# Soil organic carbon dynamics in pastures established after deforestation in the humid tropics of Costa Rica

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# Soil organic carbon dynamics in pastures established after deforestation in the humid tropics of Costa Rica

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# Stellingen

1. Accurate prediction of soil-atmosphere trace gas fluxes requires knowledge not only of actual land use, but also of land use history.

   M. Keller, E. Veldkamp, A.M. Weitz and W.A. Reiners. Effect of Pasture age on soil trace gas emissions from a deforested area of Costa Rica. Nature, *in press*.

2. De afbraaksnelheid van organische stof in de bodem neemt sterk af met de diepte.

   dit proefschrift.

3. Uitbreiding van bodem organische stof modellen met isotoop fractionering, vergroot het inzicht in de optredende processen en zal de calibratie van deze modellen verbeteren.

   dit proefschrift.

4. Verwaarlozing van compactie van bodems als gevolg van ontbossing, kan leiden tot een grote onderschatting van de hoeveelheid afgebroken bodem organische stof.

   dit proefschrift.

5. De netto $CO_2$ emissie uit de bodem, die optreedt als gevolg van het kappen en vervangen van het tropisch regenwoud door laag produktieve graslanden kan met 60 % worden verminderd indien verbeterde grassoorten worden geïntroduceerd.

   dit proefschrift.

6. $CO_2$ emissie is een slechte indicator voor netto koolstof verlies van een bodem.

   dit proefschrift.

7. In de Atlantische Zone van Costa Rica is de kans dat een stuk regenwoud wordt gekapt, in sterke mate afhankelijk van de bereikbaarheid en de bodemkwaliteit van het gebied.

dit proefschrift.

8. Een verantwoorde keuze van lokaties voor bodemkundig onderzoek kan alleen worden gemaakt met voldoende inzicht in landschapsvormende processen.

9. De huidige trend naar meer integratie van verschillende disciplines in onderzoek en onderwijs dreigt te ontaarden in oppervlakkig generalisme.

10. Multidisciplinair onderzoek mag geen doel op zich zijn; het kan een noodzakelijk middel zijn om tot de oplossing van een probleem te komen.

11. Een goede samenwerking in wetenschappelijk onderzoek begint bij een goede verstandhouding op het persoonlijke vlak.

12. De bezorgdheid van wetenschappers voor het milieu, zoals blijkt uit menig onderzoeksvoorstel, staat vaak in schril contrast met het gedrag dat dezelfde wetenschappers vertonen.

13. Het onbegrip waarmee hoog opgeleide vrouwelijke partners te maken krijgen als ze een passende baan zoeken in een ontwikkelingsland maakt op een pijnlijke manier duidelijk dat emancipatiebeleid nog niet veel meer is dan mooie woorden.

14. Afschaffing van buitensporige financiële vergoedingen die onderzoekers en ontwikkelingswerkers van bepaalde organisaties voor verblijf in de tropen ontvangen, zal een positieve impuls geven aan de kwaliteit van het geleverde werk.

Stellingen behorend bij het proefschrift 'Soil organic carbon dynamics in pastures established after deforestation in the humid tropics of Costa Rica'. Ed Veldkamp, 22 September 1993.

# Abstract

Currently, rates of deforestation in the tropics are probably higher than ever before in the past. As a consequence, changes in the earth's physical and chemical environments are proceeding at unprecedented rates. Increasing atmospheric concentrations of $CO_2$, $N_2O$ and other trace gases, caused by enhanced emissions from soils after forest clearing, show that deforestation in tropical areas is of global importance. Recent estimates suggest a net release of carbon from the world's tropics, due to deforestation, of between 0.42 and 1.60 Pg C $yr^{-1}$ (1 Pg = $10^{15}$ g) of which 0.1 to 0.3 Pg C $yr^{-1}$ are attributed to decreases in soil organic matter content. This carbon release from tropical areas is second only to the global release from the burning of fossil fuels (which is about 5.3 Pg C $yr^{-1}$).

The main objective of this thesis was to quantify the changes in soil organic carbon storage and the resulting release of $CO_2$ after the conversion of tropical rain forest to pasture on two contrasting soil types in the humid tropics of Costa Rica. To study changes in soil organic carbon storage, sites of an Andisol and an Inceptisol, cleared at different times in the past (deforestation sequences) were compared. A deforestation map, based on aerial photographs from the period 1952 - 1984, was made for a part of the Atlantic Zone of Costa Rica, providing a well documented history of forest clearing. Using GIS techniques, this deforestation map was combined with an available soil map to select the study sites. Analysis of deforestation patterns on the map demonstrated a close relation of deforestation rate with accessibility and soil quality.

Soil organic matter levels are the result of complex production and decomposition processes. The input of carbon from grass plant roots into the soil was quantified, using pulse labelling with $^{14}C$. The pulse labelling experiment revealed that root dry matter production of an improved pasture like *Brachiaria* (12 Mg $ha^{-1}$ $yr^{-1}$) was about twice the root production of a low-productive species like *Axonopus* (6 Mg $ha^{-1}$ $yr^{-1}$). Root biomass of *Brachiaria* was about three times the root biomass of *Axonopus* due to higher residence time of carbon in the root biomass of *Brachiaria* as compared to *Axonopus*. Root exudates of grass plants were found to have a minor direct contribution to the longer term carbon dynamics, either because exudation rate was small or because decomposition was fast and complete.

