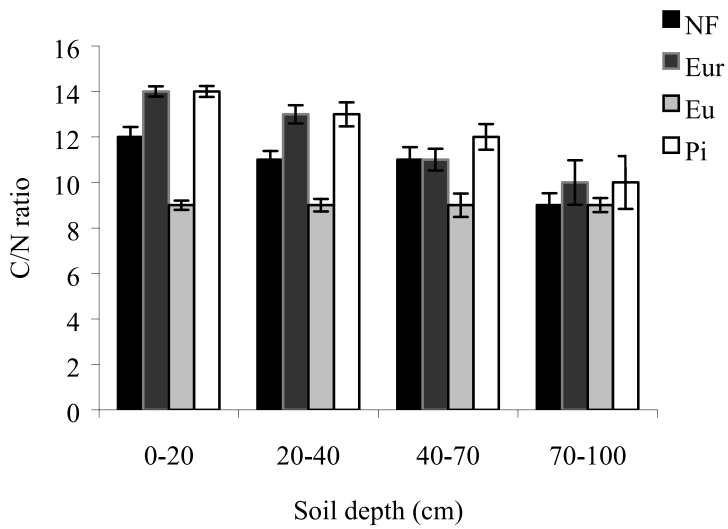


Fig. C1. C/N ratio at different soil depths as influenced by conversion of natural forest into monoculture tree plantations. NF-natural forest, Eur-third rotation *Eucalyptus*, Eu-21 years *Eucalyptus* and Pi-21 years *Pinus*. Vertical lines are standard errors (n=5).

Average mineral soil OC stock in this study ranging from 26.2–32.7 kg m^{-2} to 1 m depth (Table C5) was higher than in several other studies in the tropics (Brown and Lugo, 1982; Post et al., 1982; Post et al., 1985; Lugo et al., 1986; Brown and Lugo, 1990; Moraes et al., 1995; Lardy et al., 2002; Zinn et al., 2002). The world average for all soils is 11.7 kg m^{-2} to 1 m depth, based on the data of Eswaran et al. (1993). This difference could be due to soil forming factors, including climate, parent material, topography, vegetation, and disturbance. From the results of this study we could deduce that protected natural forests and plantations established even in the absence of climate change considerations in Ethiopia can store large amount of SOC and contribute to the mitigation of CO_2 induced climate change and global warming. Activities aimed at protection of nature, biodiversity and natural resources degradation such as soil have a major co-benefit that they contribute to C sequestration and conservation (IPCC, 1996b; Brown, 1999).

Forest type had a significant ($P<0.05$) effect on mean OC stock to 1 m mineral soil depth. SOC under the different plantations varied from 26.2–27.5 kg m^{-2} representing 80–84% of the SOC stock under the natural forest (32.7 kg m^{-2}) (Table C5). Since there is about three times as much C in the world's soils as in the atmosphere (Follett, 2001), the observed changes (–16 to –20%) in this pool can have considerable feed-back effects on the amount of CO_2 in the atmosphere and thereby on global warming. In a similar study in Brazil, Zinn et al. (2002) found OC losses of 9% and 17% to the depth of 60 cm after conversion of *Cerrado* vegetation to *Pinus* and *Eucalyptus*, respectively, in the sandy Entisol, and no net losses in Oxisol under Eucalyptus. Turner and Lambert (2000) reported reductions of about 65 and 180 t ha^{-1} OC to the depth of 50 cm under 24 years *Pinus radiata* and 20 years *Eucalyptus grandis* relative to native vegetation and pasture, respectively. While Lugo et al. (1986) found a comparable SOC level to that in mature forest stands after 30 years of forest succession.

Mean N stock to 1 m soil depth (Table C5) differed with forest type and the interaction was also significant ($P<0.01$). The natural forest and 21 years *Eucalyptus* plantation had significantly higher amount of N stock compared to third rotation *Eucalyptus* and *Pinus* stands, but both the former and the later two were not different from each other. The reductions in N stocks relative to the natural forest varied from a maximum of 0.78 kg m^{-2} (–27%) under *Pinus* to a low of 0.04 kg m^{-2} (–1.4%) under third rotation *Eucalyptus* (Table C5). The changes in S stocks due to the transformation of the natural forest into different forest plantations were non-significant, being 0.05 kg m^{-2} (–13%) under *Pinus* and 21 years *Eucalyptus*, while there was a net gain of 0.02 kg m^{-2} (+5 %) under third rotation *Eucalyptus* (Table C5). The observed losses in OC to 1 m depth mineral soil of 6.47 kg m^{-2} (–20%) under 21 years *Eucalyptus* and 5. 94 kg m^{-2} (–18%) under *Pinus* were almost 2 times less than that observed in the organic layer, while under third rotation *Eucalyptus* the opposite holds true. The reasons for the relatively small reduction in mineral soil OC compared to the organic layer could be: (1) even though harvest removes much of the biomass, inputs from roots to the mineral soil wood have been large, (2) soils with high clay and Al and Fe content stabilise organic matter and retard decomposition by complexing with Al and Fe (Oades, 1988; Veldkamp, 1994) and (3) the organic layer is most affected by management such as burning and residue management compared to the mineral soil.

The distribution of SOC, N and S stocks across the profile (Table C5) tended to follow the general trend in SOC, N and S concentrations (Table C4), decreasing from the surface to the subsoil although bulk density values increased in the opposite direction. An exception is the 40–70 cm depth, where SOC, N and S stocks were relatively elevated compared to the overlying soil layer in most of the forest types due to the high bulk density and clay content values. Nearly one-third of the total SOC, N and S stocks to 1 m depth in all forest types (Table C5) were found in the surface 0–20 cm layer. This points out the need for proper management as it represents the pool most exposed to management effects that may accelerate its decomposition and release of CO_2 to the atmosphere. Compared to the results of other studies in the tropics (Lugo et al., 1986; Lugo et al., 1990a; Tiessen et al., 1994; Bashkin and Binkley, 1998; Johnson et al., 2001; Zinn et al., 2002), SOC and N stocks to 20 cm depth in our study site were relatively high, but were comparable to the results of Brown and Lugo (1990b) and Smith et al. (2002). Based on the LSD test, surface soil OC, N and S stocks under the natural forest (Table C5) were not significantly different from the 21 years *Eucalyptus*. This could be attributable to the presence of dense understory N-fixer vegetation that contributed to surface SOC and N and the recycling of nutrients via animal excreta as the 21 years *Eucalyptus* is used for grazing purposes. However, both 21 years *Eucalyptus* and natural forest stored greater amounts of OC, N and S compared to *Pinus* and third rotation *Eucalyptus*. In a study by Lugo et al. (1990b), tree plantations with native species in the understorey vegetation were found to have higher litter nutrient concentrations compared to those which have only canopy vegetation. A reduction in surface (0–

10 cm) soil OC after plantation establishment on previously undisturbed native vegetation was reported by Turner and Lambert (2000).

The significant reduction in surface soil OC and the non-significant difference in subsoil OC between the plantations and the natural forest may be due to the effect of burning during site preparation and cultivation during planting, and to reduced inputs in the first few years of stand establishment and accelerated mineralisation of organic matter because of changes in microclimate following clearing and afforestation. Following clearing, soil C inputs dropped substantially in the first few years due to a small forest biomass and low rate of litter fall, thus C loss from this portion of the profile continues outweighing gains in C from litter production (Paul et al., 2002). Below 20 cm, treatment effects on SOC, N and S stocks were less clear although the natural forest tended to have large SOC and N stocks compared to all plantations in the 40–70 cm layer (Table C5). Detwiler (1986) reported that most studies found no effect of land use change on soil OC below 40 cm in the tropics. Bashkin and Binkley (1998) found no significant change in SOC below 55 cm after planting of *Eucalyptus* on previous cane site in Hawaii. When the data below 20 cm were combined, more than 60% of the total SOC, N and S stocks in the surface 1 m depth under all forest types were stored below 20 cm depth and the losses from this part of the profile although not significant ranged from 2.73–5.89 kg m^{-2} (13–28%) for SOC, from 0.21–0.4 kg m^{-2} (11–20%) for N and 0.06 kg m^{-2} (21%) for S. This indicates that sampling to 20 cm depth excludes large proportions of the total SOC and nutrient stocks to 1 m depth which are very important in the global carbon and nitrogen biogeochemical cycles because they are less susceptible to be oxidised and transported to the atmosphere (Brown and Lugo, 1990; Lugo and Brown, 1993). Therefore, any conclusion based on surface soil responses to changes in soil OC and nutrients that occurred after forest clearing is conservative.

CONCLUSIONS

Proper management of forests for conservation of SOC and nutrients and their efficient utilisation in growth is required to sustain forest productivity, especially on nutrient-poor sites. Different extents of OM addition to the soil may be expected between trees of different species. Our results emphasise the importance of management, stand age, and forest type in affecting the size of C and nutrient stocks. Differences in stand microclimate, depth of tree rooting, quality and hence recalcitrance of above and below ground litter could have been contributed to the observed differences. Although the age of the plantations should be viewed with caution, practices such as shortened rotation lengths or improper management may have greater implications in C and nutrient storage. However, many of the factors that affect ecosystem processes and C storage such as microclimate, abundance and diversity of macro and micro fauna and flora, and the relationships between substrate quality and decomposition rates need to be further investigated. For a more complete picture, it is also important to determine the fate of cations and other nutrients present in trees and litter, which in turn requires investigation over more than one forest rotation. The data in C, N and S stocks, as related to forest type and disturbance, indicate that C and nutrient storage in forests of Ethiopia may be influenced by forest management activities. Although the changes in carbon associated with land use change do not define the total net flux of carbon between land and the atmosphere, they represent the portion of the flux that can be attributed to direct human activity, and it is this portion that is addressed by the United Nations Framework Convention on Climate Change and by the Kyoto Protocol (IPCC, 2000). Thus, comprehensive analysis of land use, land use change and forestry is needed at the national level to evaluate changes in emissions and mitigation requirements because in the long term a feasible solution is only possible at the international level as the beneficial impacts of reduced emissions in one country can easily be offset by the emissions in another. In view of the current climate change scenarios GHGs mitigation activities in Ethiopia have to be taken by the government because the current land ownership policy of the country that gives only use rights could not encourage investments on long-term returns such as planting trees or there should be incentives as farmers do not adopt practices unless they have ownership rights and improve profitability. The relative losses or accretion rates of C, N and S in the soil are determined by the interplay of physical, chemical and biological factors (Kirschbaum, 1995). Therefore, we must further improve our understanding of these processes to gain a better understanding of the likely future interaction between the global C cycle and climate change to be able to adequately anticipate the nature of the feed-back effects that link soil and atmospheric C reservoirs.

ACKNOWLEDGEMENTS: This work was supported by the German Research Foundation (DFG). We are grateful to the Katholischer Akademischer Ausländer-Dienst (KAAD) for a postgraduate scholarship to the first author.

REFERENCES

Allen, J. C., Soil properties to forest clearing in the United States and the tropics: Geological and biological factors, *Biotropica* **17**, 15–27 (1985).

Bashkin, M. A. and Binkley, A. D., Changes in soil carbon following afforestation in Hawaii, *Ecology* **79**, 828–833 (1998).

BMZ, Federal Ministry for Economic Co-operation and Development. Environment, poverity and sustainable development. Topic papers on the environment and sustainable utilisation of natural resources in development co-operation, GTZ, Frankfurt (2002).

Brown, S., Opportunities for mitigating carbon emissions through forestry activities. Prepared for the World Bank, March 1999. Winrock International, Virginia (1999).

Brown, S. and Lugo, A. E., Effects of forest clearing and succession on the carbon and nitrogen content of soils in Puerto Rico and US Virgin Islands, *Plant and Soil* **124**, 53–64 (1990).

Brown, S. and Lugo, A. E., The storage and production of organic matter in tropical forests and their role in the global carbon cycle, *Biotropica* **14**, 161–176 (1982).

Cuevas, E., Brown, S. and Lugo, A. E., Above-and below ground organic matter storage and production in a tropical pine plantation and a paired broadleaf secondary forest, *Plant and Soil* **135**, 257–268 (1991).

Detwiler, R. P., Land use change and the global carbon cycle: the role of tropical soils, *Biogeochemistry* **2**, 67–93 (1986).

EFAP, Ethiopian Forestry Action Plan. MoNRDEP, Addis Ababa, Ethiopia. Volume I. 14p. and Volume II. P. 34 (1994).

EFAP, Ethiopian Forestry Action Program. Volume II-The challenge for development. MoNRDEP. Addis Ababa, Ethiopia (1993).

Eijsackers, H. and Zehnder, A. J. B., Litter decomposition: A Russian Matriochka doll, *Biogeochemistry* **11**, 153–174 (1990).

Eswaran, H., van den Berg, E. and Reich, P., Organic carbon in soils of the world, *Soil Science Society of America Journal* **57**, 192–194 (1993).

FAO, FAO/UNESCO Soil map of the world, revised legend with corrections and updates. World soil resources, Report No. 60, ISRIC, Wageningen, The Netherlands (1997).

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

Fisher, R. F. and Binkley, D., Ecology and management of forest soils. 3rd ed. pp.435. John Wiley & Sons, Inc. New York (2000).

Follet, R. F., Soil management concepts and carbon sequestration in cropland soils, *Soil & Tillage Research* **61**, 77–92 (2001).

Grigal, D. F. and Ohmann, L. F., Carbon storage in Upland forests of the Lake States, *Soil Science Society of America Journal* **56**, 935–943 (1992).

Hendershot, W. H., Lalande, H. and Duquels, M., Ion exchange and exchangeable cations. In Carter, M. R. (ed.) Soil sampling and methods of analysis, pp. 167–176, Lewis Publishers, Boca Raton, FL (1993).

Homann, P. S., Sollins, P., Chappell, H. N. and Stangenberger, A. G., Soil organic carbon in a mountainous, forested region: Relation to site characteristics, *Soil Science Society of America Journal* **59**, 1468–1475 (1995).

IPCC, Land use, land-use change and forestry. WMO/UNEP. Summary report for policy Makers, p. 23, IPCC (2000).

IPCC, Climate Change 1995: the Science of Climate Change. Cambridge University Press, Cambridge, IPCC (1996a).

IPCC, Technologies, policies and measures for mitigating climate change. IPCC technical paper No. 1, p. 94, IPCC (1996b).

Johnson, D. W. and Curtis, P. S., Effects of forest management on soil C and N storage: meta analysis, *Forest Ecology and Management* **140**, 227–238 (2001).

Johnson, D. W., Carbon dynamics following clear cutting of a northern hardwood forests. W. W. McFee and J. M. Kelly (eds.), *Carbon forms and functions in forest soils*, pp. 299–329, *Soil Science Society of Americal*, Madison, WI (1995).

Johnson, D. W., Effects of forest management on soil carbon storage, *Water, Air, and Soil Pollution 64*, 83–120 (1992).

Johnson, C. M., Vieira, I. C. G., Zarin, D. J., Frizano, J. and Johnson, A. H., Carbon and nutrient storage in primary and secondary forests in eastern Amazonia, *Forest Ecology and Management* **147**, 245–252 (2001).

Kirschbaum, M. U. F., The temperature dependence of soil organic matter decomposition, and the effect of global warming on soil organic C storage, *Soil Biology and Biochemistry* **27**, 753−760 (1995).

Lardy, L. C., Brossard, M., Assad, M. L. and Laurent, J. Y., Carbon and phosphorus stocks of clayey Ferralsols in Cerrado native and agroecosystems, Brazil, *Agriculture, Ecosystems and Environment* **92**, 147−158 (2002).

Lilienfein, J., Wilcke, W., Thomas, R., Vilela, L., Lima, S. C. and Zech, W., Effects of Pinus caribaea forest on the C, N, P and S status of Brazilian savanna Oxisols, *Forest Ecology and Management* **147**, 171−182 (2001).

Liping, G. and Erda, L., Carbon sink in cropland soils and the emission of greenhouse gases from paddy soils: a review of work in China, *Chemosphere-Global Change Science* **3**, 413−418 (2001).

Lugo, A. E. and Brown, S., Management of tropical soils as sinks or sources of atmospheric carbon, *Plant and Soil* **149**, 27−41 (1993).

Lugo, A. E., Wang, D. and Bormann, F. H., A comparative analysis of biomass production in five tropical tree species, *Forest Ecology and Management* **31**, 153−166 (1990a).

Lugo, A. E., Cuevas, E. and Sanchez, M. J., Nutrients and mass in litter and top soil of ten tropical tree plantations, *Plant and Soil* **125**, 263−280 (1990b).

Lugo, A. E. , Brown, S. and Chapman, J., An analytical review of production rates and stemwood biomass of tropical forest plantations, *Forest Ecology and Management* **31**, 153–166 (1988).

Lugo, A. E., Sanchez, M. J. and Brown, S., Land use and organic carbon content of some subtropical soils, *Plant and Soil* **96**, 185−196 (1986).

Michelsen, A., Lisanework, N. and Friis, I., Impacts of tree plantations in the Ethiopian highland on soil fertility, shoot and root growth, nutrient utilisation and mycorrhizal colonisation, *Forest Ecology and Management* **61**, 299−324 (1993).

Minderman, G., Addition, decomposition and accumulation of organic matter in forests, *Journal of Ecology* **56**, 355−362 (1968).

Moraes, J. L., Cerri, C. C., Melillo, J. M., Kicklighter, D., Neill, C., Skole, D. L. and Steudler, P. A., Soil carbon stocks of the Brazilian Amazon basin, *Soil Science Society of America Journal* **59**, 244−247 (1995).

Mtambanengwe, F. and Kirchmann, H., Litter from a tropical savanna woodland (Miombo): Chemical composition and C and N mineralization, *Soil Biology and Biochemistry* **27**, 1639−1651 (1995).

Oades, J. M., The retention of organic matter in soils, *Biogeochemistry* **5**, 35–70 (1988). Pastor, J and Post, W. M., Influence of climate, soil moisture and succession on forest carbon and nitrogen cycles, *Biogeochemistry* **2**, 3−27 (1986).

Paul, K. I., Polglase, P. J., Nyakuengama, J. G. and Khanna, P. K., Change in soil carbon following afforestation, *Forest Ecology and Management* **168**, 241−257 (2002).

Pohjonen, V., Establishment of fuelwood plantations in Ethiopia, *Silva Carelica* **14**, 627−636 (1989).

Post, W. M., Pastor, J., Zinke, P. J. and Stangenberger, A. G., Global patterns of soil nitrogen storage, *Nature* **317**, 613–616 (1985).

Post, W. M., Emanuel, W. R., Zinke, P. J. and Stangenberger, A. G., Soil carbon pools and world life zones, *Nature* **298**, 156−159 (1982).

Ross, G. J. and Wand, C., Extractable Al, Fe, Mn and Si. In Carter, M. R. (ed.), *Soil sampling and methods of analysis,* pp. 239−246, Lewis Publishers, Boca Raton, FL (1993).

Simmons, J. A., Fernandez, I. J., Briggs, R. D. and Delaney, M., Forest floor carbon pools and fluxes along a regional climate gradient in Maine, USA, *Forest Ecology and Management* **84**, 81−95 (1996).

Smith, C. K., Oliveira, F. A., Gholz, H. L. and Baima, A., Soil carbon stocks after forest conversion to tree plantations in lowland Amazonia, Brazil, *Forest Ecology and Management* **164**, 257−263 (2002).

Solomon, D., Fritzsche, F., Tekalign, M., Lehmann, J. and Zech, W., Soil organic matter composition in the sub-humid Ethiopian highlands as influenced by deforestation and agricultural management, *Soil Science Society of America Journal* **66**, 68−82 (2002).

Solomon, D., Lehmann, J., Tekalign, M., Fritzsche, F. and Zech, W., Sulfur fractions in particle-size separates of the sub-humid Ethiopian highlands as influenced by land use changes, *Geoderma* **102**, 41–59 (2001).

Swift, M. J., Heal, O. W. and Anderson, J. M., *Decomposition in terrestrial ecosystems*. Blackwell, Oxford (1979).

Tate, K. R., Ross, D. J., O´Brien, B. J. and Kelliher, F. M., Carbon storage and turnover, and respiratory activity, in the litter and soil of an old-growth southern beech (*NOTHOFAGUS*) forest, *Soil Biology and Biochemistry* **25**, 1601−1612 (1993).

Tiessen, H., Cuevas, E. and Chacon, P., The role of soil organic matter in sustaining soil Fertility, *Nature* **371**, 783−785 (1994).

Turner, J. and Lambert, M., Change in organic carbon in forest plantation soils in eastern Australia, *Forest Ecology and Management* **133**, 231–247 (2000).

Turner, J., Organic matter accumulation in a series of *E. grandis* plantations, *Forest Ecology and Management* **17**, 231−242 (1986).

Turner, J. and Kelly, J., Effect of radiata pine on soil chemical characteristics, *Forest Ecology* *and Management* **11**, 257−270 (1985).

Van Cleve, K. and Powers, R. F., Soil carbon, soil formation and ecosystem development. W. W. McFee and J. M. Kelly (eds.), *Carbon forms and functions in forest soils,* pp. 155−200, Soil Science Society of America, Madison, WI (1995)

Veldkamp, E., Organic carbon turnover in three tropical soils under pasture after Deforestation, *Soil Science Society of America* **58**, 175−180 (1994).