Decomposition of soil organic matter was measured using the $\delta^{13}C$ method,

which uses differences in natural $^{13}C$ isotope levels in vegetation (C3 and C4 vegetation) and soil organic matter to calculate changes in soil organic carbon. The method is applicable in soil organic matter studies where a change from C3 to C4 vegetation has occurred (or vice versa). It was demonstrated that for a correct application of the method, detailed information of changes in bulk densities accompanying changes in land use was vital. An uncertainty analysis of the $\delta^{13}C$ method demonstrated that the output of the $\delta^{13}C$ method in soil organic matter studies was highly variable due to variations in the input data. Spatial variability was the main source of the uncertainty in input data. However, variations due to sampling error and short scale variability were considerable and should not be ignored.

Information on carbon input and decomposition was integrated, using a simple structured soil organic carbon (SOC) model which included carbon isotope fractionation during decomposition and depth dependent decomposition and humification rates. With this model, the observed changes in soil organic carbon and corresponding $\delta^{13}C$ levels during the conversion from a humid tropical forest to a cattle pasture were simulated successfully for the two soil types. With the calibrated model the cumulative net $CO_2$ release was calculated. The cumulative net release of $CO_2$ for pastures with low productive grass species (*Axonopus compressus*), varied from 31.5 (Humitropept) to 60.5 Mg C ha$^{-1}$ (Hapludand) in the first 20 years after forest clearing. These cumulative emissions could be reduced to 12.0 and 24.7 Mg C ha$^{-1}$ respectively, if higher productive grass species (e.g. *Brachiaria dictyoneura*) would be introduced into the area.

Decomposition rates were strongly influenced by depth. Inclusion of deeper layers in soil organic carbon simulation studies and considering carbon isotopes will probably improve the performance of SOC models in long-term studies.

*Aan mijn ouders*

# Voorwoord

Hoewel slechts mijn naam op de voorkant staat vermeld, zou dit proefschrift zonder de bijdragen van verschillende mensen er niet zijn geweest. Ik wil dan ook graag van de gelegenheid gebruik maken om deze mensen te bedanken.

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I encountered problems with the pasture experiments. During my stay in Costa Rica I met Michael Keller, who proved to be a very enthusiastic and stimulating colleague. Our similar interests evolved in a very good cooperation which brought new insights in the dynamic character of nitrous oxide emissions after deforestation. I look forward to our future cooperation!

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Ed Veldkamp
Wageningen, juni 1993

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# CONTENTS

# *Chapter 1*

# General Introduction

# General Introduction

There is a growing concern about the increasing atmospheric concentrations of carbon dioxide, nitrous oxide and other gases, which provide an unequivocal signal of the global impact of human activities. Researchers have predicted that increasing concentrations of these gases in the atmosphere will result in significant changes in climate, because $CO_2$ and other trace gases, produced by industrial and agricultural processes absorb thermal radiation emitted by the earth's surface. Carbon dioxide is the most abundant 'greenhouse' gas and contributes about 50 % to greenhouse warming, due to its high annual emission and long atmospheric residence time.

## Tropical deforestation and the global carbon cycle.

Currently, rates of deforestation in the tropics are probably higher than ever before in the past. As a consequence, changes in the earth's physical and chemical environments are proceeding at unprecedented rates. Forest clearing not only has an important influence on the local physical environment (e.g. soil degradation, nutrient loss) but increasing atmospheric concentrations of $CO_2$, $N_2O$ and other trace gases, caused by enhanced emissions from soils after forest clearing, show that deforestation in tropical areas is of global importance. Recent estimates suggest a net release of carbon from the world's tropics, due to deforestation, of between 0.42 and 1.60 Pg C $yr^{-1}$ of which 0.1 to 0.3 Pg C $yr^{-1}$ are attributed to decreases in soil organic matter content (Detwiler and Hall, 1988). This carbon release from tropical areas is second only to the global release from the burning of fossil fuels (which is about 5.3 Pg C $yr^{-1}$).

In Latin America, the total net release of carbon between 1850 and 1985 due to changes in land use was estimated to 30 Pg C (Houghton *et al.*, 1991). The main source of this net $CO_2$ emission was the increased area of pastures. In 1985, Latin America had the largest net flux of biotic carbon (0.67 Pg C $yr^{-1}$), at least one-half of the total for the world's tropics. The largest contribution to this flux resulted from increases in the area of degraded lands, which mainly consist of degraded pastures (Houghton et al., 1991). Although loss of soil organic carbon after deforestation is less in pastures than in croplands (Detwiler, 1986), pastures play a key role in the release of biotic carbon to the atmosphere, due to the large area they cover.

## Changes in carbon stocks of pastures as a result of deforestation

Conversion from tropical forest to cropland or cattle pasture is accompanied by changes in the amount of carbon held in vegetation and soil. When forests are cleared, much of the above-ground biomass is burned. Some of the wood is harvested for products that oxidize more slowly. The rest of the above-ground and below-ground material decomposes exponentially as does the soil organic matter.

The loss of carbon from pasture soils is generally less than the loss from cropland soils, mainly because of the differences in cultivation practices. Detwiler (1986) estimated that the cultivation of tropical soils, during 5 yr after clearing, reduces their carbon content by 40%, the use of these soils for pasture reduces it by about 20%. The findings of changes in carbon stocks are extremely variable, however. In most studies a loss of carbon is found that may be as high as 40 % of the C originally present in the forest soil (Houghton et al., 1991). On the other hand there are studies in which no change in carbon content is found (Buschbacher, 1984) and some studies even report higher carbon content in the pastures than in the adjacent forest (Lugo et al., 1986).

## Objectives

As was outlined in the previous paragraph, a lot of uncertainty exists as to the effect of cattle pasture establishment after forest clearing on the soil organic carbon budget. Even less information is available if pastures are established on soils derived from volcanic material (Andisols), which are famous for their enormous soil organic carbon storage capacity.

The main objective of this thesis was to quantify the changes in soil organic carbon storage and the resulting release of $CO_2$ after the conversion of tropical rain forest to cattle pasture on two contrasting soil types in the humid tropics of Costa Rica.

## Study area

The area where the different studies, summarized in this thesis, were executed was the Atlantic Zone of Costa Rica. The climate in the Atlantic Zone is humid tropical. The mean annual temperature is 26 °C with little daily and annual variations. The mean yearly precipitation varies between 3,000 and 6,000 mm yr$^{-1}$. February to April represent a relatively dry period with mean monthly precipitation values between 100 and 300 mm. The soils in the area belong predominantly to the Andisols and Inceptisols, mainly formed in fluvio-laharic deposits of andesitic composition. The studies in this thesis were executed principally on two soil types: an Eutric Hapludand (local name: Suelo Los Diamantes) and an Andic Humitropept (local name: Suelo Neguev). The Eutric Hapludand is typical of the relatively young, fertile, non acid soils, common in the Atlantic Zone of Costa Rica, while the Andic Humitropept represents the smaller group of relatively old, nutrient-poor and acid soils.

In the Atlantic Zone of Costa Rica, large areas of tropical lowland forest have been cleared in the last 40 years. A considerable part of this area has been replaced by planted, low-productive cattle pastures. In 1986, 46.4 % of the Atlantic Zone was covered by forest. Pastures and 'wooded area' (which also corresponds to grassland) together occupied more than 50 % of the deforested area (Huising, 1993).

## Research strategy

When the research started in 1989, considerable information was available on the carbon cycle in undisturbed forests (Raich, 1980; 1983) in the Atlantic Zone of Costa Rica. In contrast, almost no information existed on the carbon cycle in cattle pastures. Therefore the studies in this thesis emphasize the production and decomposition of soil organic carbon in pastures.

As the available four years were a limited period, it was impossible to follow changes in soil organic carbon of a cleared site in time. In contrast, sites which were cleared at different intervals (deforestation sequences), were compared. The first step in the site selection was to obtain information about the deforestation history of the study area. For a part of the Atlantic Zone of Costa Rica a deforestation map was made, based on aerial photographs from the period 1952 to 1984. This deforestation map was combined with an available soil map to select the study sites.

Soil organic matter levels are the result of dynamic production and decomposition processes. The input of carbon from plant roots into the soil organic matter was quantified, using $^{14}$C. The decomposition of soil organic matter was quantified using the change in natural $^{13}$C from forest to pasture. Finally, the results of these production and decomposition studies were integrated in a dynamic simulation model which was used to calculate the resulting net $CO_2$ fluxes from soil to atmosphere.

## Outline of this thesis

In this thesis, articles are compiled that have been or will be published independently in international journals. As a consequences duplications sometimes occur.

In chapter 2, a deforestation map of a part of the Atlantic Zone of Costa Rica was analyzed to deduce the effects of soil quality and accessibility on deforestation patterns.

In chapter 3, pulse labelling with $^{14}CO_2$ was applied to estimate the input of organic C to the soil in low-productive pasture types (*Axonopus compressus*) and to test the hypothesis that replacement of these grass species by improved pastures (*Brachiaria dictyoneura*) would increase input of organic carbon into the soil.

In chapter 4, differences in $\delta^{13}C$ between soil organic carbon derived from forest and pasture were used to quantify the changes in soil organic carbon after deforestation under low productive pastures.

In chapter 5, the $\delta^{13}C$ method which was used in chapter 4 was evaluated with help of an uncertainty analysis. The influence of spatial variability on sample number requirement was calculated.

In chapter 6, the information on production and decomposition of soil organic carbon, which was gathered in the preceding chapters, was integrated in a dynamic simulation model. The field observations of soil organic carbon levels were simulated and the resulting net $CO_2$ fluxes were calculated.

In chapter 7, the conclusions from the previous chapters were summarized and the implications for future research were discussed.

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## *Chapter 2*

# Deforestation trends in the Atlantic Zone of Costa Rica: a case study

*Land degradation and rehabilitation 3, 71-84 (1992)*

E. Veldkamp, A.M. Weitz, I.G. Staritsky and E.J. Huising

# Deforestation trends in the Atlantic Zone of Costa Rica: a case study.

## Abstract

In this study we have tried to link a quantitative inventory of deforestation to possible factors driving forest clearing, like accessibility and soil quality. The Study Area covered 395 km$^2$ in the Atlantic Zone of Costa Rica. Basic information came from aerial photographs taken in 1952, 1960, 1971 to 73, 1981 and 1984. Photo interpretations were digitized and processed with the help of a monoplotting programme. Three phases in deforestation were distinguished:
1. The period before 1960, characterized by a gradual increase in the rate of deforestation, with preferential clearing along rivers and on fertile soils.
2. The period 1960-1972. This period shows an increased clearing rate caused by a new centre of colonization, Cariari. Furthermore, banana plantations entered the area and main roads were constructed. Deforestation took place preferentially along roads and on fertile soils.
3. In the period 1972-1984 the remaining forest 'islands' were cleared at an accelerated rate. This was because only small areas of forest were left by this period. Also between 1972 and 1984 the fertile soils were cleared most rapidly.