Vogt, K. A., Vogt, D. J., Brown, S., Tilley, J. P., Edmonds, R. L., Silver, R. L. and Siccama, T. G., Dynamics of forest floor and soil organic matter accumulation in boreal, temperate and tropical forests. pp.159−178. R. Lal et al. (ed.), *Soil management and greenhouse effect*. CRC Press, Boca Raton, FL (1995)

Vogt, K. A., Grier, C. C. and D. J. Vogt., Production, turnover and nutrient dynamics of above- and belowground detritus of the world forests, *Advances in Ecological Research* **15**, 303−377 (1986).

Wilcke, W., Yasin, S., Abramowski, U., Valarezo, C. and Zech. W., Nutrient storage and turnover in organic layers under tropical montane rain forest in Ecuador, *European Journal of Soil Science* **53**, 15−27 (2002).

Zinn, Y.L., Resck, D.V.S., Silva, J.E.,.Soil organic carbon as affected by afforestation with *Eucalyptus* and *Pinus* in the *Cerrado* region of Brazil, *Forest Ecology and Management* **166**, 285−294 2002.

D Water and nutrient inputs by rainfall into natural and managed forest ecosystems in south-eastern highlands of Ethiopia

Y. Ashagrie[1]* and W. Zech[1]

[1]*Institute of Soil Science and Soil Geography, University of Bayreuth, D-95440 Bayreuth, Germany*
* Correspondence: yeshanew@hotmail.com, Tel: ++49-921-55-21-46 Fax: ++49-921-55-22-46

Abstract

Water and nutrient fluxes are two of the most important biogeochemical processes directly affected by changes in land-use and land-cover. The input of rain water to the forest floor, and the composition of rainfall and throughfall water were monitored between October 2001 and September 2002 in a natural and two plantation (*Eucalyptus globulus* and *Cupressus lusitanica*) forests at Munesa, south-eastern Ethiopia. Total annual rainfall was about 1190 mm, being very much skewed in distribution with the highest (60% of the total) during the main rainy season (June to September) and the lowest (12% of the total) during the dry season (October to January). The proportions of annual rainfall that passed through the different forest canopies were 53% under *Cupressus* and the natural forest, and 82% under *Eucalyptus*. The chemistry of rainafll was weakly acidic (pH 6.7) mainly dominated by Na, Cl and Ca. In all forest types, canopy interactions produced throughfall more alkaline relative to rainafll. Annual nutrients deposition by rainafall varied from 0.08 kg $ha^{-1}yr^{-1}$ for Mg to 3.79 kg $ha^{-1}yr^{-1}$ for Na. Annual nutrients deposition by throughfall ranged between 0.03 kg $ha^{-1}yr^{-1}$ for PO_4–P under *Cupressus* and 9.28 kg $ha^{-1}yr^{-1}$ for K under the natural forest. In all forest types, throughfall fluxes of K, Mg, Ca and Cl were enriched relative to rainfall indicating higher nutrient-status of the forests. The degree of enrichment was species and nutrient specific, but was generally lowest under *Cupressus*. Whereas, the depletion of NO_3–N, NH_4–N, SO_4–S and PO_4–P fluxes in throughfall of all forest types relative to rainfall indicate that these nutrients are limiting. Overall, the results indicate that throughfall volume was influenced largely by management (tree density) while throughfall

characteristics were influenced by species. The results of this study will be helpful to establish input-output budgets of nutrients and thereby predict the sustainability of the plantation forest ecosystems.

Key words: *Cupressus*; *Eucalyptus*; Ethiopia; Throughfall; Rainfall

1. Introduction

Nutrient fluxes and ecosystem productivity are driven to a large extent by the landscape water balance because water is both a limiting natural resource in many ecosystems and the driving fluid of most nutrient fluxes. Water and nutrient cycles are two of the most important biogeochemical processes directly affected by land use and land cover. Human-induced land use changes are known to affect the spatial and temporal patterns of landscape water fluxes (Bosch and Hewlett, 1982). Increasing concern on the ecological status of water resources has resulted in physical processes-based studies that examine the influence of vegetation on precipitation water quantity and quality. The quantity of precipitation passing through the forest canopy has a significant hydrological importance because it is responsible for the recharge of groundwater and streamflow, and influences runoff and distribution of understorey vegetation. Forest stands of different tree species differ in their above ground vegetation surface area, stand structure and morphology, and can have a differential impact on rain water interception and evapotranspiration losses, hence, on soil water regimes (Pritchett, 1979; Cape et al., 1991). For example Swank and Douglass (1974) in the United States found that streamflow was reduced by 20% by converting a deciduous hardwood stand to *Pinus strobus* L.

Precipitation also plays an important role in ecosystem nutrient cycles. Rainfall contains dissolved and particulate constituents; almost nearly all of the pool of nitrogen and sulphur including important quantities of inorganic nitrogen for forest growth are derived from the atmosphere (Waring and Schlesinger, 1985). Forests are particularly effective in scavenging and retaining atmospheric particles and gases that contain nutrients due to their high surface area and aerodynamic resistance (Howsam et al., 2000; Levia and Frost, 2003). Throughfall and stem flow are the two hydrological processes responsible for the transfer of precipitation and solutes from vegetative canopy to the soil (Levia and Frost, 2003) and have been documented to play a significant role in forest geochemical cycles (Parker, 1983). The chemistry of precipitation can be changed as it passes through the forest canopy resulting from wash-off of dry depositions and leaching of leaves and branches, and uptake of nutrients by the canopy (Jordan et al., 1980; Veneklaas, 1990). Ions such as Na^{+}, Cl^{-}, $SO_4^{2-}$and PO_4^{3-} usually flow passively through the canopies (Lindberg and Lovett, 1992; Ragsdale et al., 1992; Hultberg and Ferm, 1995). On the other hand, it has been documented that nitrogen can be absorbed by tree canopies from the atmospheric inputs, and this process relates to both ammonium and nitrate ions (Potter et al., 1991; Lovett, 1992; Shubzda et al., 1995; Stachurski and Zimka, 2000).

Many studies that have been carried out on the effects of atmospheric deposition on forest ecosystems have concentrated on countries with greater risk of air pollution (Krupa, 2002). However, even in the absence of air pollution risks, such studies are also of critical importance because of the potential ecological significance of atmospheric depositions in forest ecosystems nutrient cycling and the need for such information to make reliable forest management decisions. Such studies are also useful in understanding the level of atmospheric deposition and in evaluating air quality to undertake sound environmental management practices. The objectives of this study were (1) to assess the relative importance of atmospheric deposition in the nutrient cycle and (2) to determine the effects of land use and land cover change on the hydrological and nutrient cycles in the Munesa forest, thus contributing in the development of sustainable land use systems.

2. Materials and methods

2.1. Study site and experimental design

The experimental site was located at the Munesa/Shashemene forest enterprise site (7°34´N and 38°53´E), 240 km south-east of Addis Ababa. The long term average annual precipitation of the area is about 1250 mm (Solomon et al., 2002) and shows seasonal pattern: main rainy season (June to September) and small rainy seasons (February to May) with a relatively dry season from October to January (NMSA, 1996). The soil parent materials are of volcanic origin and soils can be classified as Nitisols (FAO, 1997). A natural forest and two plantation forests (*Eucalyptus globulus* and *Cupressus lusitanica*) situated side by side were selected for this study. The natural forest was a montane tropical forest mainly dominated by old growth *Podocarpus falcatus* trees. The two plantations were established in 1980 after clearing of part of the natural forest. *Eucalyptus globulus* had 595 stems ha $^{-1}$ with a mean diameter at breast height (dbh) of 19–39 cm and a height of 30–40 m. Tree density in the *Cupressus* plantation was 672 ha $^{-1}$ with mean dbh of 29 cm and a height of 18–20 m. For each forest type, three 20x30 m plots were identified, randomly distributed and within each plot an area of about 20–25 m^2 was fenced for throughfall collector installation.

2.2. Field equipment and sampling

An automatic weather data-logger was installed at the site to record the daily weather data including rainfall. We also kept a deep freezer in a nearby town for sample storage. Rainfall and throughfall were collected using plastic funnels of 12 cm diameter and 2 l capacity established 1 m above the ground. Rainfall was collected at three localities, each with three collectors, in a large opening between the plantation and natural forests. Throughfall was collected in three replicated plots with five collectors per plot. The collectors were placed 0.8 to 1m from the trunk of the sample tree. Table tennis balls were put inside each collector to prevent loss of water by evaporation. The sampling period started in October 2001 and concluded in September 2002. Water was collected on a weekly basis. During sample collection the volume of water was registered. After each collection, the collectors were washed with deionized water or with a portion of the sample water. On each sampling day, water samples were transported to the storage facility and kept frozen until they were transported in a cool box to Germany for chemical analysis. Samples were filtered through 0.45 µm glass fibre filter papers (Schleicher & Schuell). After filtration, samples from the different collectors in one plot were proportionally bulked per plot and per sampling day. From these samples about 100 ml were used for chemical analysis.

2.3. Chemical analysis

Analysis in both rainfall and throughfall samples include pH (Orion U402-S7), total content of Ca, K, Mg, Na (plasma emission spectroscopy, ICP-AES, Integra XMP), NO_3^-, NH_4^+, PO_4^{3-}, SO_4^{2-}, Cl^- (ion chromatography, Dionex 2000i-SP). Detection limits (mg l^{-1}) were: 0.025 for NH_4^+, 0.2 for Ca^{2+}, Na^+ and Mg^{2+}, 0.25 for K^+, 0.27 for Cl^-, 0.34 for NO_3^-, 0.28 for PO_4^{3-} and 0.32 for SO_4^{2-}.

2.4. Calculations and statistical analysis

All calculations for a particular parameter in a given season were based on mean values of three plots per forest type. Volume weighted concentrations and depositions of nutrients were calculated for each season, i.e. dry season (DS), small rainy season (RS1) and main rainy season (RS2). Volume Weighted Mean Concentration of the i-nutrient ($VWMC_i$) in rainfall and throughfall were estimated from the paired measurements of nutrient concentration, and rainfall and throughfall volume in each plot.

$$VWMC_i = \Sigma^n{}_{j=1}\, C_{ij}\, .\, TF_j \,/ \Sigma^n{}_{j=1}\, TF_i \qquad (1)$$

where C_{ij} is the i-nutrient concentration in throughfall on the j-collection day, TF is the total throughfall water volume and n is the total number of sampling dates. Using rainfall and throughfall water volume, concentrations were converted into gram quantities of various nutrients cycled in liquid form for each season and summed to yield annual inputs. Canopy exchange (i.e. canopy leaching and canopy uptake) was calculated as the difference between throughfall flux of a particular nutrient and its atmospheric deposition to the rain collectors. Significance of differences of a given parameter between rainfall and throughfall, and among throughfall of the three forest types were assessed with two way ANOVA using MSTAT-C statistical package. Correlation analysis was conducted between pairs of nutrients in rainfall/throughfall and rainfall/throughfall volume and nutrient concentrations.

3. Results and discussion

3.1. Water flux

Total annual rainfall and throughfall, and its seasonal distribution during the study period (October 2001 to September 2002) are shown in Table D1. Total annual rainfall was about 1190 mm, lying very close to the past long term mean value (1250 mm) from the nearby meteorological station (Solomon et al., 2002). There was a marked variation in the distribution of rainfall among the different seasons because in Ethiopia rainfall is mainly associated to a change in the predominant wind direction (monsoon); northeast winds prevail during the dry season and westerly to southwesterly winds during the rains (NMSA, 1996). Of the total annual rainfall, the highest amount (60%) fell during the main rainy season (June to September) and the least (12%) during the dry season (October to January). As with the distribution of the total annual rainfall across the different seasons, the distribution of rainfall among the different months within a given season was very much skewed. The monthly maximum and minimum rainfalls, respectively, were 67.4 and 6.2 mm in the dry season, 136.4 and 20.8 mm in the small rainy season and 268.2

and 120 mm in the main rainy season. Daily minimum rainfall was the same in all the three seasons (0.2 mm) while the daily maximum was variable; amounting 8.2 mm in the dry season,

Table D1. Rainfall (R) and throughfall (TF) at Munesa, Ethiopia, and other selected montane tropical forest ecosystems.

Location	Altitude (m)	R/TF	DS	RS1	RS2	Total
Ethiopia[1]	2400					
Rainfall		R	141	329	720	1190
			(0.01)	(0.01)	(0.002)	(0.02)
Natural forest		TF	52b*	189b*	384b*	625b*
			(0.003)	(0.003)	(0.02)	(0.03)
Eucalyptus		TF	96a*	272a*	602a*	970a*
			(0.004)	(0.01)	(0.02)	(0.03)
Cupressus		TF	50b*	182b*	404b*	636b*
			(0.01)	(0.002)	(0.02)	(0.03)
Puerto Rico[2]	425	R	–	–	–	3750
		TF	–	–	–	2774
Malaysia[3]	870	R	–	–	–	2700
		TF	–	–	–	–
Panama[4]	1200	R	–	–	–	3510
		TF	–	–	–	2190
Jamaica[5]	1250–1310	R	–	–	–	2539
		TF	–	–	–	*c*.1270
Ecuador[6]	1900–2010	R	–	–	–	2193
		TF	–	–	–	943–1996
New Guinea[7]	2450	R	–	–	–	3800
		TF	–	–	–	2585
Colombia[8]	2550	R	–	–	–	2115
		TF	–	–	–	1854
Tanzania[9]	2300	R	–	–	–	2220–2405
		TF	–	–	–	1695–1705

Sources: [1]This study; [2]Veneklaas (1990) [3]Bruijnzeel et al. (1993); [4]Cavelier et al. (1997); [5]McDonald and Healey (2000); [6] Wilcke et al. (2001); [7]Edwards (1982); [8]Veneklaas (1991); [9]Schrumpf (2004).
DS-dry season; RS1-small rainy season; RS2-main rainy season. Values followed by the same letter in a column are not different and those followed by * are different from those of rainfall. Values in parentheses are standard errors (n=3).

39.2 mm in the small rainy season and 60 mm in the main rainy season. Of the 12 months, rainfall was less than 100 mm during the dry season and at the beginning of the small rainy season (February) and was above 200 mm only in August.

The proportion of annual rainfall that passed through the *Eucalyptus* canopy (82%) was significantly higher than the cooresponding value for *Cupressus* and the natural forest (53%) (Table D1). This means that, when stem flow is unaccounted for, about 18% to 47% of the total rainfall was intercepted. This variation was mainly attributed to the difference in leaf morphology, branch geometry and hydrophobicity, stand density and total canopy area among species. However, the possibility of spatial variations in rainfall intensity within the study area could not be ruled out. In general, interception loss was highest during the dry season (65% in *Cupressus*, 63% in the natural forest and 32% in *Eucalyptus*) not only due to the pronounced sunny days before and after rain events, but also rainfall intensity for most of the rain events was too low (< 5 mm) to produce throughfall. This has implications not only on soil moisture level but also on the input and distribution of nutrients to the different compartments of the ecosystem and their availability to plants. There was no big variability in interception between the two rainy seasons. During the monitoring period, throughfall water fluxes under the different forest types were generally less than rainfall (Table D1). This is expected since cloud water is not a factor.

Rainfall and throughfall values for different montane tropical forest areas are summarized in Table D1. Except for Jamaican and part of the Ecuadorian montane forests, throughfall values are above 60% of rainfall. In Brazil, Lilienfein and Wilcke (2004) found that throughfall was 75–85% of rainfall (1682 mm) under *Pinus caribaea* plantation. Variability in throughfall amount between different studies can be attributed in part to differences in climatic patterns, meteorological conditions, and stand density and species composition. In the Munesa forest, long sunny periods were common even during the wetter months and so there was usually plenty of time for the canopy to dry out.

3.2. Chemistry of rainfall and throughfall

Rainfall at Munesa was weakly acidic (mean pH 6.7) with most of the potential acidity being neutralised by Na and Ca (Table D2). On an equivalent basis, Na was the dominant nutrient accompanied by Cl and Ca. The VWM nutrient concentrations in rainfall ranged from 0.09 mg l^{-1} for Mg to 3.29 mg l^{-1} for Na (Table D2). VWM concentration of NH_4–N was 1.78 times higher than that of NO_3–N. It has been suggested that rain water has only minor importance as source of NO_3–N relative to NH_4–N (Stachurski and Zimka, 2002). Calcium concentration was 1.78 times greater than that of K.

Correlation analysis between pairs of nutrients in rainfall (Table D3) revealed that PO_4–P was related with more nutrients than SO_4–S and NO_3–N. None of the other nutrient pairs showed significant relationship with the exception of the significant relationship of NH_4–N with K and Mg. Nutrient concentrations in rainfall at Munesa were higher than the mean values obtained by Parker (1983) and other montane tropical forest sites summarised in Table D2 except for Mg which was lower than the Puerto Rican and Colombian sites. Perhaps dust is responsible for the observed high values at Munesa where the occurrence of sporadic but strong dust carrying wind storms are common in the surrounding semiarid rift valley areas. The differences in rainfall amount and frequency between our study site and the others could also be another source of variability. The molar ratio Ca/Na in seawater is 0.04. The corresponding values in rainfall at Munesa during the dry and rainy seasons were 0.63 and 0.48, respectively, suggesting that most of the Ca was of continental origin and from biomass burning.

In all forest types, canopy interactions produced throughfall more alkaline than bulk precipitation. The average pH values of throughfall were 7.8 in the natural forest, 7.3 in *Eucalyptus* and 7.1 in *Cupressus* stands (Table D2). The effect of forest type on throughfall pH was not significant, although pH under the two plantations tended to be slightly lower than under the natural forest. Below canopy reduction in acidity has been reported in other forest ecosystems (Veneklaas, 1990; Laclau et al., 2003; Williams et al., 2004). VWM concentrations of throughfall Ca, K, Mg and Cl were significantly increased in relation to rainfall (Table D2). There was also a slight but non-significant increase in throughfall Na concentration under all forest types in comparison to that of rainfall. Increases in K and Mg concentrations relative to those of rainfall were variable among the three forest types, being highest under the natural forest compared to the two plantations. With respect to N and P, except for NH_4–N and PO_4–P which were significantly higher in rainfall in relation to throughfall of each forest type and the two plantations, respectively, the data show differences between tree species in their ability to alter the concentrations in rainfall. The concentration of NO_3–N in rainfall was significantly lowered after passing through the canopy of *Cupressus* plantation, while under *Eucalyptus* and the natural forest the reverse holds true. Although statistically not significant for *Eucalyptus*, the concentration of SO_4–S in all forest types had increased after the passage through the canopy (Table D2). The ratios of throughfall to rainfall nutrient concentration (concentration ratios) at Munesa were higher for K, Mg and Ca and lower for Na and NH_4–N than those in Tanzanian montane forest (Schrumpf, 2004). The concentration ratios of Na in Puerto Rico, Ca in New Guinea were higher than the values for our study site while the Colombian and Ecuadorian sites had higher ratios for many of the nutrients except for Cl and Mg. The concentration ratios for K in the natural forest and *Eucalyptus* plantation of our study site were higher than those of the Colombian, Puerto Rican and New Gunean sites. The increase/decrease in rainfall nutrient

Table D2. Volume weighted mean annual nutrient concentrations (average of the three seasons, mg l^{-1}) and pH in rainfall (R) and throughfall (TF) at Munesa and other montane tropical forests

Location		Ca	K	Mg	Na	Cl	NH_4–N	NO_3–N	SO_4–S	PO_4–P	pH
Ethiopia[1]											
Rainfall	R	1.60*	0.90*	0.09*	3.29	1.83*	1.53*	0.86*	1.59*§	0.22*+	6.7*
		(0.1)	(0.3)	(0.0)	(0.3)	(0.1)	(0.3)	(0.1)	(0.1)	(0.1)	(0.5)
Natural forest	TF	4.94a	21.2a	2.23a	3.89a	8.65a	0.89a	1.53a	2.26a	0.28a	7.8a
		(0.6)	(3.7)	(0.3)	(0.4)	(1.1)	(0.02)	(0.3)	(0.2)	(0.1)	(0.8)
Eucalyptus	TF	4.29a	10.3b	1.20b	3.57a	4.96b	0.41b	1.06ab	1.63a	0.07b	7.3a
		(0.4)	(1.5)	(0.1)	(0.3)	(0.4)	(0.1)	(0.2)	(0.1)	(0.0)	(0.3)
Cupressus	TF	4.51a	10.4b	1.11b	3.39a	6.44ab	0.83a	0.53b	2.10a	0.06b	7.1a
		(0.5)	(2.0)	(0.1)	(0.3)	(1.0)	(0.2)	(0.1)	(0.1)	(0.0)	(0.5)
Puerto Rico[2]	R	0.58	0.49	0.13	1.53	–	–	–	–	–	–
	TF	1.25	5.59	0.33	3.00	–	–	–	–	–	–
New Guinea[3]	R	0.10	0.19	0.03	–	–	–	–	–	–	–
	TF	0.87	3.03	0.47	–	–	–	–	–	–	–
Ecuador[4]	R	0.18	0.17	0.06	0.86	0.54	0.12	0.13	–	–	5.3
	TF	0.9–2.3	5.3–13	0.4–1.7	1.1–1.6	1.03–2.41	0.3–0.4	0.43–0.99	–	–	4–5.7
Colombia[5]	R	0.48	0.38	0.15	1.14	0.92	0.86	–	1.24	0.034	4.4
	TF	1.46	5.14	0.58	1.45	1.96	1.16	–	2.20	0.09	5.6
Tanzania[6]	R	0.1–0.2	0.3–0.5	0.04–0.05	0.3–0.4	–	0.2	0.1-0.2	–	–	–
	TF	0.13–0.34	1.5–3.0	0.1–0.2	0.5–0.9	–	0.2	0.03	–	–	–

Sources: [1]This study; [2]Veneklaas (1990); [3]Edwards (1982); [4]Wilcke et al. (2001) [5]Veneklaas (1991); [6]Schrumpf (2004).

Values followed by the different letter in a column are different. Values followed by * in rainfall are different from those of throughfall of each forest type and those followed by § and + are not different from those of *Eucalyptus* and natural forest, respectively. Values in parentheses are standard errors (n=3).

concentrations as it passes down through the forest canopy is the result of numerous, well-documented canopy interactions, which change the composition of rainfall resulting in enrichment (foliar leaching) or losses (foliar uptake) with regard to throughfall concentrations (Parker, 1983; Lindberg et al., 1986).

Throughfall VWM nutrient concentrations were found to be consistently higher in the natural forest than in *Cupressus* and *Eucalyptus* although the differences for some of the nutrients were not significant (Table D2). This was caused by differences in dry deposition and canopy interception capacity which is a result of several factors such as stand density, canopy area and leaf morphology. The roughness of the natural forest canopy together with its high leaf area index (Fetene and Beck, 2004) compared to the two even-aged plantations might have enabled to effectively scavenge dry depositions. In addition to washing of materials deposited on the canopy surface, internal sources possibly exudation of intracellular solutes from transpiring leaves during dry periods could also contribute to such enrichment (Parker, 1983). The leathery leaves of *Eucalyptus* and the needles of *Cupressus* were likely preventing the leaching of their water-soluble nutrients by rainfall. The lower nutrient concentrations and higher volumes of *Eucalyptus* throughfall reflect the reduced contact between precipitation and the *Eucalyptus* plantation canopy compared to the natural forest. The concentration of NH_4–N in throughfall below *Cupressus* was significantly higher than below *Eucalyptus*. The concentrations of Ca, Na, and SO_4–S in throughfall were not significantly variable among the three forest types. Throughfall nutrient concentrations were dominated by K>Cl>Ca>Na>SO_4–S in all forest types relative to the other nutrients. The concentrations of most nutrients in throughfall of the natural forest at Munesa were generally higher than the concentrations in other montane tropical forests (Table D2).

Table 3. Correlation coefficients for (i) nutrient concentrations and rainfall (R)/throughfall (TF) volume and between pair of nutrients in (ii) rainfall, and throughfall of (iii) the natural forest, (iv) *Eucalyptus* and (v) *Cupressus.*

i. R/TF	Ca	K	Mg	Na	Cl	NH_4–N	NO_3–N	SO_4–S	PO_4–P
Rainfall	ns	ns	ns	ns	ns	ns	ns	ns	ns
Natural forest	-0.88**	-0.70*	-0.81**	ns	-0.83**	-0.76*	-0.89**	ns	-0.72*
Eucalyptus	0.89**	ns	0.81**	ns	-0.89***	-0.87**	-0.91**	ns	ns
Cupressus	-0.88**	-0.88*	-0.89***	ns	-0.77**	ns	-0.71**	ns	ns
ii. Rainfall									
Ca	–	ns	ns	ns	ns	ns	ns	ns	ns
K		–	ns	ns	ns	0.88***	ns	ns	0.90**
Mg			–	ns	ns	0.76*	0.68*	ns	0.68*
Na				–	ns	ns	ns	0.79**	ns
Cl					–	ns	ns	ns	ns
NH_4–N						–	ns	ns	0.95**
NO_3–N							–	ns	ns
SO_4–S								–	ns
iii.Natural forest									
Ca	–	0.80**	0.85**	ns	0.94***	0.70*	0.88***	ns	ns
K		–	0.91***	ns	0.92***	ns	0.89***	ns	0.66*
Mg			–	ns	0.90***	ns	0.94***	ns	ns
Na				–	ns	ns	ns	0.88	ns
Cl					–	0.68*	0.92***	ns	0.72*
NH_4–N						–	0.77**	0.66*	ns
NO_3–N							–	ns	0.67*
SO_4–S								–	ns
iv.Eucalyptus									
Ca	–	ns	0.93***	ns	-0.71*	-0.66*	-0.73*	ns	ns
K		–	ns	ns	ns	ns	ns	ns	ns
Mg			–	ns	-0.70*	-0.71*	-0.75*	ns	ns
Na				–	ns	ns	ns	0.80**	ns
Cl					–	0.83**	0.82**	ns	0.69*
NH_4–N						–	0.98***	ns	0.81**
NO_3–N							–	ns	0.73*
SO_4–S								–	0.84**
v.*Cupressus*									
Ca	–	0.98**	0.97***	-0.67*	0.79**	ns	0.75*	ns	ns
K		–	0.99***	-0.73*	0.75*	ns	0.84**	ns	ns
Mg			–	-0.69*	0.73*	ns	0.82**	ns	ns
Na				–	-0.74*	ns	-0.73*	ns	ns
Cl					–	ns	ns	ns	ns
NH_4–N						–	ns	ns	ns
NO_3–N							–	ns	ns
SO_4–S								–	ns

*** P<0.001; ** P<0.01; * P<0.05; ns-not significant.

Magnesium and NH4–N in the higher altitudes of Ecuador and NH_4–N, PO4–P and SO4–S in Colombia were higher than the values for the two plantation forests at Munesa. Sulfate S in Colombian, Na in Puerto Rican and K in Ecuadorian forests were more or less similar to the values in our study forests. Table D3 shows that throughfall NO_3–N and Cl concentrations were well correlated with most of the nutrients in comparison to SO_4–S and PO_4–P, but the relationships with Ca and Mg in *Eucalyptus* and Na in *Cupressus* were negative. However, NH_4–N, SO_4–S and PO_4–P in throughfall below *Cupressus* and K below *Eucalyptus* did not show significant relationships with any nutrient. Sodium in throughfall under *Cupressus* was negatively correlated with more nutrients than that under *Eucalyptus* and natural forest in which Na was related only with SO_4–S.

The temporal trends in rainfall nutrient concentrations (Table D4) indicate that except for Na, which showed a slight increasing tendency from the dry season to the wet season no other nutrients showed a discernible trend with time. This was further reflected by the non-significant relationship of each nutrient and rainfall volume. The ratios of dry season to main rainy season element concentrations were 1.01 for Ca, 2.19 for K, 2.8 for Mg, 3.5 for NH_4–N, 3 for PO_4–P, 1.82 for NO_3–N, 1.06 for Cl, 0.77 for Na and 0.85 for SO_4–S. These figures confirm that the edaphic and biomass burning sources are more important in the dry season for most of the nutrients. In the wet season, the gases released from agricultural fields and combustion of fuels can be progressively scavenged by rain bearing clouds. This is particularly true for SO_4–S. The seasonal patterns in nutrient concentrations in throughfall of each forest type were similar (Table D4), being highest, with few exceptions, during the dry season (October–January) presumably due to wash-off of dry deposition accumulated on the canopy during dry periods by intermittent low-volume rain events. Reductions in nutrient concentrations during the wet period relative to the dry period are no doubt a result of frequent washing of the canopy and dilution effects. The ratios of dry season to wet season nutrient concentrations in throughfall followed the trend in rainfall. With the exceptions of Na and SO_4–S in all forest types, K in the throughfall under *Eucalyptus* and PO_4–P under *Cupressus* and *Eucalyptus*, the correlations between nutrient concentrations and throughfall volume of each forest type were negative and significant. This indicates that the concentrations in throughfall were diluted in proportion to the total rainfall which in turn indicates that nutrient depositions at Munesa may be limited by atmospheric concentrations and internal sources and not by rainfall. Interestingly, the relationship of Ca and Mg with throughfall volume under *Eucalyptus* were positive suggesting that the amounts of leachable Ca and Mg from the leathery leaves of *Eucalyptus* become increasing when rainfall volumes are large or intensities are high.

3.3. Nutrient fluxes

The annual total weights of nutrients (Na, Ca, K, Mg, NH_4–N, Cl, SO_4–S, NO_3–N, PO_4–P) reaching the soil (Table D5) were 14 kg $ha^{-1}yr^{-1}$ under *Cupressus*, 24 kg $ha^{-1}yr^{-1}$ under *Eucalyptus* and 21 kg $ha^{-1}yr^{-1}$ under the natural forest. Of these values 12 kg $ha^{-1}yr^{-1}$can be explained by the incident rainfall, while 2, 9, and 12 kg $ha^{-1}yr^{-1}$ under *Cupressus*, natural forest and *Eucalyptus*, respectively, derived from dry deposition and leaching of intracellular solutes from the canopy. These variations in total weight of nutrient amounts among the forest types were mainly due to the variability in throughfall water volume, canopy interaction and canopy surface area, and leachability of leaves and branches. In spite of the same amount of throughfall water with that of *Cupressus* and about 30% less than *Eucalyptus*, the annual total weight of nutrients reaching the ground under the natural forest was higher than under *Cupressus* and very close to that of *Eucalyptus*. This suggests that the much rougher surface of the natural forest canopy increased the deposition area and effectively intercepted dust carrying winds. In addition, the leaves and branches of natural forest appear to be more leachable than the needles of *Cupressus* and the leaves of *Eucalyptus*.

Table D4. Volume weighted mean nutrient concentrations (mg l^{-1}) in rainfall and throughfall of the three forest types in the dry season (DS), small rainy season (RS1) and main rainy season (RS2).

	Season	Ca	Cl	K	Mg	Na	NH_4−N	NO_3−N	PO_4−P	SO_4−S
Rainfall	DS	1.85A	1.87A	0.81A	0.14A	2.92A	1.96A	1.22A	0.24A	1.53A
		(0.08)	(0.16)	(0.27)	(0.03)	(0.62)	(0.32)	(0.04)	(0.12)	(0.16)
	RS1	1.11B	1.84A	1.53A	0.09A	3.18A	2.07A	0.69A	0.33A	1.45A
		(0.04)	(0.03)	(0.57)	(0.04)	(0.35)	(0.40)	(0.07)	(0.28)	(0.06)
	RS2	1.84A	1.77A	0.37A	0.05A	3.77A	0.56A	0.67A	0.08A	1.80A
		(0.08)	(0.05)	(0.15)	(0.02)	(0.11)	(0.22)	(0.23)	(0.03)	(0.03)
Natural forest	DS	6.62*a	11.72*a	29.7*a	3.29*a	4.62*a	1.97a	2.71*a	0.47a	2.84*a
		(0.54)	(1.84)	(7.9)	(0.52)	(0.53)	(0.64)	(0.33)	(0.14)	(0.21)
	RS1	4.86*abcd	8.82*a	21.8*ab	2.00*abc	3.11a	0.55a	1.34c	0.15a	1.98a
		(0.89)	(0.68)	(3.64)	(0.23)	(0.85)	(0.003)	(0.30)	(0.07)	(0.35)
	RS2	3.35d	5.41*a	12.2*bcd	1.40*bc	3.95a	0.15a	0.55def	0.03a	1.98a
		(0.37)	(0.99)	(3.06)	(0.29)	(0.31)	(0.02)	(0.18)	(0.01)	(0.07)
Eucalyptus	DS	3.45*d	6.64*a	9.08cd	0.87bc	4.09a	0.88a	1.94b	0.14a	1.88a
		(0.09)	(0.19)	(3.07)	(0.05)	(0.63)	(0.08)	(0.10)	(0.05)	(0.38)
	RS1	3.84*cd	4.80*a	13.2bcd	1.24bc	2.79a	0.29a	0.94d	0.05a	1.46a
		(0.61)	(0.46)	(3.08)	(0.21)	(0.49)	(0.07)	(0.14)	(0.03)	(0.19)
	RS2	5.71*abc	3.65*a	8.98cd	2.62ab	3.96a	0.08a	0.38f	0.03a	1.59a
		(0.23)	(0.17)	(0.84)	(0.99)	(0.46)	(0.01)	(0.10)	(0.001)	(0.07)
Cupressus	DS	6.15*ab	8.10*a	17.5*bc	1.52*abc	2.87a	1.06a	0.84de	0.06a	2.35a
		(0.5)	(0.60)	(0.41)	(0.11)	(0.7)	(0.30)	(0.11)	(0.02)	(0.22)
	RS1	4.22*bcd	7.90*a	9.01*cd	1.04*bc	3.17a	0.66a	0.45ef	0.09a	1.83a
		(0.42)	(1.83)	(1.41)	(0.09)	(0.43)	(0.23)	(0.09)	(0.04)	(0.30)
	RS2	3.17*d	3.31*a	4.77*d	0.76*c	4.14a	0.76a	0.30f	0.04a	2.10
		(0.16)	(0.37)	(1.86)	(0.02)	(0.19)	(0.41)	(0.13)	(0.01)	(0.19)

Means followed by the same lower case letters in a column are not different among forest types for each season ($P<0.05$). Rainfall values followed by the same upper case letters in a column are not different ($P<0.05$). Values followed by * in each forest type are different ($P<0.05$) from the corresponding values in rainfall. Numbers in parentheses are standard errors (n=3).

Table D5. Annual nutrient fluxes (kg $ha^{-1}yr^{-1}$) in rainfall and throughfall, leaching and canopy uptake in the different forest types, Munesa, Ethiopia

Nutrient	Rainfall	Natural forest	Net throughfall	Enrichment Factor	*Eucalyptus*	Net throughfall	Enrichment Factor	*Cupressus*	Net throughfall	Enrichment Factor
Ca	1.77	2.29b	0.52	1.29	4.36*a	2.59	2.46	2.13b	0.36	1.20
K	0.80	9.28*a	8.48	11.6	9.01*a	8.21	11.3	4.02*a	3.22	5.03
Mg	0.08	0.98*ab	0.90	11.3	1.22*a	1.14	15.3	0.52*b	0.44	6.5
Na	3.79	2.13*b	-1.66	0.56	3.22a	-0.57	0.85	2.17*b	-1.62	0.57
Cl	1.95	3.91*a	1.96	2.01	3.76*ab	1.81	1.93	2.87b	0.92	1.47
NH_4–N	1.25	0.24a	-1.01	0.19	0.19a	-1.06	0.15	0.44a	-0.81	0.35
NO_3–N	0.80	0.54a	-0.26	0.68	0.61*a	-0.19	0.76	0.23*a	-0.57	0.29
SO_4–S	1.81	1.16*a	-0.65	0.64	1.40a	-0.41	0.77	1.18*a	-0.63	0.65
PO_4–P	0.18	0.06a	-0.12	0.33	0.04a	-0.14	0.22	0.03a	-0.15	0.17

Values followed by different letters in a row are different. Values followed by * are different from those of rainfall.

Total annual deposition of nutrients by rainfall increased in the order: Mg<PO_4–P<K< NO_3–N < NH_4–N < SO_4–S ≤Ca< Cl<Na. According to Table D5 nearly all nutrients fluxes in the throughfall were significantly different. Only some nutrients behave different; e.g. NH_4–N and PO_4–P in all forest types, NO_3–N and Ca in *Cupressus* and the natural forest, and SO_4–S in *Eucalyptus*. Annual nutrient fluxes in rainfall and throughfall in the Munesa forest (Table D5) were lower than the values for other montane tropical forests (Table D6). The greatest variability in rainfall and throughfall inputs between our study and others could be due to variability in

rainfall amount, species composition and canopy structure, and the availability of nutrients from atmospheric and rock weathering processes and exposure to acid precipitation. Throughfall inputs of Ca, Na, Mg, and Cl were significantly different among forest types. Although statistically not significant for some of the nutrients, throughfall in *Cupressus* had the lowest fluxes of each nutrient compared to the natural forest and *Eucalyptus*, NH_4–N was an exception. *Eucalyptus* was found to have relatively the highest throughfall input of Ca, Na, Mg, SO_4–S and NO_3–N compared to the natural forest, mainly due to high volume of water under the *Eucalyptus* plantation. The inputs of PO_4–P and Cl were slightly highest under the natural forest compared to *Eucalyptus*, mainly resulting from high concentration.

Total annual throughfall input can be arranged in the following order: K > Cl > Na > Ca > SO_4–S >Mg > NO_3–N > NH_4–N > PO_4–P (Table D5). Net throughfall nutrient inputs result from a combination of dry deposition wash-off plus leaching of intracellular solutes from leaves and branches (Parker, 1983). Net throughfall data for K, Cl, Ca and Mg indicated leaching of these nutrients from the canopy (Table D5). Although atmospheric deposition of K was lower than those of Ca and Cl, net throughfall K was higher than Ca and Cl in all forest types. According to common opinions, ions absorbed by the canopy such as NH_4^+ and H^+ could be the cause of ion exchange reactions and consequent leaching of K^+, Ca^{2+} and Mg^{2+} from the plant tissues, potassium in particular. The magnitude of nutrient leaching varied from forest type to forest type. In general, leaching was lowest in *Cupressus* plantation and highest in the natural forest and *Eucalyptus* plantation with Ca and Mg in *Eucalyptus* and K and Cl in the natural forest being highest, suggesting that leaves are leached more than needles.

Table D6. Annual nutrient fluxes (kg $ha^{-1} yr^{-1}$) in rainfall (R) and throughfall (TF) in selected montane tropical forests.

Location	Altitude(m)	R/TF	Rainfall(mm)	Ca	Mg	K	Na	Cl	NH_4–N	NO_3–N	SO_4–S	PO_4–P
Puerto Rico[1]	425	R	3750	21.8	4.9	18.2	57.2	–	–	–	–	–
		TF	2774	34.8	9.2	155	83.2	–	–	–	–	–
Malaysia[2]	870	R	2700	4	1.2	4	–	23	2.5	5	24	0.1
		TF	–	12	7	23	–	59	8.5	10	43	0.11
Panama[3]	1200	R	3510	27.9	4.1	13.5	63.5	34.5	–	–	13.2	0.7
		TF	2190	35.1	7.6	63.2	131.2	49.6	–	–	6.1	2.2
Jamaica[4]	1250-1310	R	2539	–	–	–	–	–	–	–	–	–
		TF	c.1270	21.6	1.2	68	–	–	4.8	2.3	–	3
Ecuador[5]	1900-2010	R	2193	3.9	7.1	3.7	19–20	12	2.6	3	–	–
		TF	943–1996	15–28	7.1–21	76–142	22	15–30	2.6–7.2	5.9–12	–	–
New Guinea[6]	2450	R	3800	3.6	1.3	7.3	–	–	–	–	–	0.5
		TF	2585	22.6	12.2	78.4	–	–	–	–	–	3
Colombia[7]	2550	R	2115	10.1	3.2	7.9	24.1	19.4	18.3	–	26.2	0.72
		TF	1854	27.1	10.7	95.2	26.9	36.3	21.5	–	40.9	1.67
Tanzania[8]	2300	R	2220–2405	2.3	0.9	7.5	6.2	–	3.05	2.85	–	–
		TF	1695–1705	3.5	2	35	11	–	3.3	0.85	–	–

Sources: [1]Veneklaas (1990); [2]Bruijnzeel et al. (1993); [3]Cavelier et al. (1997); [4]McDonald and Healey (2000); [5]Wilcke et al. (2001); [6]Edwards (1982); [7]Veneklaas (1991); [8] Schrumpf (2004).

The data in net throughfall NH_4–N, Na, SO_4–S, NO_3–N and PO_4–P fluxes (Table D5) indicate absorption by the canopies of all forest types. The magnitude of absorption was different for the different nutrients; highest for Na in *Cupressus* plantation (1.62 kg $ha^{-1}yr^{-1}$) and natural forest (1.66 kg $ha^{-1}yr^{-1}$) and NH_4–N (1.06 kg $ha^{-1}yr^{-1}$) in *Eucalyptus* plantation and lowest for PO_4–P (range 0.12 to 0.15 kg $ha^{-1}yr^{-1}$). The annual amount of NH_4–N taken up by the natural forest and *Eucalyptus* plantation was slightly higher than that by the *Cupressus* plantation and was higher by about 5 times in *Eucalyptus* plantation, 4 times in the natural forest and 3 times in *Cupressus* plantation than NO_3–N uptake. The fact that NH_4–N was more readily assimilated or sorbed than NO_3–N was observed by Emmett et al. (1998) in N addition experiments with $NaNO_3$ and NH_4NO_3 where the ecosystem demand for NH_4^+ continued after NO_3^- leaching had occurred. *Eucalyptus* and *Cupressus* trees had slightly higher PO_4–P uptake compared to the natural forest. The amounts of SO_4–S and Na taken up were higher in the natural and *Cupressus* forests compared to *Eucalyptus*. Nitrate N was taken up in larger quantity by *Cupressus* than by the other two forest types (Table D5). Foliar uptake has been reported for N (Parker, 1983; Cavelier et al., 1997) and PO_4–P and occasionally for Ca and SO_4–S (Parker, 1983). In many instances foliar interaction with SO_4–S and Na is believed to be negligible, however, evidences to support the theory of SO_4–S (Mahendrappa, 1990; Cavelier et al., 1997) and Na (Moreno et al., 2001; Wilcke et al., 2001) absorptions by canopies are reported. Nutrient enrichment in throughfall relative to rainfall was higher for K and Mg as shown by high enrichment factors (Table D5).