**Key words**: analysis of deforestation patterns, aerial photograph interpretation, Costa Rica, deforestation trends, accessibility and deforestation, digital processing of aerial photographs.

## Introduction

Deforestation in tropical countries is a complex problem. Much research has been done on the forces driving deforestation, with respect to demographic, social, economic and biological factors (e.g. Sanchez, 1990), while other studies deal with the changes in forest cover and deforestation rates in tropical countries (e.g. Green, 1982; Keogh, 1984; Tucker, *et al.*, 1984). Such inventory studies generally do not attempt to demonstrate relationships between deforestation patterns and the physical environment or infrastructure, because basic information such as topographical maps and soil maps is usually not available or is of poor quality. An exception is the study by Sader & Joyce (1988) who found a relation between total forest cleared and slope gradient in the Central Valley of Costa Rica. Furthermore, development of roads providing access to the forest was an important factor in Costa Rican deforestation during the period 1940-1983. Our study links a quantitative inventory of deforestation to possible driving factors, like accessibility and soil quality. We tested the hypotheses that deforestation preferentially takes place:

- along major rivers;
- along roads;
- on fertile soils.

We selected a study area, characterized by rapid deforestation during the last 25 years. Aerial photographs were used for a detailed reconstruction of deforestation patterns and trends at different points in time.

### *Study Area*

The study area is situated in the Atlantic Zone of Costa Rica, about 60 km northeast of San José and partially covers the districts of Roxana, Rita and Cariari. The area forms a quadrangle of 525 km$^2$ and extends from about 10°15' N to 10°30' N and 83°35' W to 83°50' W (Figure 1). As the coverage of the photographs is not complete (in particular a part of the 1971 to 73 record is missing) an area of 395 km$^2$ was used in the study. The climate is humid tropical; the average annual temperature is about 26 °C and mean annual rainfall is between 3,000 and 6,000 mm.

The study area is situated at the foot of the Turrialba Volcano. Altitudes range from 10 m in the north to about 180 m above sea-level in the south. The area is covered with laharic, fluviolaharic and fluvial deposits, mostly of andesitic

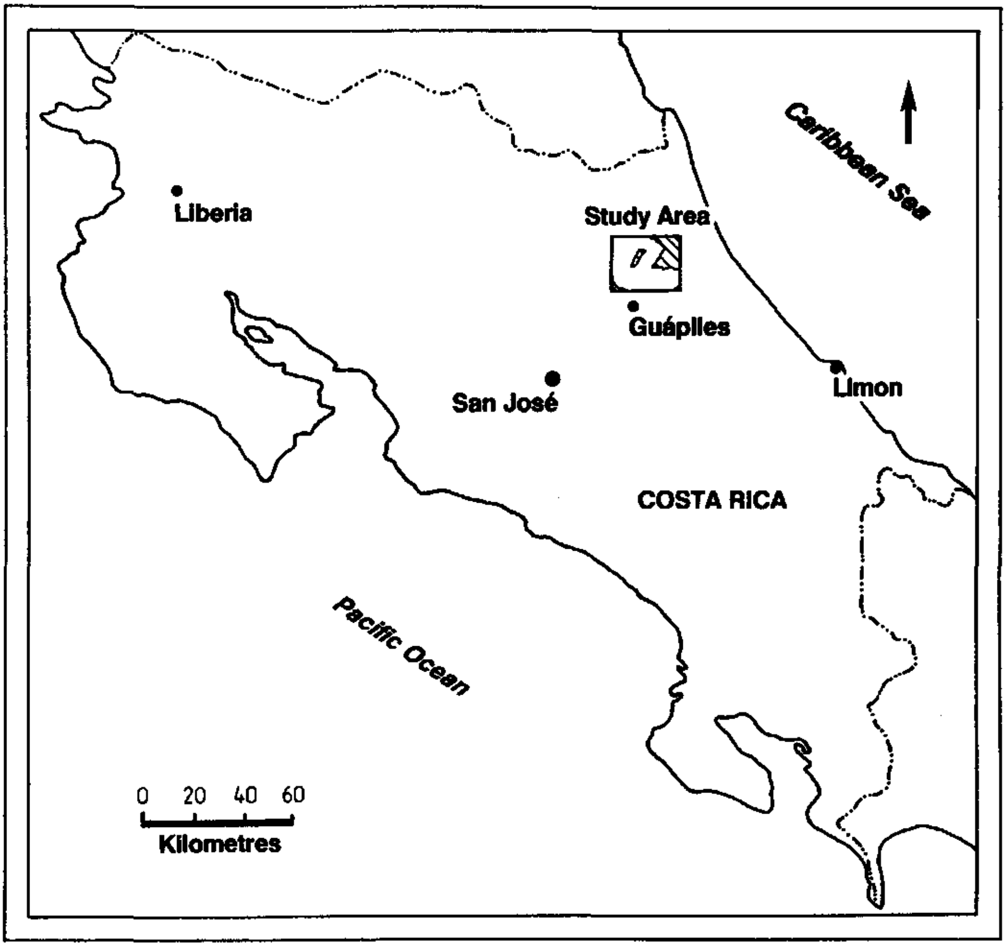

Figure 1. Location of Study Area

composition. The majority of the soils in the study area are fertile and classified as Andisols (Udivitrands & Hapludands) or Inceptisols (Eutropepts & Humitropepts) (USDA, 1975; ICOMAND, 1988).

The natural vegetation consists of tropical lowland rainforest, typical species in the forest are *Carapa slateri* and *Penthaclethra macroloba* (Nuhn & Perez, 1967). Tosi (1969) distinguished the Life Zones Premontane, Moist Forest and Tropical Moist Forest in the area. The two main types of land use after deforestation are low-input pastures and high-input banana plantations. Nowadays the study area is one of the main banana producing areas of Costa Rica.

## Materials

### *Aerial photographs and topographic maps*

The primary sources of information for the study were aerial photographs taken in 1952 (1:40,000), 1960 (1:60,000), 1971 (1:20,000), 1973 (1:25,000), 1981 (1:35,000) and 1984 (1:80,000). The photograph numbers are listed in Table 1. The 1984 photographs are colour infrared (false colour), the others are panchromatic black-and-white photographs. Because the 1973 photographs only cover the southern half of the study area and those of 1971 the north-western quarter, both sets were used together. For calculation purposes the year 1972 represents the 1971/73 coverage. Other primary sources of information used in the study are topographical base maps number 3446-I, 3446-IV, 3447-II and 3447-III (scale 1:50,000).

**Table 1.** Aerial photograph series used in the study.

| Date | Scale | Line | Number |
|---|---|---|---|
| 24/04/1952 | 1:40,000 | L17 | 658-664 |
| | | L18 | 639-645 |
| | | L19 | 620-624 |
| 11/03/1960 | 1:60,000 | L70 | 3022-3026 |
| | | L71 | 3050-3054 |
| 20/10/1971 | 1:20,000 | L5 | 855-859 |
| | | L6 | 874-881 |
| | | L7 | 914-916 |
| 23/08/1973 | 1:25,000 | L250 | 122-131 |
| | | L254 | 150-161 |
| | | L258 | 188-198 |
| 17/03/1981 | 1:35,000 | L254 | 24893-24901 |
| | | L260.6 | 24904-24910 |
| | | L266.5 | 24930-24936 |
| 20/04/1984 | 1:80,000 | L17E | 85-89 |
| | | L18O | 32-35 |

### *Soil quality, river, and infrastructure map*

The soil map of the North Atlantic Zone of Costa Rica (scale 1:100,000), used in this study was prepared by Wielemaker & Oosterom (1990) and is available as soil information system. We used a soil quality map based on the suitability classification of the information system. For our study we simplified the classification of Wielemaker & Oosterom (1990) for practical reasons. The original five suitability classes were converted into three suitability classes. Both classifications are based on soil fertility. The three suitability classes are:

1. Fertile soils, areas with little restrictions for agricultural use.
2. Moderately fertile soils, areas with some to severe restrictions for agricultural use.
3. Unfertile soils, areas which should be given protection.

The resulting soil quality map (Figure 2) shows the area of each soil class per polygon (as percentages). The two main rivers in the area are the Rio Tortuguero and the Rio Jimenez. The digitized rivers were also extracted from the information system build by Wielemaker & Oosterom (1990). The infrastructure map was generated in the same way. Both river and infrastructure maps were based on the 1984 aerial photographs.

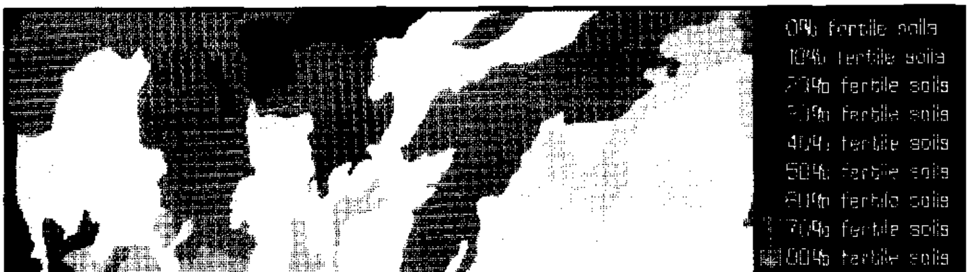

Figure 2. Example of soil quality map. Scale approx. 1:150,000

## Methods

### *Aerial photograph interpretation*

Because the edges of aerial photographs, exhibit focal distortion, only the central part could be used for interpretation. The boundaries of the interpretation areas of the individual photographs were linked to those of the adjacent aerial photographs. In this way a mosaic of interpretation areas was obtained, covering the whole study area. During the interpretation, three types of land use were distinguished:

- Apparently undisturbed or only slightly disturbed forest. The maximum detectable human influence being the felling of some trees, but the land had remained essentially forest.
- Banana plantation.
- Rest group. This includes any other land use or land features (e.g. urban area, grasslands, etc.).

Photo interpretation was done using a Topcon mirror stereoscope, on transparent foil with a 0.5 mm pencil.

### *Digital processing of the aerial photograph interpretation*

Information of the aerial photographs was transformed into a map by digital monoplotting (Makarovic, 1973; Radwan & Makarovic, 1980). We used the monoplotting system as described by Molenaar & Stuiver (1987). In monoplotting, interpretation results are digitized as photograph coordinates and numerically transformed into terrain coordinates. From the database built, information can be extracted to make maps, map overlays or photo image-overlays.