Nutrient fluxes varied considerably from season to season and were highest during the wet season (Table D7). This can be attributed to higher rainfall volume or intensity although concentrations of most elements tended to be higher in the dry season (Table D4). This seasonal pattern of variation in fluxes indicated that except for the few relatively high-volume dry season rain events, throughfall in dry season is not likely to provide a major nutrient source via root uptake for overstorey tree species. Shallow-rooted understorey vegetation probably periodically utilise wet-deposited nutrients following dry season rain events of sufficient magnitude to infiltrate the upper rooting zone. Contrary, higher rainfall intensity or volume in the main rainy season may lead to nutrient losses in seepage and runoff. Increased deposition of nutrients from the dry season to the wet season was also reported elsewhere (Cavelier et al., 1997; Clark et al., 1998; Lilienfein and Wilcke, 2004). Comparison of net throughfall fluxes among seasons indicated clear temporal patterns of canopy leaching and very different chemical speciation associated with biological uptake (Fig. D1). As with total throughfall input, net throughfall inputs of nutrients were higher during the rainy season. Ammonium–N and PO_4–P were taken up in larger quantity during the small rainy season by all forest types, while Na, SO_4–S and NO_3–N were mainly taken up during the main rainy season. Calcium in *Cupressus* and natural forest indicated intermediate behaviour: a tendency towards absorption during the main rainy season and canopy leaching during the dry and small rainy seasons (Fig. D1). A similar behaviour was also observed for NO_3–N in *Eucalyptus* and the natural forest.

Table D7. Seasonal variability in nutrient fluxes (kg ha $^{-1}$ season $^{-1}$) in rainfall and throughfall of the three forest types.

	Season	Ca	Cl	K	Mg	Na	NH_4–N	NO_3–N	PO_4–P	SO_4–S
Rainfall	DS	0.24B (0.01)	0.24BC (0.01)	0.11A (0.04)	0.02A (0.04)	0.37C (0.07)	0.26A (0.05)	0.16A (0.01)	0.03A (0.02)	0.20BC (0.02)
	RS1	0.33B (0.01)	0.55B (0.01)	0.46A (0.17)	0.03A (0.01)	0.95B (0.09)	0.62A (0.42)	0.21A (0.02)	0.10A (0.08)	0.43B (0.02)
	RS2	1.20A (0.05)	1.16A (0.03)	0.24A (0.09)	0.03A (0.01)	2.47A (0.07)	0.37A (0.15)	0.44A (0.15)	0.05A (0.02)	3.53A 0.02)
Natural forest	DS	0.31c (0.04)	0.56ef (0.11)	1.44*bc (0.45)	0.16cd (0.03)	0.22a (0.02)	0.09a (0.03)	0.13a (0.02)	0.02a (0.01)	0.13a (0.00)
	RS1	0.82*b (0.14)	1.49*abc (0.11)	3.70*a (0.66)	0.34*bc (0.04)	0.53*a (0.14)	0.09a (0.00)	0.23a (0.05)	0.03a (0.01)	0.33a (0.06)
	RS2	1.16b (0.08)	1.86*ab (0.27)	4.15*a (0.92)	0.48*b (0.08)	1.39*a (0.17)	0.05a (0.01)	0.19a (0.06)	0.01a (0.00)	0.69*a (0.06)
Eucalyptus	DS	0.30c (0.01)	0.58*def (0.02)	0.76c (0.24)	0.08d (0.00)	0.35a (0.05)	0.08a (0.01)	0.17a (0.01)	0.01a (0.00)	0.16a (0.03)
	RS1	0.96*b (0.18)	1.19*cde (0.13)	3.30*ab (0.85)	0.31*bc (0.06)	0.70a (0.14)	0.07a (0.02)	0.23a (0.04)	0.01a (0.01)	0.36a (0.06)
	RS2	3.12*a (0.16)	1.99*a (0.05)	4.93*a (0.61)	0.84*a (0.07)	2.18a (0.31)	0.04a (0.01)	0.21a (0.06)	0.01a (0.01)	0.87*a (0.06)
Cupressus	DS	0.27c (0.02)	0.36f (0.02)	0.78*c (0.09)	0.07*d (0.01)	0.13*a (0.03)	0.05a (0.02)	0.04a (0.01)	0.003a (0.00)	0.11a (0.01)
	RS1	0.70*bc (0.06)	1.30*bc (0.28)	1.48*bc (0.22)	0.17*cd (0.01)	0.53*a (0.08)	0.11a (0.04)	0.08a (0.02)	0.01a (0.01)	0.30a (0.05)
	RS2	1.16b (0.04)	1.22bcd (0.15)	1.76*bc (0.21)	0.28*bcd (0.02)	1.51*a (0.02)	0.28a (0.14)	0.11a (0.05)	0.01a (0.01)	0.77*a (0.04)

Means followed by the same lower case letters in a column are not different among forest types for each season (P<0.05). Rainfall values followed by the same upper case letters in a column are not different (P<0.05). Values followed by * in each forest type are different (P<0.05) from the corresponding values in rainfall. DS: dry season, RS1: small rainy season, RS2: main rainy season. Numbers in parentheses are standard errors (n=3).

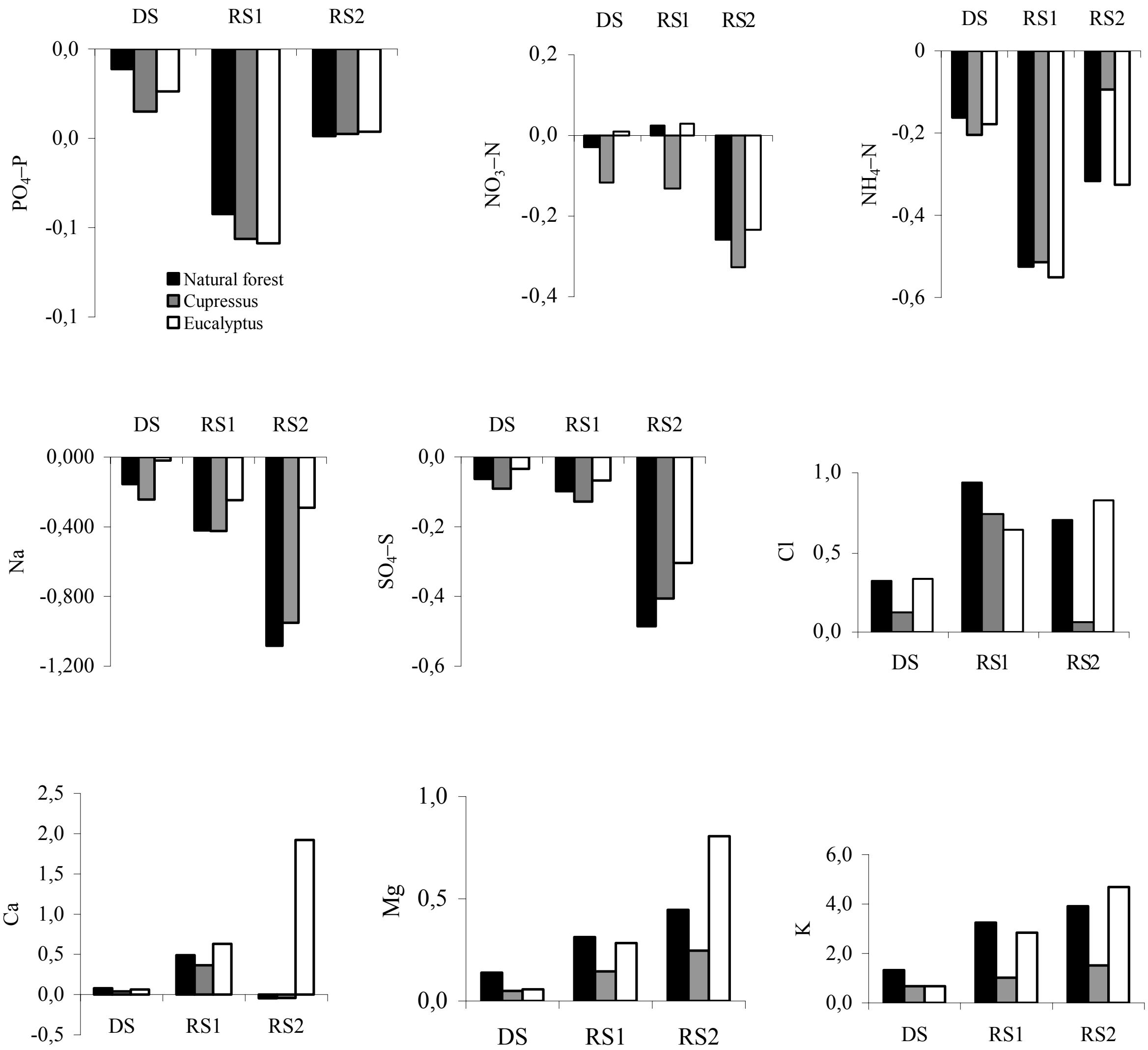


Fig. D1. Seasonal variations in net throughfall nutrient fluxes (kg $ha^{-1}season^{-1}$) in the three forest types. DS-dry season; RS1-small rainy season; RS2-main rainy season.

4. Conclusions

The results of this study showed that the natural forest and *Cupressus* plantation canopies intercepted large proportion of the total annual rainfall compared to the *Eucalyptus* plantation canopy. Most of the potential acidity in rainfall was neutralized by Na and Ca resulting in a weakly acidic pH. Nutrients such as K, Ca, Cl and Mg were leached from the canopy of each forest type making throughfall more alkaline than rainfall, in agreement with most of the literature. Nitrogen, phosphorous and sulphur were absorbed from atmospheric sources by the canopy of each forest type suggesting that these nutrients are limiting in the study area. Throughfall characteristics and the magnitude of the nutrient fluxes in throughfall are highly dependent upon crown density and species. The inputs of large proportion of rain water under *Eucalyptus* resulted in a proportionally higher nutrient fluxes followed by the natural forest. In general, except for K and Mg raifall seems to contribute in an important way to the annual nutrient demand for growth of the forests studied. The within-canopy source of inputs for those with lower external deposition (mainly of K and Mg) appeared to be one way that the forest stands recycle needed nutrients. The information from this study will be useful to establish nutrient input and output budgets of the different forest ecosystems. However, additional information is needed on the relationship between above-ground nutrient return pathways and below-ground processes such as soil chemical and biological processes, and nutrient mobility, particularly in monoculture plantation systems, where overall sustainability is often debated.

Acknowledgements

We thank the German Research Foundation (DFG) for its support of the research through grant number Gu406–8.

References

Bosch, J.M., Hewlett, J.D., 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evaporation. J. Hydrol. 55, 3–23.

Bruijnzeel, L.A., Waterloo, M.J., Proctor, J., Kuiters, A.T., Kotterink, B., 1993. Hydrological observations in montane forests on Gunung Silam, Sabah, Malaysia, with special reference to the `Massenerhebung´effect. J. Ecol. 81, 145–167.

Cape, J.N., Brown, A.H.F., Robertson, S.M.C., Howson, G., Parerson, I.S., 1991. Interspecies comparisons of throughfall and stemflow at three sites in northern Britain. For. Ecol. Manage. 46, 165–177.

Cavelier, J., Jaramillo, M., Solis, D., Leon, D., 1997. Water balance and nutrient inputs in bulk precipitation in tropical montane cloud forest in Panama. J. Hydrol. 193, 83–96.

Clark, K.L., Nadkarni, N.M., Schaefer, D., Gholz, H.L., 1998. Atmospheric deposition and net retention of ions by the canopy in a tropical montane forest, Monteverde, Costa Rica. J. Trop. Ecol. 14, 27–45.

Edwards, P.J., 1982. Studies of mineral cycling in a montane rain forest in New Guinea, V. Rates of cycling in throughfall and litter fall. J. Ecol. 70, 807–827.

Emmett, B.A., Reynolds, B., Silgram, M., Sparks, T.H., Woods, C., 1998. The consequences of chronic nitrogen additions on N cycling and soil water chemistry in Sitka spruce stand, North Wales. For. Ecol. Manage. 101, 165–175.

FAO, 1997. FAO/UNESCO Soil map of the world, revised legend with corrections and updates. World soil resources, Report No. 60, ISRIC, Wageningen, The Netherlands.

Fetene, M., Beck, E.H., 2004. Water relations of indigenous versus exotic tree species, growing at the same site in tropical montane forest in southern Ethiopia. Trees 18, 428–435.

Howsam, M., Jones, K.C., Ineson, P., 2000. PAHs in the soils of a mature, mixed-deciduous

(*Quercus-Fraxinus*) woodland and the surrounding pasture. Water Air Soil Pollut. 121, 379–398.

Hultberg, H., Ferm, M., 1995. Measurements of atmospheric deposition and internal circulation of base cations to a forested catchment area. Water Air Soil pollut. 85, 2235–2240.

Jordan, C., Golley, F., Hall, J., 1980. Nutrient scavenging of rainfall by the canopy of an Amazonian rain forest. Biotropica 12, 61–66.

Krupa, S.V., 2002. Sampling and physico-chemical analysis of precipitation: a review. Environ. Pollut. 120, 565–594.

Laclau, J.P., Ranger, J., Bouillet, J.P., Nzila, J.D., Deleporte, P., 2003. Nutrient cycling in a clonal stand of *Eucalyptus* and an adjacent savanna ecosystem in Congo. 1. Chemical composition of rainfall, throughfall and stemflow solutions. For. Ecol. Manage. 176, 105–19.

Levia, D.F., Frost, E.E., 2003. A review and evaluation of stemflow literature in the hydrologic and biogeochemical cycles of forested and agricultural ecosystems. J. Hydrol. 274, 1–29.

Lilienfein, J., Wilcke, W., 2004. Water and element input into native, agri- and silvicultural ecosystems of the Brazilian savanna. Biogeochemistry 67, 183–212.

Lindberg, S.E., Lovett, G.M., 1992. Deposition and forest canopy interactions of airborne sulphur: results from integrated forest study. Atmosph. Environ. 26, 1477–1492.

Lindberg, S.E., Lovett, G.M., Richter, D.D., Johnson, D.W., 1986. Atmospheric deposition and canopy interactions of major ions in a forest. Science 231, 141–145.

Lovett, G.M., 1992. Atmospheric deposition and canopy interactions of nitrogen. In: Johnson, D. W., Lindberg, S. E. (eds.), Atmospheric deposition and forest nutrient cycling. Ecological Studies 91. Springer-Verlag, New York, NY, USA, pp. 152–166.

Mahendrappa, M.K., 1990. Partitioning of rainwater and chemicals into throughfall and stemflow in different forest stands. For. Ecol. Manage. 30, 65–72.

McDonald, M.A., Healey, J.R., 2000. Nutrient cycling in secondary forests in the Blue Mountains of Jamaica. For. Ecol. Manage. 139, 257–278.

Moreno, G., Gallardo, J.F., Bussotti, F., 2001. Canopy modification of atmospheric deposition in oligotrophic *Quercus pyrenaica* forests of an unpolluted region (Central-Western Spain). For. Ecol. Manage. 149, 47–60.

NMSA, 1996. Climatic and agroclimatic resources of Ethiopia. Vol. 1, No. 1. National Meteorological Service Agency of Ethiopia, Addis Ababa. 137 pp.

Parker, G.G., 1983. Throughfall and stemflow in the forest nutrient cycle. Adv. Ecol. Res. 13, 57–133.

Potter, C.S., Ragsdale, H.L., Swank, W.T., 1991. Atmospheric deposition and foliar leaching in a regenerating southern Appalachian forest canopy. J. Ecol. 79, 97–115.

Pritchett, W.L., 1979. Properties and management of forest soils. John Wiley, New York, NY, 500 pp.

Ragsdale, H.L., Lindberg, S.E., Lovett, G.M., Schaefer, D.A., 1992. Atmospheric deposition and throughfall fluxes of base cations. In: Johnson, D.W., Lindberg, S.E. (Eds.), Atmospheric deposition and forest nutrient cycling. Ecological Studies 91. Springer-Verlag, New York, NY, USA, pp. 235–253.

Schrumpf, M., 2004. Biogeochemical investigations in old growth and disturbed forest sites at mount Kilimanjaro. Dissertation, PhD., Bayreuth University.

Shubzda, J., Lindberg, S.E., Garten, C.T., Nodvin, S.C., 1995. Elevation trends in the fluxes of sulphur and nitrogen in throughfall in the southern Appalachian Mountains: some surprising result. Water Air Soil Pollut. 85, 2265–2270.

Solomon, D., Fritzsche, F., Lehmann, J., Tekalign, M., Zech, W., 2002. Soil organic natter dynamics in the subhumid agroecosystems of the Ethiopian highlands: Evidence from natural ^{13}C abundance and particle-size fractionation. Soil Sci. Soc. Am. J. 66, 969–978.

Stachurski, A., Zimka, J.R., 2002. Atmospheric deposition and ionic concentrations within a beech canopy in the Karkonosze Mountains. Environ. Pollut. 118, 75–87.

Stachurski, A., Zimka, J.R., 2000. Atmospheric input of elements to forest ecosystems: a method of estimation using artificial foliage placed above rain collectors. Environ. Pollut. 110, 345–356.

Swank, W.T., Douglass, J.E., 1974. Streamflow greatly reduced by converting deciduous hardwood stands to pine. Science 185, 857–859.

Veneklaas, E.J., 1991. Litterfall, nutrient fluxes in two montane tropical rain forests, Colombia. J. Trop. Ecol. 7, 319–336.

Veneklaas, E.J., 1990. Nutrient fluxes in bulk precipitation and throughfall in two montane tropical rain forests, Colombia. J. Ecol. 78, 974–992.

Waring, R.H., Schlesinger, W.H., 1985. Forest ecosystems: Concepts and management. Academic Press, Orlando, USA, 340 pp.

Wilcke, W., Yasin, S., Valarezo, C., Zech, W., 2001. Change in water quality during the passage through a tropical montane rain forest in Ecuador. Biogeochemistry 55, 45–72.

Williams, M.R., Filoso, S., Lefebvre, P., 2004. Effects of land-use change on solute fluxes to floodplain lakes of the central Amazon. Biogeochemistry 68, 259–275.

E Dynamics of dissolved nutrients in forest floor leachates: Comparison of a natural forest ecosystem with tree species plantations in south-east Ethiopia

Y. Ashagrie[1]* and W. Zech[1]

[1]*Institute of Soil Science and Soil Geography, University of Bayreuth, D-95440 Bayreuth, Germany*
* Correspondence: yeshanew@hotmail.com,Tel: ++49-921-55-21-46 Fax: +49-921-55-22-46

Abstract

A large portion of the nutrient stocks in forest ecosystems may be preserved in the organic layer. The rate at which the accumulated nutrients are released and transferred to the different compartments of the ecosystem for recycling is of prime significance to forest productivity. The dynamics of nutrients in water passing through the forest floors of two plantation forests (*Cupressus lusitanica* and *Eucalyptus globulus*) and an adjacent natural forest were monitored over a one year period at Munesa, south-eastern Ethiopia. The results showed that, in all forest types, after K, Ca and Cl were the most abundant nutrients leached from the forest floor to the mineral soil. The concentration of PO_4–P was the least of all the nutrients analysed followed by NH_4–N. The concentration of NO_3–N in the natural forest was about 10 times higher than that of NH_4–N, but 8 and 3 times higher than that of NH_4–N under *Eucalyptus* and *Cupressus*, respectively. The lower NH_4–N concentration relative to NO_3–N was probably a result of nitrification, vegetation uptake, adsorption or assimilation by microbes. No significant differences in concentrations of most of the nutrients were observed among forest types. Magnesium and NO_3–N were significantly higher under the natural forest and *Eucalyptus* than under *Cupressus*. The low C/N ratios in the forest floors of *Eucalyptus* and natural forest might have triggered nitrification in comparison to *Cupressus* which had high C/N ratio in the organic layer. The concentration of K was higher under *Eucalyptus* than under the natural forest and *Cupressus*. Except for NH_4–N, which was depleted in relation to throughfall in the natural forest, the concentrations of all other nutrients were enriched in litter leachate in relation to both rainfall and throughfall. However, with the exceptions of NO_3–N in all forest types, Ca under *Cupressus* and PO_4–P under

Cupressus and *Eucalyptus*, all measured nutrient fluxes from the forest floor to the mineral soil decreased in relation to throughfall fluxes. Generally, the results show that despite the differences in tree species composition among the forest types the organic layer acted as a sink for most of the nutrients.

Key words: *Cupressus*, *Eucalyptus*, Forest floor leachate, Ethiopia

1. Introduction

The organic layer in forest ecosystems plays a key role in soil development, carbon and nutrient cycling, runoff control, moisture retention, and is often the medium in which fine root development occurs with priority (Wells and Davey, 1966). A large portion of the nutrient stocks in forest ecosystems may be preserved in the organic layer (Wells and Davey, 1966; Youngberg, 1966). The rate at which the accumulated nutrients are released and transferred to the different compartments of the ecosystem for recycling is of prime significance to forest productivity (Youngberg, 1966). Organic matter decomposition is a key process in the turnover and cycling of nutrients held in the organic layer of forest ecosystems. Numerous factors including forest type and composition are known to affect the decomposition processes of organic matter.