Photograph processing by digital monoplotting is explained in Figure 3. To perform a transformation by means of numerical substitution a digitized photograph interpretation, a digital height model (DHM) and transformation parameters are needed. The transformation parameters were calculated using the MONOPLOT program (Molenaar & Stuiver, 1987). Ground control points were used for which both photo and terrain coordinates were known. The terrain coordinates for the ground control points were obtained from the topographical map. Contour lines from topographical maps were digitized, resulting in a DHM from terrain coordinates. With the obtained transformation parameters the DHM

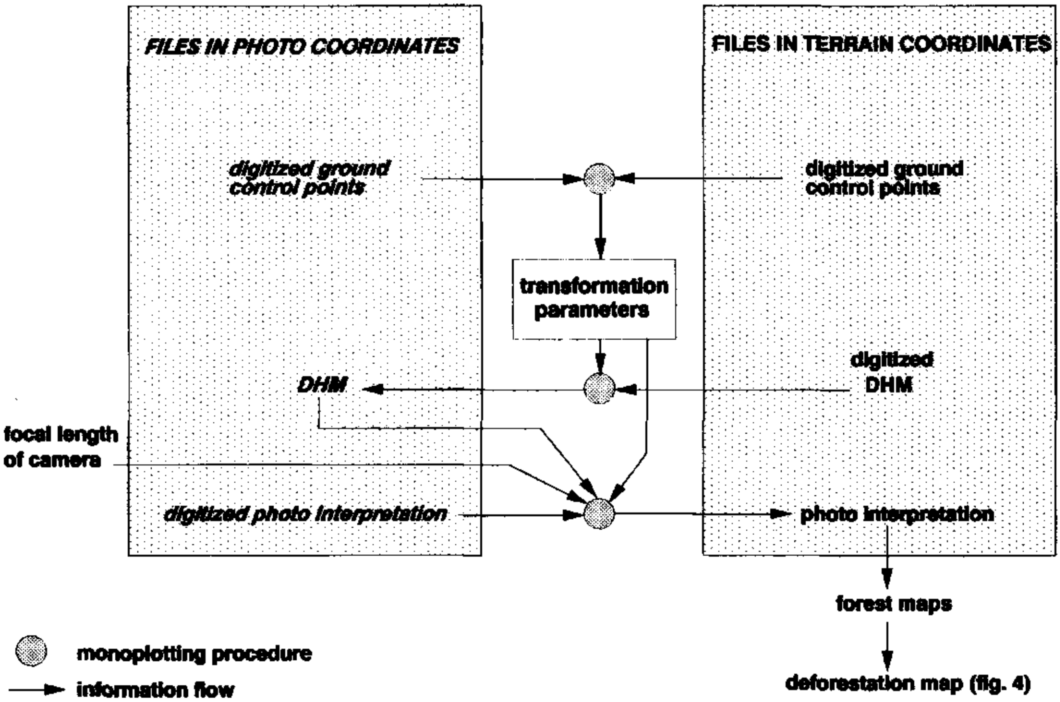

Figure 3. Flow chart of aerial photograph interpretation processing.

was transformed from terrain to photo coordinates. The photo interpretations including the borders of the interpretation areas were digitized on a DIGIPAD using the digitizing module of ILWIS (ITC, 1991). This resulted in a file with interpretation lines as photo coordinates. Transformation parameters, the DHM and information on the principal distance (focal length) of the camera, were used to transform the photo interpretations in photo coordinates into terrain coordinates. After transformation all single photo interpretation files in terrain coordinates were joined. Interpretation lines of adjacent aerial photographs were linked together and intermediate interpretation area boundaries were deleted, resulting in five maps with forest areas for different years. The forest maps of the five years, thus obtained, were copied together resulting in a deforestation map of the study area (Figure 4).

## *Calculations and map manipulations*

The deforestation rates were expressed on two relative scales. The deforestation rate in relation to the forest area in 1910 was calculated by the equation 1.

Equation 2 is based on that of Sader & Joyce (1988).

$$\textit{Deforestation Rate}\ (y^{-1}) = \frac{\left(\frac{F_1 - F_2}{F_0}\right)}{N} \times 100 \quad (\textit{eq.}\ 1)$$

$$\textit{Deforestation Rate}\ (y^{-1}) = \frac{\left(\frac{F_1 - F_2}{F_1}\right)}{N} \times 100 \quad (\textit{eq.}2)$$

where:
$F_0$ = forest area at beginning of deforestation (=1910)
$F_1$ = forest area at beginning of reference period
$F_2$ = forest area at end of reference period
N = number of years in reference period

With equation 1, it is possible to relate the clearing rate of each reference period to the original forest area, whereas equation 2 gives insight in the deforestation rates in relation to the remaining forest area.

To deduce the influence of the two main rivers on the deforestation history, the deforestation rate in the area close to the rivers (<500 m from the river) was compared to that in the area far from the rivers (>500 m). We used a distance of 500 m to get a <500 m area covering about 25% of the study area. To relate the effect of roads to the deforestation history, the same procedure was followed for an area within 250 m distance. An area of 250 m was considered instead of a 500 m area, because the road network is relatively dense.

For the three soil qualities, the deforestation rates were also calculated and compared.

Map manipulation was necessary to combine information on soil type, infrastructure and deforestation. By means of the ERDAS system (ERDAS, 1990), the area within 500 m of the two main rivers was delineated and converted into grids (50 * 50 m). The same was done for the main roads with a 250 m area along the roads. The soil quality map was also converted to grid format.

Overlaying of the different maps and area calculation was done in the ERDAS system (ERDAS, 1990).

Figure 4. Deforestation map 1952-1984. Scale approx. 1:150,000

## Results and Discussion

We present first an overview of the general deforestation trends, subdivided per period. Next the relationship between clearing rates, accessibility, soil quality and the introduction of banana plantations is discussed. No attempt is made to describe the socio-economic and political factors which influence deforestation, although some factors are mentioned to support the interpretation of the clearing rates and patterns. Deforestation in the study area must have started around 1910. We base this assumption on the construction of the railway from Guácimo to Carillo at the end of the nineteenth century, running about 8 km south of the study area. As can be seen in Figure 5 and Table 2, since 1910 there has been a lot of change. The area under forest has gradually decreased until in 1984 only 2,239 ha (5.7%) of the original forest was left. Between 1910 and 1984, three deforestation periods can be distinguished, each characterized by a specific

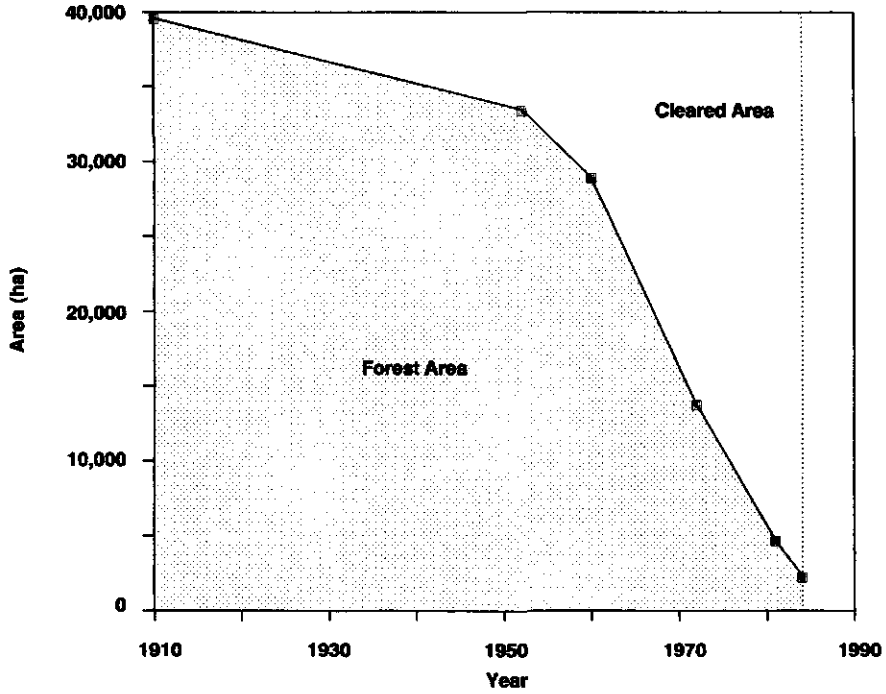

Figure 5. Change in forest area 1910-1984.

clearing pattern.

## *The period before 1960*

Until 1960 a clearly visible deforestation frontier existed, together with small cleared spots in the forest beyond. The colonization of the area developed from Guápiles northward. Deforestation took place on a relatively small scale, clearing patterns were more-or-less irregular, influenced by local natural features like rivers. The deforestation rate gradually increased between 1952 and 1960 (Figure 6). On the aerial photographs the main land use identified after deforestation was grassland.

## *The period 1960-1972*

The picture completely changed in the period 1960-1972. In this period the colonies Cariari and Astúa Pirie were founded by the ITCO (Instituto de Tierras y Colonización), a government land reform organization. Cariari, became the centre of colonization (Brooijmans and van Sluys, 1990). In the same period

**Table 2.** Temporal changes in forest area and deforestation rate in the study area.

| Year | Forest area | | Deforestation rate |
|---|---|---|---|
| | (ha) | (%) | (ha/yr) |
| 1910 | 39537 | 100.0 | 0 |
| 1952 | 33373 | 84.4 | 147 |
| 1960 | 28881 | 73.1 | 561 |
| 1972 | 13731 | 34.7 | 1263 |
| 1981 | 4639 | 11.7 | 1010 |
| 1984 | 2239 | 5.7 | 800 |

banana plantations were introduced into the study area and attracted many people. Both the founding of Cariari, and the introduction of banana plantations seem to be responsible for the large increase in deforestation rate (Figure 6) and the complete shift in deforestation pattern after 1960. In 1972 no clear deforestation frontier was visible, but areas in 'the middle of the forest' were cleared. This was mainly done by large banana plantations. In the period 1960-1972 an area of 15,150 ha was cleared. Of this cleared area, in 1972 5,886 ha (38.9%) was under banana plantations.

### *The period 1972-1984*

Due to the relatively small forest area left, the deforestation rate as calculated in equation 1 decreased after 1972, while the deforestation rate calculated in equation 2 continued to increase (Figure 6). The period is characterized by a decreasing size of the forest "islands". The bulk of deforestation took place in the area north of the study area (Wielemaker, 1990).