Modifications of forest floor properties that parallel changes in the forest cover/composition are mainly related to microclimate and the quality of the organic material (Berg et al., 1993). Of the various factors related to the quality of litter, the importance of nitrogen and phosphorus contents (Vogt et al., 1986), C/N ratio (Berg et al., 1998; Kurka et al., 2000) and lignin content (Johansson et al., 1995) have been emphasized. Species composition affects the quantity of litterfall (Ashagrie et al., 2003) and root litter (Helmisaari, 1995), stand structure also affects the amount of solar radiation (Zhang and Zark, 1995) and of precipitation reaching the soil (Mahendrappa, 1990), all of which affect decomposition and consequently nutrient transport to the mineral soil. One of the factors controlling the rate of decomposition of organic matter is the availability of readily soluble carbon by throughfall and stemflow, a ready source of carbohydrates for decomposers (Mahendrappa, 1990). Nutrients released in the forest floor by mineralization may be directly assimilated by roots or partly leached into the mineral soil, together with nutrients deposited by throughfall. It is likely that variation in the quantity of water passing the forest floor and root density in the organic layer influence these processes with consequent effects on nutrient leaching to the mineral soil. The objective of this study was to monitor the dynamics of nutrients in water as it passes through the forest floors of the different forest ecosystems growing under comparable site conditions.

2. Materials and methods

2.1. Study site

The study was conducted in the Munesa/Shashemene forest (7°34´N and 38°53´E) located on the eastern escarpment of the central Ethiopian rift valley, 240 km south-east of Addis Ababa. The study site is characterised by a sub-humid tropical climate with a mean annual rainfall of 1250 mm and mean annual temperature of 19 °C (Solomon et al., 2002). Rainfall is bimodal, most of it falling in July and August. The soils of the study area are classified as Nitisols (FAO, 1997). They are well drained, reddish brown in colour and characterised by increasing clay contents with increasing soil depth (49 − 50% clay in the A horizon; 63 − 74% clay in the Bt2 horizon). CEC varies between 37− 51 $cmol_c\ kg^{-1}$ soil in the A horizon. The physical and chemical properties of the organic layers under the three forest types are presented in Table E1. Plantations with *Cupressus lusitanica* and *Eucalyptus globulus*, and an adjacent natural forest were selected for this study. The forest plantations were established in 1980 after clearing of part of the natural forest. Clearing was done manually and the aboveground biomass was burned on site. The natural forest was dominated by old-growth *Podocarpus falcatus* trees and is regarded as a montane tropical forest. Tree density was 672 trees ha^{-1}, diameter at breast height (dbh) was 29 cm and height was 18–20 m for *Cupressus*. There was almost no ground vegetation under *Cupressus*. The *Eucalyptus* plantation was sparsely stocked with 595 trees ha^{-1} and had a native understorey canopy tree (*Croton macrostachys*) and shrubs, notably *Acanthopale pubescens*, *Achyrospermum schimperi*, *Bothriocline schimperi*, *Carex spicato-paniculata*, *Hypoestes forskaolli*. The ground layer was covered with dense grass and broad-leaved herbaceous species. The mean height of *Eucalyptus* was 30–40 m and dbh was 19–39 cm.

2.2. Equipment and sampling

In each forest type, three replicated (20x30 m) plots were randomly located. Within each plot, an area of about 20–25 m^2 was fenced for the installation of equipment. Zero-tension lysimeters made of plastic boxes (0.15 x 0.15 m) were placed horizontally in the contact zone between the forest floor and the mineral soil to measure the amount of water percolating through the forest floor. Three boxes were used per plot. The boxes were connected to a 2 l bottle placed in a soil pit. To avoid any solid material entering the boxes and bottles, a fine wire mesh (0.5 mm) was attached to the upper part of each plate. Samples were taken from October 2001 to September 2002 on a weekly basis. After sampling, the solution was immediately transported to the storage facility and kept frozen until they were transported in a cool box to Germany for chemical analysis. Samples were filtered through pre-washed glass fiber filters (0.45 µm pore size). After filtration, the solution collected from the three lysimeters was proportionally combined by plot prior to chemical analysis, yielding one sample per sampling day.

Table E1. Chemical and physical properties of the organic layer under the three forest types.

	thickness /cm	pH	C	N	S	P	C/N	C/P	N/P	Ca	Mg	K	Na
			______ g kg^{-1} ________							______ g kg^{-1} ________			
Natural forest	3.7	7.0	390	16	1.67	0.8	25	501	20	21	3.5	3.6	0.54
Eucalyptus	2.8	6.3	387	17	1.7	1.13	26	450	16	23	3.20	4.26	0.63
Cupressus	4.3	5.2	451	8.7	1.13	0.72	53	694	13	19	1.93	1.65	0.28

2.3. Chemical analysis

Samples were analysed for pH, total content of Ca, K, Mg, Na (plasma emission spectroscopy, ICP-AES, Integra XMP), and NO_3^-, NH_4^+, PO_4^{3-}, SO_4^{2-}, Cl^- (ion chromatography, Dionex 2000i-SP). Detection limits (mg l^{-1}) were: 0.025 for NH_4^+, 0.2 for Ca^{2+}, Na^+ and Mg^{2+}, 0.25 for K^+, 0.27 for Cl^-, 0.34 for NO_3^-, 0.28 for PO_4^{3-} and 0.32 for SO_4^{2-}.

2.4. Calculations and statistical analysis

Volume weighted concentrations were calculated for each season, i.e. dry season (DS) (October–January), small rainy season (RS1) (February–May) and main rainy season (RS2) (June–September). Volume Weighted Mean Concentration of the i-nutrient ($VWMC_i$) in forest floor leachate was estimated from the paired measurements of nutrient concentration and the volume of water flowing from the forest floor.

$$VWMC_i = \Sigma^n{}_{j=1}\, C_{ij} \,.\, Lf_j / \Sigma^n{}_{j=1}\, Lf_i \qquad (1)$$

where C_{ij} is the i-nutrient concentration in water collected on the j-collection day, Lf is the total water volume flowing through the forest floor and n is the total number of sampling dates. The seasonal fluxes of nutrients in forest floor leachate were calculated as the product of the VWMC of each nutrient and the cumulative water volumes measured for each season and summed to yield annual fluxes. Significance of differences among forest types of a given nutrient was assessed with two-way ANOVA using MSTAT-C statistical package. Correlation analysis was conducted between pairs of nutrients in forest floor leachate, and nutrient concentrations and forest floor leachate volume and pH.

3. Results and discussion

3.1. Nutrient concentrations

The mean annual nutrient concentrations in forest floor leachates of the three forest types are presented in Table E2. Nutrient concentrations in our study forests were generally higher than those reported for other montane tropical forests (Ecuador, Wilcke et al., 2001; Tanzania, Schrumpf, 2004), *Pinus* plantations in Brazil (Lilienfein et al., 2000, 2001) and from *Eucalyptus* plantations in Congo (Laclau et al., 2003a, b). Potassium and Mg concentrations in the higher altitudes of the Ecuadorian forests were somewhat equivalent to the values analysed in our study forests. In all three forest ecosystems under study, after K, Ca and Cl were the most abundant nutrients leached from the forest floor to the mineral soil. The concentration of PO_4–P was the least of all the nutrients followed by NH_4–N. The concentration of NO_3–N was about 10, 8 and 3 times higher than that of NH_4-N below the natural forest, *Eucalyptus* and *Cupressus*, respectively. The lower NH_4–N concentration relative to NO_3–N was probably a result of nitrification, vegetation uptake, adsorption (cation exchange) or assimilation by microbes.

In all forest types, forest floor leachates were enriched in almost all nutrients in relation to both precipitation and throughfall. However, the pattern was tree species and nutrient specific. For example, NH4–N was depleted in relation to rainfall in all forest types and also in relation to throughfall in the natural forest. The total nutrient composition of forest floor leachates below *Eucalyptus*, *Cupressus* and natural forest were about 3, 2 and 1.58 times, respectively, those of throughfall with the proportional level enrichment following the order: NO_3–N>Ca>Mg>PO_4–P>Cl>K≈Na>SO_4–S under the natural forest, NO_3–N> PO_4–P>Mg>Ca > K>Cl> NH_4–N >SO_4–S>Na under *Eucalyptus* and NO_3–N> Ca >PO_4–P>Mg> Cl>K> Na >NH_4–N >SO_4–S under *Cupressus*. Forest floor leachates under the two plantations were more enriched relative to throughfall than under the natural forest. Nitrate-N was enriched by

325% under the natural forest, by 494% under *Cupressus* and by 514% under *Eucalyptus*. The increase in NO_3–N concentrations in forest floor leachate, vis-à-vis throughfall, attests to considerable nitrifier activity. The results may also imply that nitrogen is not a limiting nutrient in our study site. It has been suggested that in the presence of abundant NH_4–N or organic N, nitrate is rejected even by microorganisms as N source (Joergensen and Meyer, 1990). Calcium enrichment ranged from 173% under the natural forest to 270% under *Cupressus* and Mg from 109% under the natural forest to 128% under *Cupressus* (Table E2). In forest soils, the forest floor horizons generally are characterised by intense microbial activity, high mineralization and accelerated nutrient release (Foster, 1985). Nutrient concentrations in forest floor leachates are not only affected by mineralization and plant uptake, but also by the amounts of water passing through the forest floor as a result of water uptake by roots and evaporation. Water flowing through the forest floor was higher under *Eucalyptus* than under the other two forest types (data not shown), but the lower amount of water under the natural forest and *Cupressus* did not cause a proportional increase in nutrient concentrations in the forest floor leachate probably due to root uptake and microbial immobilisation. Schrumpf (2004) in Tanzania also reported increased nutrient concentrations in forest floor leachates relative to throughfall with the exception of K which was lower than in throughfall. Tobon et al. (2004) and Laclau et al. (2003a, b) reported depletion of litter leachate nutrient concentrations relative to throughfall for Colombian Amazonia forests and *Eucalyptus* plantation in Congo, respectively. Wilcke et al. (2001) in Ecuador reported an increase in litter leachate Ca, Mg, NO_3–N, Na and total S concentrations relative to throughfall. The results of the later authors were not consistent on NH_4–N, K and total P concentrations.

Table E2. Volume weighted mean nutrient concentrations (mg l^{-1}) and pH in forest floor leachates during the dry and two rainy seasons and mean annual solute concentration changes (CC) relative to throughfall (litter leachate minus throughfall).

Forest/season	Ca	K	Mg	Na	Cl	NH_4–N	NO_3–N	SO_4–S	PO_4–P	pH
Natural forest										
DS	19.0 (2.3)	43.1 (11)	6.72 (1.0)	3.36 (0.3)	21.0 (3.0)	0.97 (0.5)	9.28 (1.0)	2.74 (0.1)	0.87 (0.4)	7.1 (0.3)
RS1	11.8 (1.7)	28.5 (2.4)	4.14 (0.4)	5.57 (1.9)	11.7 (1.0)	0.87 (0.2)	7.28 (0.5)	2.60 (0.2)	0.41 (0.1)	7.1 (0.2)
RS2	9.67 (2.2)	8.96 (0.3)	3.07 (0.6)	5.74 (0.3)	4.25 (0.4)	0.18 (0.02)	2.97 (0.5)	2.59 (0.1)	0.06 (0.04)	7.1 (0.5)
Mean	13.5 (1.8)	26.8 **b**(5.9)	4.65 **a**(0.6)	4.89 (0.7)	12.3 (2.6)	0.67(0.2)	6.51**a**(1.0)	2.64 (0.1)	0.45 (0.2)	7.1 (0.4)
CC	8.56	5.60	2.42	1.00	3.65	−0.22	4.98	0.38	0.17	-0.7
Eucalyptus										
DS	12.7 (0.4)	42.3 (7.6)	3.44 (0.1)	5.76 (0.4)	16.7 (2.4)	0.46 (0.1)	5.67 (0.9)	2.51 (0.1)	0.44 (0.06)	6.0 (0.1)
RS1	19.6 (5.7)	54.7 (17)	5.78 (1.7)	6.92 (1.8)	27.2 (8.0)	1.48 (0.6)	9.44 (3.5)	3.93 (0.8)	0.55 (0.07)	6.9 (0.5)
RS2	6.71 (0.2)	8.36 (1.5)	1.82 (0.1)	5.30 (0.3)	4.80 (0.5)	0.19 (0.0)	1.10 (0.1)	2.27 (0.2)	0.02 (0.02)	6.5 (0.2)
Mean	13.0 (2.5)	35.1 **a**(8.8)	3.68 **a**(6.0)	6.00 (0.6)	16.2 (4.0)	0.71 (0.3)	5.40 **a**(1.6)	2.90 (0.4)	0.33 (0.08)	6.6 (0.2)
CC	9.21	16.5	3.45	1.32	7.34	0.26	5.45	1.01	0.38	0.07
Cupressus										
DS	20.1 (3.3)	21.2 (1.9)	3.07 (0.4)	4.08 (1.5)	17.3 (1.8)	2.08 (0.5)	2.45 (1.0)	2.70 (0.8)	0.26 (0.12)	7.0 (0.5)
RS1	16.5 (2.3)	23.1 (1.6)	2.66 (0.2)	4.50 (0.6)	15.2 (0.9)	0.89 (0.3)	2.58 (1.1)	2.73 (0.4)	0.26 (0.1)	7.1 (0.2)
RS2	10.3 (0.4)	7.94 (0.7)	1.88 (0.1)	6.86 (0.4)	4.87 (0.5)	0.55 (0.0)	4.42 (0.7)	3.30 (0.3)	0.15 (0.03)	7.2 (0.4)
Mean	16.7 (1.8)	17.4 **c**(2.5)	2.53 **b**(0.29	5.15 (2.0)	12.5 (2.0)	1.17 (0.3)	3.15 **b**(0.6)	2.91 (0.3)	0.22 (0.05)	7.1 (0.3)
CC	12.2	7.00	1.42	1.76	6.06	0.34	2.62	0.81	0.16	0
Tanzania[1]	0.8	1.4	0.3	0.8	–	0.3	0.9	–	–	–
Colombia[2]	0.1–0.2	0.2–0.3	0.06–0.1	0.25–0.3	0.2–0.4	0.2–0.30	0.15–0.2	0.6–0.9	0.004–0.01	
Congo[3]	0.32	0.12	0.30	1.00	0.74	0.10	0.05	0.35	0.09	4.3
Ecuador[4]	2.3–8	4–28	1.4–5	1.3–1.5	0.83–4.4	0.25–0.71	0.6–5.7	–	–	4.8–7
Brazil[5]	0.1–0.3	0.2–1.3	0.03–0.13	0.26–2.25	–	0.97	0.75	–	–	–

DS-dry season, RS1-small rainy season, RS2-main rainy season.
Source : [1]Schrumpf (2004), [2]Tobon et al. (2004), [3]Laclau et al. (2003a, b), [4]Wilcke et al. (2001); [5]Lilienfein et al. (2000, 2001).
In a column means followed by the same lower case letters are not different. Values in parentheses are standard errors (n=3).

The pH of the forest floor leachate under the natural forest and *Eucalyptus* was lower by a unit of 0.7 in comparison to that in throughfall, while pH in the *Cupressus* plantation forest floor leachate remained unchanged. In forest soils acids are produced in many ways (Ulrich, 1980; Nilsson et al., 1982). For example, H ions are excreted from tree roots in response to cation uptake or organic acids are released from decomposing plant litter. Microbial mineralization of N and S in soil can also lead to SO_4 and NO_3 formation by acid producing reactions (Foster, 1985).

Table E3. Correlation matrices between litter leachate volume and nutrient concentrations in the organic layers of the three forest types.

Water volume	Ca	Mg	K	Na	Cl	NH_4–N	NO_3–N	PO_4–P	SO_4–S
Overall	0.38*	ns	ns	ns	ns	ns	0.32*	0.36*	ns
Natural forest	0.85**	0.81**	0.74*	ns	0.89***	ns	0.85**	ns	ns
Eucalyptus	ns	ns	ns	ns	ns	ns	ns	ns	ns
Cupressus	ns	ns	ns	ns	ns	ns	ns	ns	ns

*** P<0.001; ** P<0.01; * P<0.05; ns-not significant.

Water passing through the forest floors of the ecosystems under study did not differ in concentration of most nutrients except for K, Mg and NO_3–N (Table E2). Also, the interaction of forest type by season was not significant for any of the nutrients. Nitrate–N and Mg were significantly highest under the natural forest and lowest under *Cupressus*. Potassium was highest under *Eucalyptus* and lowest under *Cupressus*, while Mg and NO_3–N were slightly but not significantly higher below the natural forest than below *Eucalyptus*. Whereas, although statistically not significant, NH_4–N was slightly higher under *Cupressus* in comparison with the other two forest types, indicating lacking ammonium uptake or absence of an active nitrifying community. The low C/N ratios in the forest floors of *Eucalyptus* and natural forest might have triggered nitrification in comparison to *Cupressus* which had high C/N ratio in the organic layer (Table E1). Soils with C/N ratios >25-30 and low nutrient concentrations are reported to be poor-nitrifying (Gundersen and Rasmussen, 1990; Gundersen et al., 1998).

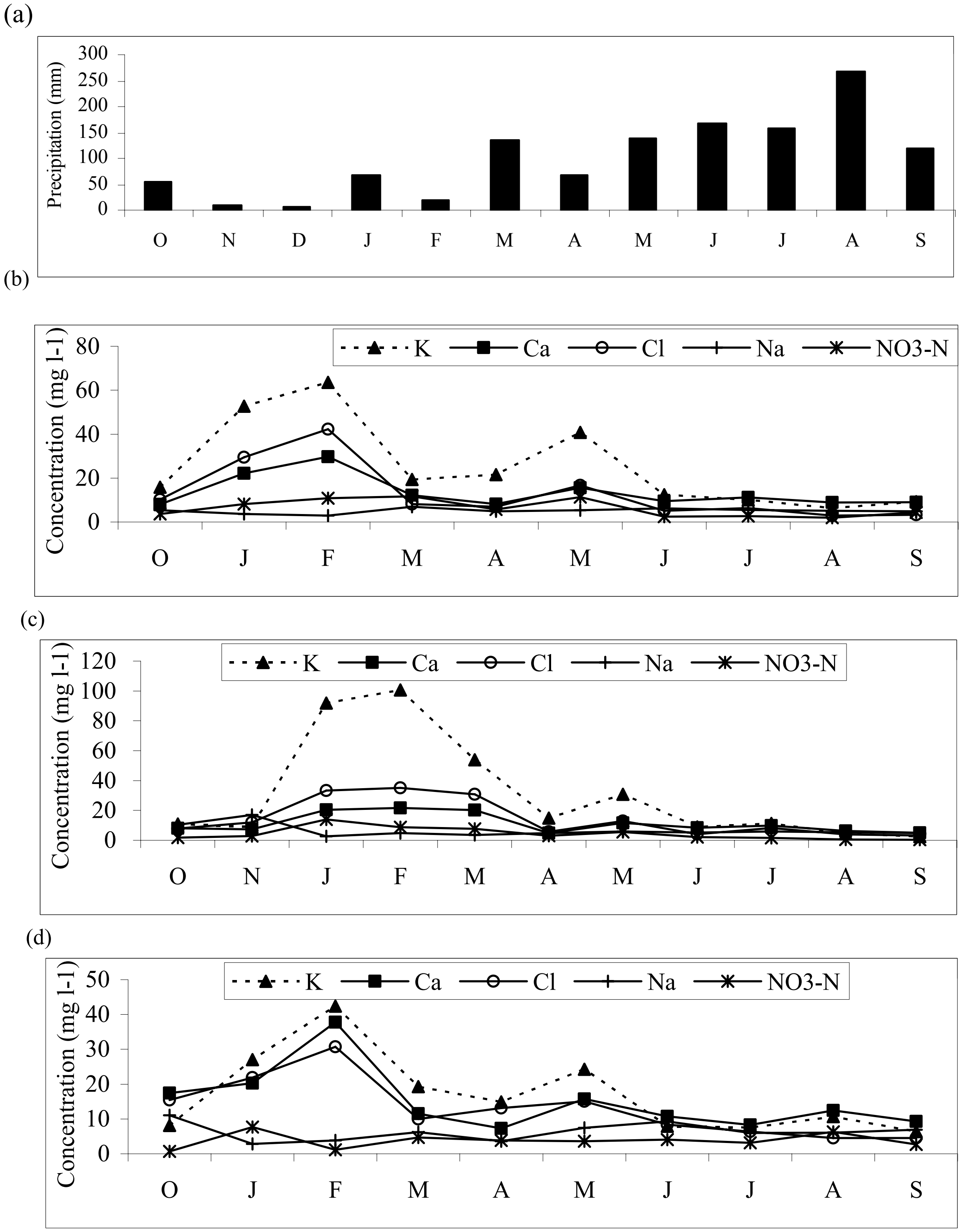
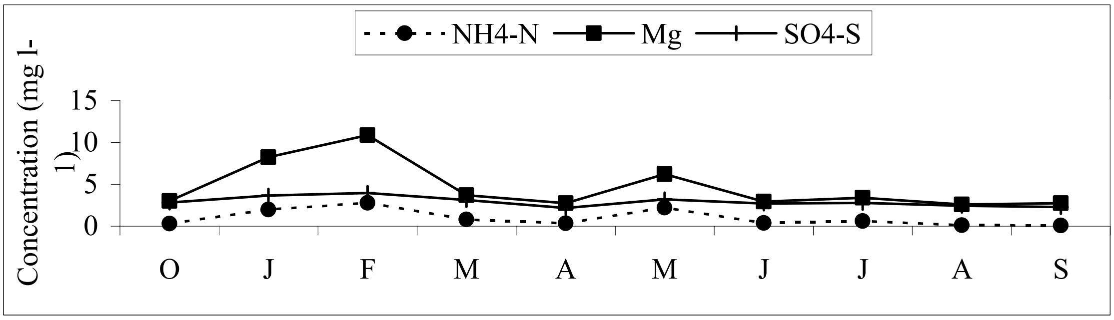
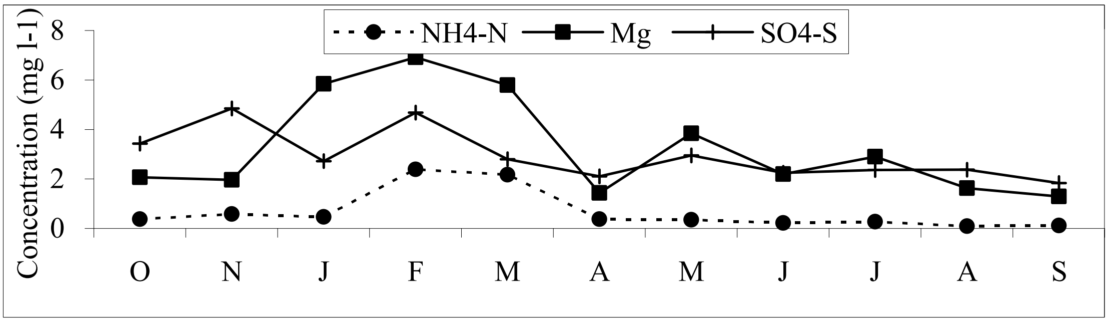


Fig.E1. Monthly precipitation (a) and examples of the monthly patterns of nutrient concentrations in forest floor leachates below the natural forest (b), *Eucalyptus* plantation (c) and *Cupressus* plantation (d).

(e)



(f)



(g)

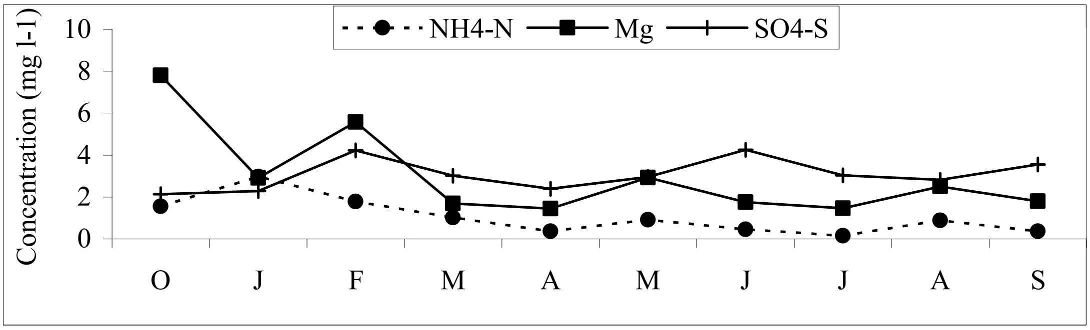


Fig. E1. Continued. Natural forest (e), *Eucalyptus* plantation (f) and *Cupressus* plantation (g).

Nearly no significant correlations between nutrient concentrations and forest floor leachate volume were observed; only in the natural forest Ca, Mg, K, Cl and NO_3–N revealed a positive significant relationship with the water volume (Table E3). None of the correlations between nutrient pairs, and nutrients and pH were significant in either of the forest types. The seasonal patterns of nutrient concentrations (Table E2) in the different forest types were variable. In the natural forest, except for Na which showed an increasing trend from the dry season to the wet season, all other nutrients decreased sharply from the dry season to the wet season with increasing rainfall amount and intensity. In the *Eucalyptus* plantation, nutrient concentrations were highest in the small rainy season and least in the main rainy season. This could be due to a lag in nutrient leaching during the dry season when low-volume intermittent rains reach the forest floor to permit mineralization but being insufficient to percolate vertically and transport nutrients to the mineral soil before the small rainy season. Stand density of the *Eucalyptus* plantation was very low compared to the other two forest types and consequently there had been throughfall water even during small rain events when there was no water under the other two forest types. In *Cupressus* plantation, the trend followed that of the natural forest except for K, NO_3–N, and SO_4–S, for which the concentrations were higher in the small rainy season compared to the dry season. The monthly nutrient dynamics (Fig.1b–g) show a clear variation among the different months; high concentrations during the dormant period (dry and small rainy season) and low concentrations during the vegetative period (main rainy season) probably due to high nutrient uptake by roots.

3.2. Nutrient fluxes

Nutrient fluxes to the mineral soil were not significantly different among forest types, but were slightly higher under *Eucalyptus* than below the other two forest types (Table E4). In general, large fluxes were observed for K, Ca, Cl, Na and NO_3–N. Measured nutrient exports to the mineral soil decreased in relation to throughfall fluxes for most of the nutrients indicating that nutrients deposited from throughfall as well as those released from decomposition are effectively taken up by plant roots or are immobilised. The nutrients that decreased most were K≈NH_4–N> Cl> SO_4–S in the natural forest, Na>SO_4–S>Mg>K in the *Eucalyptus* plantation and NH_4–N > SO_4–S>Na>Ca in the *Cupressus* plantation. Nitrate–N exports from the forest floor exceeded the inputs via throughfall by about 161% under *Cupressus* and 70% under the natural forest and *Eucalyptus* (Table D5 & Table E4).

Table E4. Seasonal nutrient fluxes (kg ha^{-1} season^{-1}) from the forest floor to the mineral soil under natural forest and below *Eucalyptus* and *Cupressus* in south-eastern Ethiopia.

Forest/Season	Ca	K	Mg	Na	Cl	NH_4–N	NO_3–N	SO_4–S	PO_4–P
Natural forest									
DS	0.32	0.72	0.11	0.06	0.35	0.02	0.15	0.05	0.01
	(0.12)	(0.45)	(0.06)	(0.01)	(0.17)	(0.01)	(0.06)	(0.01)	(0.001)
RS1	0.77	1.85	0.27	0.36	0.76	0.06	0.47	0.17	0.03
	(0.04)	(0.14)	(0.00)	(0.14)	(0.03)	(0.01)	(0.05)	(0.03)	(0.00)
RS2	0.96	0.89	0.31	0.57	0.42	0.02	0.30	0.26	0.01
	(0.15)	(0.05)	(0.04)	(0.05)	(0.02)	(0.00)	(0.04)	(0.02)	(0.00)
Total	2.05	3.46	0.69	0.99	1.53	0.09	0.92	0.47	0.05
Eucalyptus									
DS	0.50	1.68	0.14	0.23	0.66	0.02	0.23	0.10	0.02
	(0.04)	(0.31)	(0.02)	(0.02)	(0.09)	(0.00)	(0.02)	(0.00)	(0.00)
RS1	1.46	4.07	0.43	0.51	2.02	0.11	0.70	0.29	0.04
	(0.54)	(1.56)	(0.16)	(0.17)	(0.82)	(0.06)	(0.30)	(0.08)	(0.01)
RS2	0.68	0.85	0.18	0.54	0.49	0.02	0.11	0.23	0.002
	(0.13)	(0.20)	(0.04)	(0.10)	(0.10)	(0.00)	(0.01)	(0.04)	(0.00)
Total	2.64	6.60	0.75	1.28	3.17	0.15	1.04	0.62	0.06
Cupressus									
DS	0.48	0.50	0.07	0.10	0.41	0.05	0.06	0.06	0.01
	(0.10)	(0.16)	(0.02)	(0.02)	(0.10)	(0.02)	(0.01)	(0.01)	(0.00)
RS1	1.40	1.96	0.23	0.38	1.29	0.08	0.22	0.23	0.02
	(0.29)	(0.66)	(0.06)	(0.06)	(0.37)	(0.06)	(0.02)	(0.05)	(0.00)
RS2	0.98	0.75	0.18	0.65	0.46	0.05	0.42	0.31	0.01
	(0.33)	(0.27)	(0.06)	(0.02)	(0.18)	(0.02)	(0.1)	(0.09)	(0.00)
Total	2.86	3.21	0.48	1.13	2.16	0.18	0.60	0.60	0.04

DS-dry season, RS1-small rainy season, RS2-main rainy season. Values in parentheses are standard errors (n=3).

Calcium and PO_4–P exports by leaching out of the *Cupressus* forest floor were 34% and 33% higher than the corresponding throughfall inputs; below *Eucalyptus* as high as 50% more PO_4–P was exported in comparison to the input by throughfall. The temporal patterns in fluxes appeared to be nutrient and tree species specific. In the *Eucalyptus* plantation, except for Na which increased sharply from the dry season to the wet season, the fluxes of all other nutrients were highest in the small rainy season and least in the dry season. In the natural forest and in the *Cupressus* plantation, the seasonal changes in nutrient fluxes showed the following: Ca, Mg, Na and SO_4–S in the natural forest and Na, NO_3–N and SO_4–S in the

Cupressus plantation increased with increasing rainfall amount and intensity, while the fluxes of the other nutrients followed the trend below *Eucalyptus*.

4. Conclusion

The results of this study showed that in general differences in the forest cover did not influence the concentration of most nutrients in forest floor leachates. However, Mg and NO_3-N concentrations below *Cupressus* were significantly lower than under *Eucalyptus* and under the natural forest. The concentration of K was significantly higher below *Eucalyptus* than below the natural forest and *Cupressus*. In all forest types, K was the most abundant nutrient followed by Ca and Cl, while PO_4-P followed by NH_4-N was the least. Almost all of the nutrients in the forest floor leachates of the three forest types were enriched in relation to throughfall, being even higher in the two plantations. Calcium, Mg, NO_3–N and PO_4–P were the most enriched. Nearly no significant correlations between nutrient concentrations and forest floor leachate volume were observed in the two plantation forest ecosystems, while in the natural forest some significant correlations were evident. The annual flux of K was larger than any other nutrient. Measured nutrient exports to the mineral soil decreased in relation to throughfall fluxes for most of the nutrients. The nutrients that decreased most were SO_4–S, Na, K, Cl and NH_4–N. However, the export of NO_3–N from the forest floor exceeded the inputs via throughfall by about 161% under *Cupressus* and 70% under the natural forest and *Eucalyptus*. Overall, the results show that despite differences in tree species composition among the three forest types, the organic layer in all forest types acted as a sink for most of the nutrients deposited by throughfall and being released by the decomposition of organic matter.

Acknowledgements

We thank the German Research Foundation (DFG) for its support of the research through grant number Gu406–8.

References

Ashagrie, Y., Zech, W., Guggenberger, G., Teketay, D., 2003. Changes in soil organic carbon, nitrogen and sulphur stocks due to the conversion of natural forest into tree plantations (*Pinus patula* and *Eucalyptus globulus*) in the highlands of Ethiopia. World Resour. Rev. 15, 462–482.

Berg, B., Berg, M.P.,Bottner, P., Box, E., Breymeyer, A., Calvo de Anta R., Couteaux, M., Escudero, A., Gallardo, A., Kratz, W., Madeira, M., Mällkönen, E., McClaugherty, V., Munoz, F., Piussi, P., Remacle, J., and Virzo de Santo, A., 1993. Litter mass loss rates in pine forests of Europe and Eastern United States: Some relationships with climate and litter quality. Biogeochemistry 20, 127-159.

Berg, M.P., Kniese, J.P., Zoomer, R., Verhoef, H.A., 1998. Long-term decomposition of successive organic strata in a nitrogen saturated Scots pine forest soil. For. Ecol. Manage. 107, 159-172.

FAO, 1997. FAO/UNESCO Soil map of the world, revised legened with corrections and updates. World soil resources, Report No. 60, ISRIC, Wageningen, The Netherlands.

Foster, N.W., 1985. Acid precipitation and soil solution chemistry within a maple–birch forest in Canada. For. Ecol. Manage. 12, 215–231.

Gundersen, P., Callesen, I., de Vries, W., 1998. Nitrate leaching in forest ecosystems is related to forest floor C/N ratios. Environ. Pollut. 102, 403–407.

Gundersen, P., Rasmussen, L., 1990. Nitrification in forest soils: effects of nitrogen deposition on soil acidification and aluminium release. Rev. Environ. Contamin. Toxicol. 13, 1–45.

Helmisaari, H-S., 1995. Nutrient cycling in *Pinus sylvestris* stands in eastern Finland. Plant Soil, 168-169, 327-336.

Joergensen, R.G., Meyer, B., 1990. Nutrient changes in decomposing beech leaf litter assessed using a solution flux approach. J. Soil Sci. 41, 279-293.

Johansson, M-B., Berg, B., Meentemeyer, V., 1995. Litter mass-loss rates in late stages of decomposition in a climatic transect of pine forests. Long-term decomposition in a Scots pine forest. IX. Can. J. Bot. 73, 1509-1521.

Kurka, A-M., Starr, M., Heikinheimo, M., Sabrinoja, M., 2000. Decomposition of cellulose strips in relation to climate, litterfall nitrogen, phosphorus and C/N ratio in natural boreal forests. Plant Soil 219, 91-101.

Laclau, J-P., Ranger, J., Bouillet, J-P., Nzila, J.D., Deleporte, P., 2003a. Nutrient cycling in a clonal stand of *Eucalyptus* and an adjacent savanna ecosystem in Congo. 1. Chemical composition of rainfall, throughfall and stemflow solutions. For. Ecol. Manage. 176, 105–119.

Laclau, J-P., Ranger, J., Nzila, J.D., Bouillet, J-P., Deleporte, P., 2003b. Nutrient cycling in a clonal stand of *Eucalyptus* and an adjacent savanna ecosystem in Congo. 2. Chemical composition of soil solutions. For. Ecol. Manage. 180, 527−544.

Lilienfein, J., Wilcke, W., Ayarza, M.A., Vilela, L., Lima, S.D.C., Zech, W., 2000. Soil acidification in *Pinus caribaea* forests on Brazilian savanna Oxisols. For. Ecol. Manage. 128, 145−157.

Lilienfein, J., Wilcke, W., Thomas, R., Vilela, L., Lima, S.D.C., Zech, W., 2001. Effects of *Pinus caribaea* forests on the C, N, P, and S status of Brazilian savanna Oxisols. For. Ecol. Manage. 147, 171−182.

Mahendrappa, M.K. 1990. Partitioning of rainwater and chemicals into throughfall and stemflow in different forest stands. For. Ecol. Manage. 30, 65-72.

Nilsson, S.I., Miller, H.G., Miller, J.D., 1982. Forest growth as a possible cause of soil and water acidification; an examination of the concepts. Oikos 39, 40−49.

Schrumpf, M., 2004. Biogeochemical investigations in old growth and disturbed forest sites at mount Kilimanjaro. Dissertation, PhD., Bayreuth University.

Solomon, D., Fritzsche, F., Tekalign, M., Lehmann, J., Zech, W., 2002. Soil organic matter composition in the sub humid Ethiopian highlands as influenced by deforestation and agricultural management. Soil Sci. Soci. Am. J. 66, 68–82.

Tobon, G., Sevink, J., Verstraten, J.M., 2004. Litterflow chemistry and nutrient uptake from the forest floor in northwest Amazonian forest ecosystems. Biogeochemistry 69, 315–339.

Ulrich, B., 1980. Production and consumption of hydrogen ions in the ecosphere. In: Hutchinson, T.C., Havas, M. (eds.), Effects of Acid Precipitation on Terrestrial Ecosystems. Plenum press, New York, pp.255–282.

Vogt, K.A., Grier, C.C., Vogt, D.J., 1986. Production, turnover and nutrient dynamics of above- and belowground detritus of the world forests, Adv. Ecol. Res. 15, 303–377.

Wells, C.G., Davey, C.B., 1966. Cation exchange characteristics of forest floor materials. Soil Sci. Soc. Am. Proc. 30, 399-402.

Wilcke, W., Yasin, S., Valarezo, C., Zech, W., 2001. Change in water quality during the passage through a tropical montane rain forest in Ecuador. Biogeochemistry 55, 45–72.

Wilcke, W., Yasin, S., Abramowski, U., Valarezo, C., Zech, W., 2002. Nutrient storage and turnover in organic layers under tropical montane rain forest in Ecuador. Eur. J. Soil Sci. 53, 15–27.

Youngberg, C.T., 1966. Forest floors in Douglas-fir forests: 1. dry weight and chemical properties. Soil Sci. Soc. Am. Proc. 30, 406-409.

Zhang, Q., Zark, J. C., 1995. Effects of gap size on litter decomposition and microbial activity in a subtropical forest. Ecology 76, 2196-2204.

F Geochemistry of inorganic nutrients in water percolating through the mineral soils under two exotic tree species plantations and an adjacent natural forest in south-east Ethiopia

Y. Ashagrie[1]

[1]*Institute of Soil Science and Soil Geography, University of Bayreuth, D-95440 Bayreuth, Germany*
Email : yeshanew@hotmail.com,Tel: ++49-921-55-21-46 Fax: ++49-921-55-22-46

Abstract

The dynamics of nutrients in water passing through the mineral soils under two exotic tree species plantations (*Cupressus lusitanica* and *Eucalyptus globulus*) and an adjacent *Podocarpus falcatus* dominated natural forest were examined during the main rainy season (June–September) in 2002 at Munesa, south-eastern Ethiopia. The soil solutions collected from the different stands were dominated by Ca, NO_3–N and Na in *Cupressus* plantation and by Na and Cl in the natural forest and by Ca in *Eucalyptus* plantation. The vertical patterns in median nutrient concentrations showed a decreasing trend for most of the nutrients under the natural forest and *Eucalyptus* plantation. The concentrations of Cl and Na in all forest types and Ca, Mg and NO_3–N below *Cupressus* increased with increasing soil depth. Nutrient concentrations showed a great variation between *Cupressus* on the one side, and the natural forest and *Eucalyptus* plantation on the other side, especially with respect to Ca, Mg and NO_3–N concentrations. The concentrations of Ca, Mg and NO_3–N below *Cupressus* were 7, 3.4 and 17 times higher than under the natural forest and 2, 2.4 and 4 times higher under *Eucalyptus*, suggesting that these nutrients under *Cupressus* are in excess of tree and microbial requirements. These variations among stands were mainly due to large differences in the subsoil (1 m soil depth) concentrations. The concentrations of Ca, Mg and NO_3–N at 1 m soil depth under the natural forest were 8, 7 and 23 times lower than under *Cupressus*. The corresponding figures under *Eucalyptus* were 3, 4 and 81 times lower than under *Cupressus*. These results suggest relatively tight nutrient cycling in the natural forest and *Eucalyptus* plantation. Potassium, Na, NH_4–N and SO_4–S concentrations were similar among the three

forest types. Overall, ecosystem-specific patterns of vegetation composition and associated demand for nutrients appear to control nutrient concentrations and rates of nutrient leaching in the forest ecosystems under study.

Key words: *Cupressus*, Ethiopia, *Eucalyptus*, nutrient leaching, plantation, soil solution

1. Introduction

In the last few decades large areas of forest plantations (*ca*. 200,000 ha), predominantly exotic species (*Eucalyptus* spp., *Cupressus lusitanica* and *Pinus* spp.) have been established in Ethiopia to satisfy the growing wood demands of the population and to rehabilitate degraded lands (EFAP, 1994). Also the fast growing nature of exotic species and favourable economic returns from forest plantations have encouraged the conversion of slow growing and low productive secondary natural forests into plantations. The conversion of natural forest ecosystems into monoculture plantations can change the nutrient cycling processes through changes in species composition. Changes in forest species composition influence the overall nutrient cycling owing to differential patterns among plant species in litter production and turnover and nutrient accumulation (Gosz, 1981; Lugo, 1992; Ashagrie et al., 2003), rooting (Alban, 1982) and canopy geometry and area (Mahendrappa,1990). Previous investigations on the effects of forest plantations on soil properties in Ethiopia have focused on changes to solid phase soil properties (Michelsen et al., 1993; Ashagrie et al., 2003; Lemenih et al., 2004). These studies generally indicate that the changes in soil properties after plantation establishment are species specific. However, up to know it remains unknown, how sustainable such forest plantations are?

Soil solution chemistry, which is an important aspect in studying nutrient cycling in forest ecosystems, is only beginning to be investigated in Ethiopia. In contrast to bulk soil properties, which typically react slowly on changes in land-use, soil solution chemistry is often a sensitive indicator of biogeochemical processes in forests, responding quickly to various changes and may provide an early indication of the long-term changes in soils associated with land use changes (Ranger et al., 2001; McDowell et al., 2004). Studies of solute concentrations and fluxes through forest ecosystems have been conducted mainly in North America (Likens et al., 1977) and Europe (Gundersen et al., 1998; de Vries et al., 2003). The chemistry of soil solution can change as it passes down through the soil profiles (Tokuchi et al., 1993). These changes reflect the biological and chemical processes which occur through the movement of soil solution within the soil. The objective of this work was to assess whether ecosystem-specific differences lead to differences in rates of nutrient retention and leaching at Munesa, south eastern Ethiopia.

2. Materials and methods

2.1. Study area

The study was conducted in the Munesa/Shashemene forest (7°34´N and 38°53´E) located some 240 km south east of Addis Ababa. The altitude of the study site is 2400 m. Precipitation is bimodal most of it falling in July and August. Mean annual precipitation and mean annual temperature of the study area are 1250 mm and 19 °C, respectively. The experimental design was set up in three stands situated side by side: an old growth *Podocarpus falcatus* dominated natural forest, and two exotic tree species plantations (*Eucalyptus globulus* and *Cupressus lusitanica*) established in 1980 after clearing of part of the natural forest. Within each forest type three replicated plots (20x30 m) were randomly located. The *Eucalyptus* plantation was sparsely stocked with 595 trees ha^{-1} and had native understorey canopy tree (*Croton macrostachys*) and shrubs notably *Acanthopale pubescens*, *Achyrospermum schimperi*, *Bothriocline schimperi*, *Carex spicato-paniculata*, *Hypoestes forskaolli*. The ground layer was covered with dense grass and broad-leaved herbaceous species. The mean height of *Eucalyptus* was 30–40 m and mean diameter at breast height (dbh) was 19−39 cm. The *Cupressus* plantation had a relatively dense standing stock (672 trees ha^{-1}) with almost no ground vegetation. The mean height of *Cupressus* was 18–20 m and dbh was 29 cm. Soil properties under the two plantations prior to their establishment were assumed to be similar to those under the natural forest. The soils of the study area were classified as Nitisols (FAO, 1997). Some selected chemical and physical properties of the soils under the three forest types are presented in Tables F1&F2.

2.2. Field equipment and soil solution sampling

In each of the nine plots an area of 20x30 m was fenced at the centre and porous cup ceramic tension lysimeters and tensiometers were installed at three depths (0.2, 0.5 and 1 m below soil surface). Tensiometers were placed approximately 0.5 m away from the suction lysimeters. In total three lysimeters and tensiometers were placed per depth. The three suction cups per depth and in each plot were connected to one collecting bottle. Lysimeters and tensiometers were installed in May 2001 and solution samples retrieved during the main rainy season (June to September 2001) were discarded to allow ions on the exchange surfaces of the ceramic to equilibrate with the soil solution. In 2002 solution samples were taken only during the main rainy season when water inputs exceeded soil water storage capacity and evapotranspiration. Sampling of soil solution was done every week by applying vacuum produced by vacuum pumps based on the tensiometer readings at each soil depth. After each collection, the collector bottles were washed with deionized water or with a portion of the sample water. On each sampling day, water samples were transported to the storage facility and kept frozen until they were transported in a cool box to Germany for chemical analysis. Samples were filtered through 0.45 μm glass fibre filters (Schleicher & Schuell) before analysis.

2.3. Chemical analysis

Soil organic C, N and S concentrations were determined using a CHNS−analyzer (Vario EL, Elementar Analysensysteme, GmbH, Hanau, Germany). Cation exchange capacity (CEC) was determined with 1 M NH_4OAc (pH=7.0) following the procedure of Hendershot et al., (1993). Dithionite–citrate–bicarbonate extractable aluminium and iron (Al_d, Fe_d) and oxalate-extractable aluminium and iron (Al_o, Fe_o) were determined according to Ross and Wang (1993).

Table F1. Some chemical and physical properties of the soils under the studied forests.

Horizons	Depth (cm)	pH_{KCl}	CEC $cmol_c\ kg^{-1}$ soil (1M NH_4 OAc)	Al_d	Fe_d	Al_0	Fe_0	Ca	K	Mg	Na	sand	silt	clay
				g kg^{-1} soil										
Natural Forest														
A	0–15	5.4	51.1	3.7	43	3.4	9.1	5.4	0.59	0.91	0.01	200	300	500
AB	15–29	5.3	32.8	4.6	55	4.5	7.6	0.5	0.25	0.29	nd	230	230	540
Bt1	29–68	4.7	30.6	4.7	58	4.5	7.2	0.2	0.23	0.19	0.01	80	180	740
Bt2	68–108	4.5	29.2	4.1	57	3.9	6.2	0.03	0.25	0.12	0.06	80	180	740
***Eucalyptus* plantation**														
A	0–10	5.3	37.2	3.8	43	3.5	8.6	4.3	0.71	0.53	0.02	140	370	490
AB	10–27	5.1	31.1	3.9	46	2.8	7.3	1.8	0.74	0.38	0.02	140	300	560
Bt1	27–69	4.8	34.2	3.6	51	2.6	6.7	1.8	0.34	0.27	0.03	100	240	660
Bt2	69–106+	4.7	32.2	3.5	53	2.4	6.2	1.9	0.32	0.29	0.01	100	170	730
***Cupressus* plantation**														
A	0–25	5.6	36.7	4.1	48	3.1	10	3.3	0.53	0.56	0.01	90	270	640
AB	25–41	5.1	33.9	4.3	55	3.0	9.4	3.3	0.67	0.50	nd	140	360	500
Bt1	41–81	4.8	32.5	4.4	58	3.3	7.4	3.0	0.69	0.47	nd	60	170	770
Bt2	81–105+	4.6	31.7	4.9	59	3.6	6.0	2.8	0.37	0.45	nd	60	310	630

Al_d and Fe_d: Dithionite–citrate–bicarbonate extractable aluminium and iron.
Al_o and Fe_o: Oxalate-extractable aluminium and iron.
nd: not detectable

Table F2. SOC, N and S concentrations (g kg $^{-1}$) and C/N ratios at different soil depths under the three forest types.

	0–20	20–40	40–70	70–100
Natural forest				
SOC	61.3	32.3	28.8	17.4
TN	5.2	2.1	2.4	1.7
TS	0.6	0.4	0.4	0.2
C/N	12	11	11	9
Cupressus				
SOC	65.2	20.9	17.0	16.1
TN	6.5	2.1	1.8	1.7
TS	0.7	0.2	0.2	0.2
C/N	10	10	9	9
Eucalyptus				
SOC	59.3	27.7	17.0	12.9
TN	3.5	2.7	1.8	1.8
TS	0.6	0.5	0.3	0.3
C/N	9	9	9	9

SOC: soil organic carbon; TN: total nitrogen; TS: total sulphur

Total contents of Ca, K, Mg, Na were determined by plasma emission spectroscopy, ICP-AES, Integra XMP, Cl^-, NO_3^-, NH_4^+, PO_4^{3-}, SO_4^{2-} by ion chromatography, Dionex 2000i-SP and pH (Orion U402-S7). Detection limits (mg l^{-1}) were: 0.025 for NH_4^+, 0.2 for Ca^{2+}, Na^+ and Mg^{2+}, 0.25 for K^+, 0.27 for Cl^-, 0.34 for NO_3^-, 0.28 for PO_4^{3-} and 0.32 for SO_4^{2-}.

2.4. Data analysis

Random missing values occurred throughout the study due to variations in water input and evapotranspiration under the different stands, variations in micro relief of the different plots within a stand, and equipment failure. In addition, too little sample water volume in some plots did not allow measurement of all elements. Furthermore, the concentration data for each element was generally highly variable. Therefore, statistical analysis based on mean values can not be justified; instead median values were used.

3. Results and discussion

Table F3 informs about the ranges of minimum and maximum values for pH and nutrient concentrations at different soil depths. The concentration of PO_4–P was generally below the detection limit in all the three forest types. Phosphate is relatively insoluble and readily fixed by soil colloids (Brady and Weil, 1999). Potassium and NH_4–N were also often below the detection limit in almost all forest types. Similar results have been reported by Lilienfein et al. (2001) in Brazilian Oxisols under *Cerrado* and *Pinus* plantation. Magnesium and NO_3–N under the natural forest and *Eucalyptus* plantation at all depths, and NO_3–N below *Cupressus* at 0.5 m depth had minimum values less than one. Low concentrations of K, Mg and NH_4–N and NO_3–N in soil solutions probably can be attributed to the high biological demand for these nutrients in the forests under study. The median mean concentrations of nutrients (Table F4) in the soil solution below the studied forests were in the order: Na>Cl>Ca>SO_4–S>Mg>NO_3–N>K>NH_4–N under the natural forest, Na>Ca> SO_4–S>Cl>Mg> NO_3–N>K> NH_4–N under *Eucalyptus* and NO_3–N>Ca>Na>Cl>Mg>SO_4–S>K> NH_4–N under *Cupressus*.

With the exceptions of Mg, Na and NO_3–N at all depths under *Cupressus*, Na at all depths and SO_4–S at 0.2 m depth below *Eucalyptus*, and Na and SO_4–S at the 0.2 m depth under the natural forest which increased in relation to those in the forest floor leachate, the concentrations of all other elements decreased in relation to the concentrations in the forest floor leachate (Paper E). Of the nutrients, especially K, Mg, NH_4–N and PO_4–P decreased to a great degree, indicating root uptake and adsorption by the exchange complex. Decreases in nutrient concentration from the organic layer to the mineral soils were more pronounced under the natural forest and *Eucalyptus* than under *Cupressus*. An increase of the Mg concentration in the mineral soil solution down to 2 m and of the NO_3–N concentration down to 1.2 m soil depth relative to forest floor lecheate under *Pinus* plantation were also reported by Lilienfein et al. (2000, 2001). The later authors reported a decrease in forest floor lecheate Ca, K, Na and NH_4–N concentrations in relation to the concentrations in the mineral soil. Laclau et al. (2003) in Congo described an increase in NH_4–N, NO_3–N and SO_4–S concentrations and a decrease in Ca, K, Mg, Na and PO_4–P concentrations in the forest floor leachate in relation to the concentrations in the mineral soil under *Eucalyptus* plantation. Schrumpf (2004) observed a decrease in Ca, K, Mg, Na and NH_4–N concentrations and an increase in NO_3–N concentration in the mineral soil solution in relation to the forest floor leachate below a tropical montane forest at south-west exposed slopes of Mount Kilimanjaro. Below the natural forest and *Eucalyptus*, mineral soil solution pH was higher than the forest floor leachate at all depths while under *Cupressus* the reverse holds true except at 1 m depth where the mineral soil solution had higher pH value than the forest floor leachate. Increased pH values at different soil depths in relation to the forest floor have been reported by Laclau et al. (2003).

Table F3. Ranges of nutrient concentrations (mg l^{-1}) and pH in soil solution at different soil depths under the three forest types.

Element	depth (m)	Natural forest	*Eucalyptus*	*Cupressus*
pH	0.2	6–8.3	6.6–7.6	5.7–7.7
	0.5	6.6–7.8	6.9–8.3	5.2–7.2
	1.0	6.9–7.7	6.7–8.3	6.7–7.7
Ca	0.2	1.6–24.5	3.9–20.3	8.4–49.3
	0.5	1.0–2.8	2.1–11.5	3.8–16.2
	1.0	1.8–7.7	1.9–5.0	11.1–16.2
Cl	0.2	2.0–9.4	1.3–8.8	2.0–6.3
	0.5	2.0–6.5	0.8–8.7	2.4–11.8
	1.0	2.9–9.5	1.7–5.3	2.8–9.2
K	0.2	nd–8.5	0.4–11.5	1.2–17.2
	0.5	nd–3.4	nd–29.6	nd–5.7
	1.0	2.4–20.2	nd–3.9	nd–0.4
Mg	0.2	1.1–7.3	0.9–6.4	3.1–16.6
	0.5	0.6–3.0	0.4–4.1	2.3–5.9
	1.0	0.6–3.0	0.6–1.5	4.0–5.5
Na	0.2	4.2–6.6	5.0–8.8	5.1–8.7
	0.5	3.4–8.8	2.4–7.9	4.1–7.5
	1.0	2.1–6.3	5.1–13.5	5.6–10.4
NH_4–N	0.2	nd–0.7	nd–0.1	nd–0.6
	0.5	nd–0.4	nd–0.4	nd–0.1
	1.0	nd–0.1	nd–0.1	nd–0.1
NO_3–N	0.2	0.2–1.5	0.3–3.8	10.6–58.5
	0.5	0–1.6	0.9–11.5	0.1–21.5
	1.0	0.5–1.1	nd–3.9	8.6–15.9
SO_4–S	0.2	1.7–4.0	2.5–5.4	1.9–5.8
	0.5	2.0–3.7	1.0–3.9	1.7–3.7
	1.0	0.8–3.0	1.8–4.6	1–2.3

nd: not detected

Table F4. Median nutrient concentrations (mg l^{-1}) and pH at different soil depths under the three forest types.

Element	Depth (m)	Natural forest	*Eucalyptus*	*Cupressus*
pH	0.2	7.2	7.1	6.7
	0.5	7.2	7.0	6.6
	1.0	7.2	7.3	7.3
Mean		**7.2**	**7.1**	**6.9**
Ca	0.2	3.41	9.99	12.5
	0.5	3.27	4.60	9.35
	1.0	1.62	3.84	12.8
Mean		**2.77**	**5.81**	**11.6**
Cl	0.2	2.75	1.99	2.87
	0.5	3.94	2.27	6.77
	1.0	5.76	2.52	6.96
Mean		**4.15**	**2.26**	**5.53**
K	0.2	1.13	2.01	4.49
	0.5	nd	1.36	nd
	1.0	nd	0.42	nd
Mean		**0.38**	**1.26**	**1.50**
Mg	0.2	2.20	3.19	5.02
	0.5	1.18	1.31	3.74
	1.0	0.65	1.12	4.76
Mean		**1.34**	**1.87**	**4.51**
Na	0.2	5.33	6.18	5.92
	0.5	4.71	6.60	6.16
	1.0	4.73	6.81	7.14
Mean		**4.92**	**6.53**	**6.41**
NH_4–N	0.2	0.08	0.03	0.08
	0.5	0.03	0.03	0.03
	1.0	0.03	nd	0.04
Mean		**0.05**	**0.02**	**0.05**
NO_3–N	0.2	1.05	4.17	17.7
	0.5	0.78	1.05	9.28
	1.0	0.57	0.16	12.9
Mean		**0.8**	**1.79**	**13.3**
SO_4–S	0.2	3.45	3.83	2.87
	0.5	2.26	2.46	2.62
	1.0	1.70	2.45	1.40
Mean		**2.47**	**2.91**	**2.30**

nd: not detected

The vertical patterns of the median K, NH_4–N and SO_4–S concentrations in soil solution below all forest types and median Ca, Mg and NO_3–N concentrations under *Eucalyptus* plantation and the natural forest decreased steadily with increasing soil depth, presumably due to adsorption by the soil colloid or to plant and microbial uptake (Table F4). In contrast, under the *Cupressus* plantation, the concentrations of Ca, Mg and NO_3–N decreased from 0.2 m depth to 0.5 m depth and then increased at the depth of 1 m. This pattern appears to follow the root distribution and concurrent nutrient uptake as the roots of *Cupressus* are confined to the surface 0.5 m (Ashagrie, pers. observation). Whereas, increased concentrations of these nutrients at the depth of 1 m indicates losses by leaching. The degrees of Ca and NO_3–N leaching were higher than that of Mg leaching (Table F4). Nitrate is a very mobile anion in the soil and its adsorption is small and, if not taken up by plants or microflora, leaching occurs during periods of excess precipitation (Gundersen and Rasmussen, 1995; Rasmussen, 1998).

Increased concentrations of Ca and Mg in the subsoil (1 m depth) under *Cupressus* can be attributed to the combination of increased anion availability for leaching, increased H^+ generated from nitrification displacing cations on exchange sites, and/or limited plant uptake (Foster, 1985). The increase in Cl and Na concentrations with depth (Table F4) may indicate that these nutrients are mobile and biotic demand for them is low. In a similar study in Tanzania, Schrumpf (2004) reported a decrease in mean Ca, Mg, Na, NO_3–N concentrations with depth. Her results show no clear trend in the K and NH_4–N concentrations with increasing soil depth. Lilienfein et al. (2000, 2001) reported an increase in Ca, K, Mg and Na concentrations between 0.3 and 2 m mineral soil depth under *Cerrado* vegetation and *Pinus* plantation and an increase in NO_3–N concentration under *Pinus* but a slight decrease under *Cerrado*. An increase in Ca, Mg and NO_3–N concentrations and a decrease in K and SO_4–S concentrations with increasing soil depth have also been reported in a coniferous forest in Japan (Tokuchi et al., 1993). The pH of the soil solution below the natural forest under study was the same at all depths, but was slightly different at different depths under the two plantations (Table F4).

In each forest type, regardless of soil depth, there were statistically significant correlations between the major nutrient elements (Table F5). In the *Eucalyptus* plantation, SO_4–S and Ca, Mg and Na correlated significantly. Very few significant correlations were observed between other nutrients, and Cl and NO_3–N. In the natural forest and *Cupressus* plantation, most of the metal elements were correlated with Cl, NO_3–N and SO_4–S, but the relationships with NO_3–N in the natural forest and Cl in *Cupressus* were negative. The positive significant correlations between metallic elements and Cl, NO_3–N and SO_4–S indicate that cations were moved with the chloride, nitrate and sulphate anions. In *Cupressus* plantation, most of the correlations between elements were stronger than below the natural forest and *Eucalyptus*. The correlations between metallic elements and NO_3–N were stronger than the correlations with SO_4–S and Cl. The correlations between sulphate and cations were generally weaker than between nitrate and cations (Ranger et al., 2001). Correlations between nutrient concentrations and pH were not significant under *Cupressus* and *Eucalyptus* while below the natural forest few significant correlations were observed.

Median mean nutrient concentrations among forest types (Table F4) generally showed differences in nutrient retention capacity of the different forest ecosystems. Overall, the variations in median mean nutrient concentrations were higher between *Cupressus* on the one hand and the natural forest and *Eucalyptus* plantation on the other than between the later two.

Table F5. Correlation matrices between nutrient concentrations, and between nutrient concentrations and pH.

	Ca	Cl	K	Mg	Na	NH_4–N	NO_3–N	SO_4–S	pH
Natural forest									
Ca	–	ns	ns	0.93*	ns	0.37*	-0.46**	0.39*	0.31*
Cl	–	–	0.34*	ns	0.31*	ns	-0.37*	ns	ns
K	–	–	–	ns	ns	ns	ns	-0.31*	ns
Mg	–	–	–	–	0.34*	0.40*	-0.53*	0.51*	0.29*
Na	–	–	–	–	–	ns	-0.44**	0.56*	ns
NH_4–N	–	–	–	–	–	–	ns	0.42*	0.39*
NO_3–N	–	–	–	–	–	–	–	-0.28*	0.36*
SO_4–S	–	–	–	–	–	–	–	–	ns
pH	–	–	–	–	–	–	–	–	–
Eucalyptus									
Ca	–	ns	ns	0.94*	ns	ns	ns	0.59**	ns
Cl	–	–	0.64*	ns	ns	0.41*	ns	0.31**	ns
K	–	–	–	ns	ns	0.73*	ns	ns	ns
Mg	–	–	–	–	ns	ns	ns	0.60**	ns
Na	–	–	–	–	–	ns	ns	0.59**	ns
NH_4–N	–	–	–	–	–	–	0.31*	ns	ns
NO_3–N	–	–	–	–	–	–	–	ns	ns
SO_4–S	–	–	–	–	–	–	–	–	ns
pH	–	–	–	–	–	–	–	–	–
Cupressus									
Ca	–	-0.42*	0.81*	0.98**	ns	0.37*	0.96***	0.64*	ns
Cl	–	–	-0.53*	-0.39*	ns	ns	-0.49*	ns	ns
K	–	–	–	0.81*	ns	0.60**	0.85**	0.72*	ns
Mg	–	–	–	–	ns	0.36*	0.98**	0.62*	ns
Na	–	–	–	–	–	ns	ns	ns	ns
NH_4–N	–	–	–	–	–	–	0.42*	0.44*	ns
NO_3–N	–	–	–	–	–	–	–	0.63**	ns
SO_4–S	–	–	–	–	–	–	–	–	ns
pH	–	–	–	–	–	–	–	–	–

*** $P<0.001$; ** $P<0.01$; * $P<0.05$; ns-not significant.

Median mean Ca and NO_3–N concentrations under *Cupressus* were, respectively, 4 and 17 times higher than those below the natural forest, whereas under *Eucalyptus* the corresponding concentrations were 2 and 7 times lower than under *Cupressus*. Magnesium concentrations under *Eucalyptus* and the natural forest were 2.4 and 3.4 times less than those under *Cupressus* plantation. The concentration of Cl below *Eucalyptus* was about 2 times less than that under *Cupressus* and the natural forest. The higher Ca, Mg and NO_3–N concentrations in the soil solution under *Cupressus* relative to the other two forest types may indicate that these nutrients were in excess of tree and microbial requirements. In contrast, in the natural forest and *Eucalyptus* plantation, lower concentrations of the above nutrients could be associated with the species composition and its nutrients demand and uptake. Denitrification is not considered to be of importance in this well drained soil. Decreases in solute concentrations could also be due to dilution of nutrient concentrations in soil solution. However, the volume of water in the mineral soil as collected by the suction cups was highest under *Cupressus* at any depth in comparison to the other two forest types, except at the depth of 0.5 m where the water volume was highest under the natural forest (data not shown).

Much of the observed differences in median mean Ca, Mg and NO_3–N concentrations were attributed to the large differences in the subsoil (1 m depth) concentrations. The concentrations of Ca, Mg and NO_3–N at 1 m depth were 3, 4 and 81 times higher, respectively, under *Cupressus* than under *Eucalyptus*; the corresponding concentrations under the natural forest were 8, 7 and 23 times lower than under *Cupressus* (Table F4). The lower median Ca, Mg and NO_3–N concentrations under the natural forest and *Eucalyptus* plantation relative to *Cupressus* can be explained by the presence of diverse plant species with different rooting zones which serve as safety-net by intercepting/capturing nutrients at different soil depths within the profile (Schroth et al., 1999). In a ^{15}N tracer study made by Fischer (2004) in the same experimental plots large proportion of ^{15}N applied at the surface (0 m soil depth) under *Cupressus* was found in the deeper soil layer (0.3–0.6 m) confirming that heavy leaching had occurred in the *Cupressus* plantation.

Differential responses of soil solution nutrient concentrations to forest management and species have been reported (Callesen et al., 1999; De Schrijver et al., 2000; Lilienfein et al., 2000; 2001; Laclau et al., 2003). Nutrients in the soil solution where vegetation is poorly distributed or lacking may be potentially lost by leaching (Titus et al., 1997; Iseman et al., 1999). The later authors reported low nutrient leaching in a clear cut harvested plot colonized with an early successional species compared to the same clear cut plot where the density of an early successional species was low. Although the variations were not as pronounced as with that of *Cupressus*, the natural forest had about 2 times lower median mean Ca and NO_3–N concentrations as compared with those below *Eucalyptus*. The concentrations of K, Na, NH_4–N and SO_4–S were almost similar among the forest types. The median mean pH value under *Cupressus* was lowest compared to the natural forest and *Eucalyptus* (Table F4).

The temporal trends of the nutrient concentrations at any one depth were small within a given forest type (data not shown), but for some of the nutrients there seem to be a slight increase towards the end of the rainy season (Figs.F1–3). The concentrations of Ca, Mg and NO_3–N at any one time and depth usually were higher below the *Cupressus* plantation than below the other two forest types. Similar results have been observed by Lilienfein et al. (2001) in NO_3–N and total dissolved N concentrations under *Pinus* plantation relative to the native *Cerrado* growing on Oxisols in central Brazilian. No clear variations in Ca, Mg and NO_3–N concentrations were seen between *Eucalyptus* and the natural forest, particularly at the depth of 0.2 and 1 m (Figs.F1–3).

At the 0.5 m depth, however, concentrations appear to be higher under *Eucalyptus* plantation than under the natural forest.

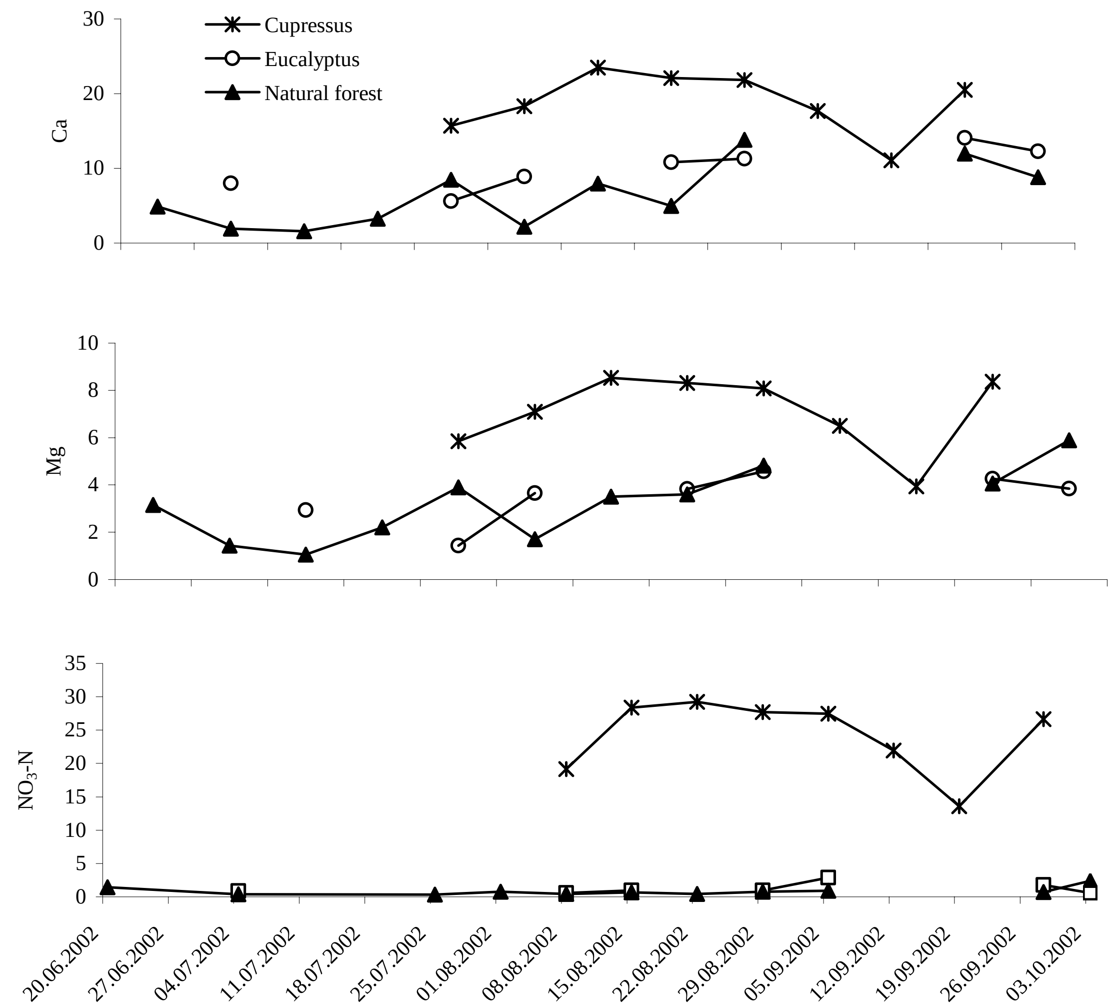


Fig. F1. Temporal trends in Ca, Mg and NO_3–N concentrations (mg l^{-1}) in the soil solution at 0.2 m soil depth under the three forest types.

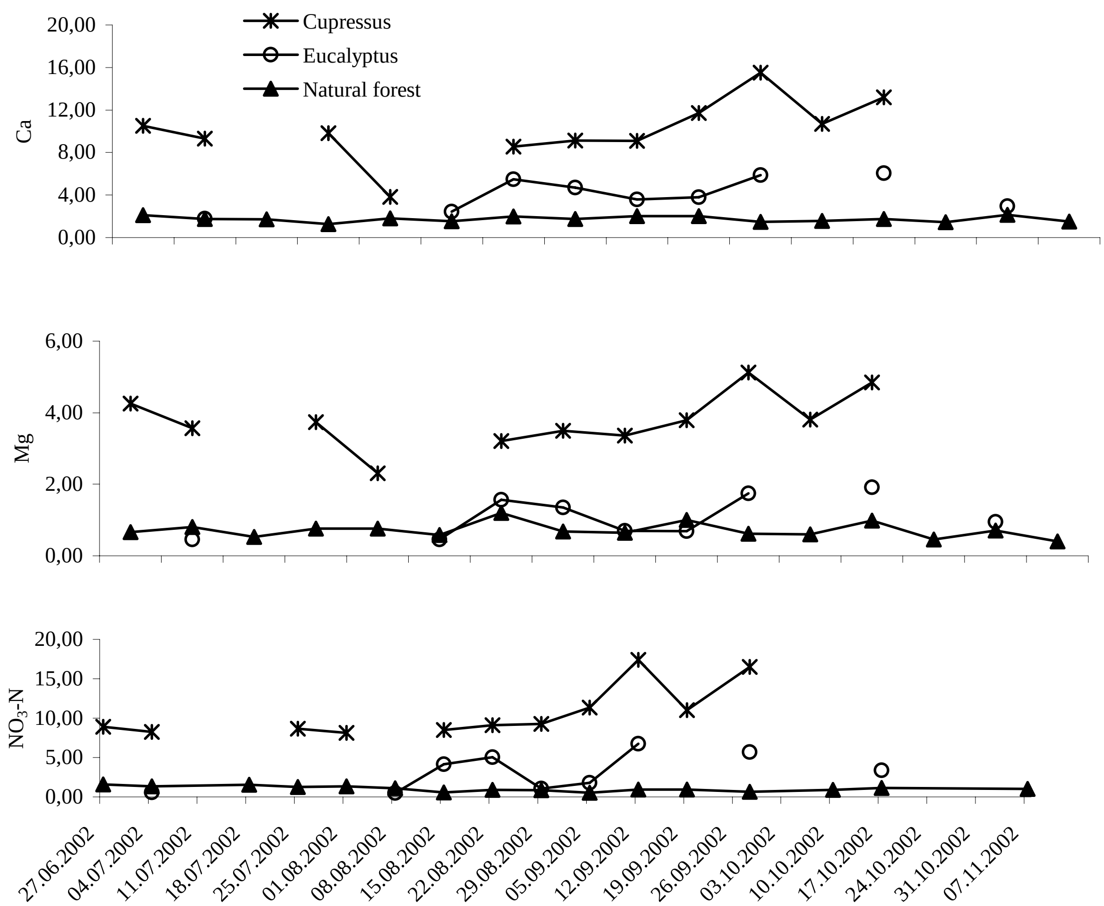


Fig. F2. Temporal trends in Ca, Mg and NO_3–N concentrations (mg l^{-1}) in the soil solution at 0.5 m soil depth under the three forest types.

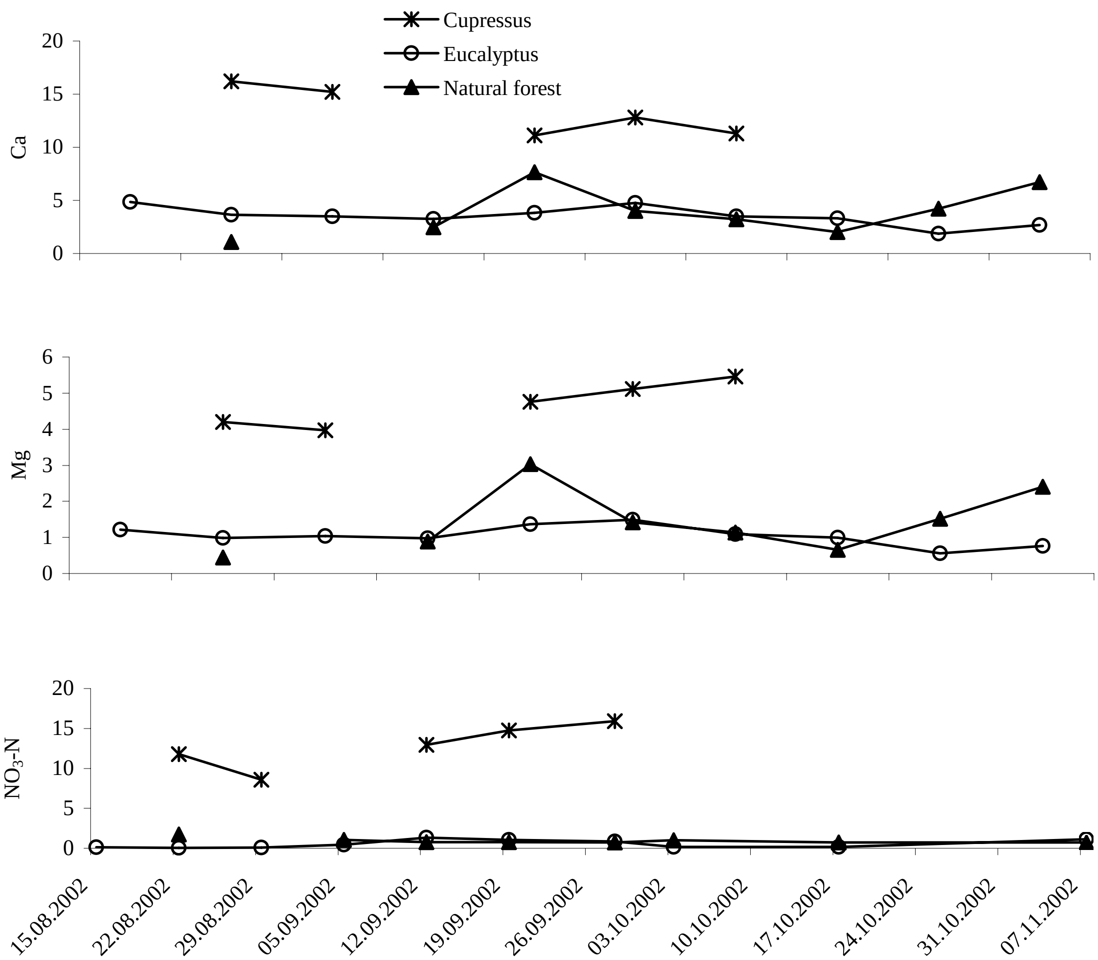


Fig. F3. Temporal trends in Ca, Mg and NO_3–N concentrations (mg l^{-1}) in the soil solution at 1 m soil depth under the three forest types.

5. Conclusion

The results showed that the nutrient concentrations in the mineral soil solution at Munesa responded differently to forest management and tree species. Ecosystem-specific patterns of vegetation composition and associated demand for nutrients appear to control nutrient dynamics. It was found that the soil solution under *Cupressus* plantation had higher Ca, Mg and NO_3–N concentrations than under *Eucalyptus* plantation and the natural forest at any one depth. The low rates of nutrient leaching in the natural forest and *Eucalyptus* plantation relative to *Cupressus*, probably resulted from high demand for nutrients due to the presence of diverse plant species with different rooting zones. The concentrations of Ca, Mg and NO_3–N under *Cupressus*, and Na and Cl under all forest types appeared to increase with depth increments. The concentrations of K, NH_4–N and SO_4–S under all forest types, and Ca, Mg and NO_3–N under the natural forest and *Eucalyptus* plantation decreased with depth increments. From the ecological point of view, the presence of cations and mineralised nitrogen in the soil solution of the subsoil is a sign of leaching, and may indicate that the site is not characterised by tight nutrient-cycling. The leaching of NO_3^- and exchange of H^+ produced by nitrification with basic cations, especially of Ca^{2+} and Mg^{2+} might be the cause for the leaching of these nutrients under *Cupressus*. The fact that NO_3–N and basic cations were leached from soils under *Cupressus* may be due to a high release of Ca, Mg and NO_3–N periodically that probably exceeded the retention capacity of the system. Loss of basic cations from the *Cupressus* ecosystem may, in the short-term, reduce site fertility and contribute to the onset of nutrient deficiencies. However, the positive impacts of annual cycling of nutrients through uptake by roots, fine root turnover, above-ground litter deposition and atmospheric inputs act to maintain fertility of the soils. Weathering, one of the most poorly quantified components of nutrient budgets, also acts to counteract loss of cations from the system.

Acknowledgments

I thank the German Research Foundation (DFG) for its support of the research through grant number Gu406–8.

References

Alban, D.H., 1982. Effects of nutrient accumulation by aspen, spruce and pine on soil properties. Soil Sci. Soc. Am. J. 46, 853–861.

Ashagrie, Y., Zech, W., Guggenberger, G., Teketay D., 2003. Changes in soil organic carbon, nitrogen and sulphur stocks due to the conversion of natural forest into tree plantations (*Pinus patula* and *Eucalyptus globulus*) in the highlands of Ethiopia. World Resour. Rev. 15, 462−482.

Brady , N.C., Weil, R.R., 1999. The nature and properties of soils, 12th Edition. Prentice Hall. Upper Sadalle River, NJ.

Callesen, I., Raulund-Rasmussen, K., Gundersen, P., Stryhn, H., 1999. Nitrate concentrations in soil solutions below Danish forests. For. Ecol. Manage. 114, 71−82.

De Schrijver, A., van Hoydonck, G., Nachtergale, L., De Keersmaeker, L., Mussche, S., Lust, N., 2000. Comparison of nitrate leaching under silver birch (*Betula pendula*) and Corsican pine (*Pinus nigra* ssp. Laricio) in Flanders (Belgium). Water, Air, Soil Pollut. 122, 77−91.

De Vries, W., Reinds, G.J., Vel, E., 2003. Intensive monitoring of forest ecosystems in Europe 2: Atmospheric deposition and its impacts on soil solution chemistry. For. Ecol. Manage. 174, 97−115.

EFAP. 1994 Ethiopian Forestry Action Plan. MoNRDEP, Addis Ababa, Ethiopia. Volume I.14p. and volume II. pp. 21− 34

FAO 1997 FAO/UNESCO Soil map of the world, revised legened with corrections and updates. World soil resources, Report No. 60, ISRIC, Wageningen, The Netherlands.

Fischer, H., 2004. Stickstoff in verschiedenen waldökosystemen des Munessa-Waldes (Süd Äthiopien): Stickstoffpools und ^{15}N-Tracerdynamik. Diplomarbeit. Bayreuth University. 100 pp.

Foster, N.W., 1985. Acid precipitation and soil solution chemistry within a maple–birch forest in Canada. For. Ecol. Manage. 12, 215–231.

Gosz, J.R., 1981. Nitrogen cycling in coniferous ecosystems. Ecol. Bull. (Stockholm) 33, 405–426.

Gundersen, P., Rasmussen, L., 1995. Nitrogen mobility in a nitrogen limited forest at Klosterhede, Denmark, examined by NH_4NO_3 addition. For. Ecol. Manage. 71, 75–88.

Gundersen, P., Emmet, B.A., Kjonaas, O.J., Koopmans, C.J., Tietema, A., 1998. Impact of nitrogen deposition on nitrogen cycling in forests: a synthesis of NITREX data. For. Ecol. Manage. 101, 37–55.

Hendershot, W. H., Lalande, H. and Duquels, M. 1993. Ion exchange and exchangeable cations. In Carter, M. R. (ed.) Soil sampling and methods of analysis. pp. 167–176. Can. Soc. Soil Sci.

Iseman, T.M., Zak, D.R., Holmes, W.E., Merrill, A.G., 1999. Revegetation and nitrate leaching from Lake States northern hardwood forests following harvest. Soil Sci. Soc. Am. J. 63, 1424–1429.

Laclau, J-P., Ranger, J., Nzila, J.D., Bouillet, J-P., Deleporte, P., 2003. Nutrient cycling in a clonal stand of *Eucalyptus* and an adjacent savanna ecosystem in Congo. 2. Chemical composition of soil solutions. For. Ecol. Manage. 180, 527–544.

Lemenih, M., Olsson, M., Karltun, E., 2004. Comparison of soil attributes under *Cupressus lusitanica* and *Eucalyptus saligna* established on abandoned farmlands with continuously cropped farmlands and natural forest in Ethiopia. For. Ecol. Manage. 195, 57–67.

Likens, G.E., Bormann, F.H., Pierce, R.S., Eaton, J.S., Johnson, N.M., 1977. Biogeochemistry of a forested ecosystem, Springer-Verlag, New York, 146 pp.

Lilienfein, J., Wilcke, W., Thomas, R., Vilela, L., Lima, S.D.C., Zech, W., 2001. Effects Pinus Caaribaea forests on the C, N, P and S status of Brazilian savanna Oxisols. For. Ecol. Manage. 147, 171–182.

Lilienfein, J., Wilcke, W., Ayarza, M.A., Vilela, L., Lima, S.D.C., Zech, W., 2000. Soil acidification in *Pinus caribaea* forests on Brazilian savanna Oxisols. For. Ecol. Manage. 128, 145–157.

Lugo, A.E., 1992. Comparison of tropical tree plantations with secondary forests of similar age. Ecol. Monogr. 62, 1–41.

Mahendrappa, M.K. 1990. Partitioning of rainwater and chemicals in to throughfall and stemflow in different forest stands. For. Ecol. Manage. 30, 65-72.

McDowell,W.H., Magill, A.H, Aitkenhead-Peterson, J.A., Aber, J.D., Merriam, J.L., Kaushal, S.S., 2004. Effects of chronic nitrogen amendment on dissolved organic matter and inorganic nitrogen in soil solution. For. Ecol. Manage. 196, 29–41.

Michelsen, A., Lisanework, N., Friis, I., 1993. Impacts of tree plantations in the Ethiopian highland on soil fertility, shoot and root growth, nutrient utilisation and mycorrhizal colonisation. For. Ecol. Manage. 61, 299–324.

Ranger, J., Marques, R., Jussy, J-H., 2001. Forest soil dynamics during stand development assessed by lysimeter and centrifuge solutions. For. Ecol. Manage. 144, 129–145.

Rasmussen, L., 1998. Effects of afforestation and deforestation on the deposition, cycling and leaching of elements. Agric. Ecosyst. Environ. 67, 153–159.

Ross, G. J. and Wand, C. 1993. Extractable Al, Fe, Mn and Si. In Carter, M. R. (ed.) Soil sampling and methods of analysis. pp. 239–246. Can. Soc. Soil Sci.

Schroth, G., da Silva, L.F., Seixas, R., Teixeira, W.G., Macedo, J.L.V., Zech, W., 1999. Subsoil accumulation of mineral nitrogen under polyculture and monoculture plantations, fallow and primary forest in a ferralitic Amazonian upland soil. Agric. Ecosyst. Environ. 75, 109–120.

Schrumpf, M., 2004. Biogeochemical investigations in old growth and disturbed forest sites at mount Kilimanjaro. Dissertation, PhD., Bayreuth University.

Titus, B.D., Roberts, B.A., Derring, K.W., 1997. Soil solution concentrations on three white birch sites in central Newfoundland following different harvesting intensities. Biomass and Bioenergy 13, 313–330.

Tokuchi, N., Takeda, H., Iwatsubo, G., 1993. Vertical changes in soil solution chemistry in soil profiles under coniferous forest. Geoderma 59, 57–73.

8. DECLARATION/ERKLÄRUNG

Hiermit erkläre ich, dass ich die Arbeit selbständig verfasst und keine anderen als die vom mir angegebenen Quellen and Hilfsmittel benutzt habe.

Ferner erkläre ich, dass ich anderweitig mit oder ohne Erfolg nicht versucht habe, diese Dissertation einzureichen. Ich habe keine gleichartige Doktorprüfung an einer anderen Hochschule endgültig nicht bestanden.

Bayreuth, November 2004