### *Soil quality*

In the study area 80.6 per cent of the soils was classified as 'fertile', 12.4% as moderately fertile and 7.0% as unfertile. To deduce the influence of the soil quality on the deforestation rate, the deforestation per soil class was analyzed (Table 3 and Figure 7). In Figure 7 the cumulative deforestation (in per cent) is plotted against the time. The slope of the line is a measure of the deforestation rate. Until 1972 the fertile soil line is steeper than the slope of the other two lines, indicating a preference to clear fertile soils. Clearing rates on fertile soils decrease after 1972, while the deforestation rates of moderately fertile and even more on unfertile soils continued to increase until 1984. The 'back-log' of forest clearing on the less fertile soils, caused by preferential clearing on fertile soils until 1972, probably caused this accelerated deforestation on the former soils. However, the preference to clear fertile soils remained, and is reflected by the deforestation rates calculated by equation 2 (Table 3). In 1984 the deforestation rates for fertile, moderately fertile and unfertile soils were 19.6, 14.6 and 13.6 per cent per year, respectively. These numbers indicate that one of the reasons to clear was the search for agricultural land, since fertile soils were cleared preferentially. The colonizers probably knew how to distinguish between fertile and unfertile soils. More recently poor soils are also being cleared, because land is becoming more scarce.

### *Accessibility*

One of the most important factors to have influenced the deforestation history of the area is accessibility. Before 1960 there was hardly any infrastructure.

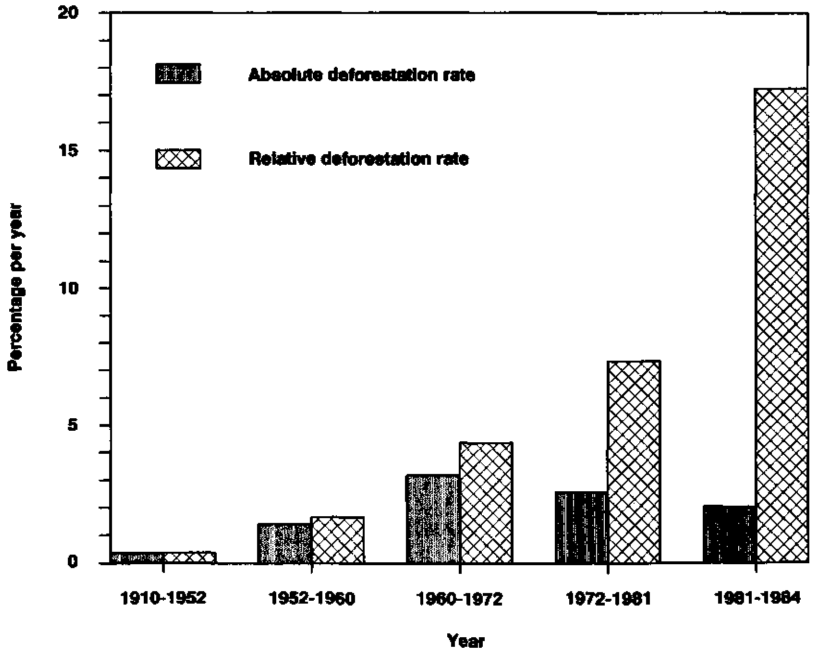

Figure 6. Deforestation rates in the Study Area as calculated by equations 1 and 2

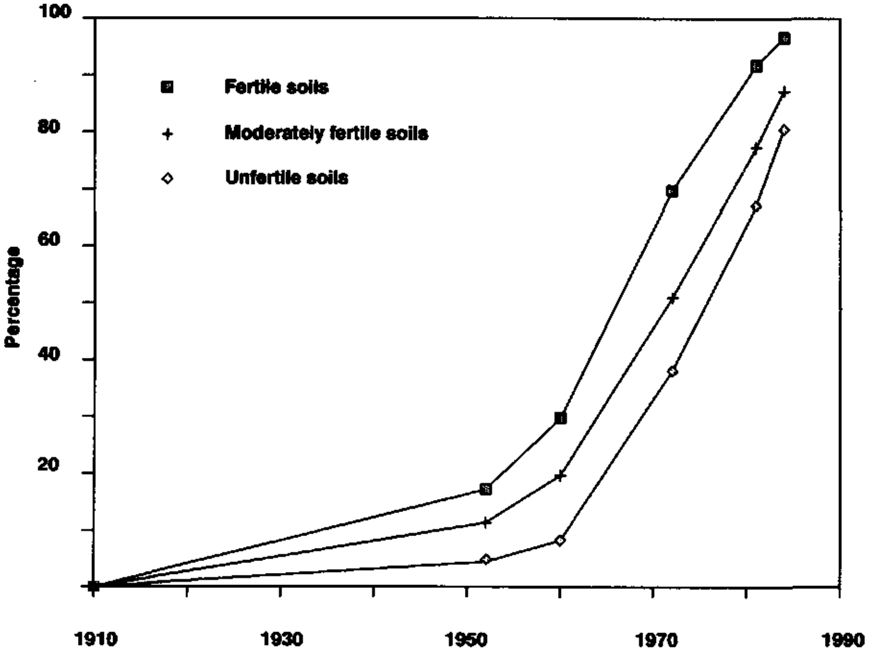

Figure 7. Cumulative deforestation in time of fertile, moderately fertile and unfertile soils

Only rivers offered easy access to the forest. The area <500 m from the major rivers covers 8,874 ha (22.4%), the area >500 m from the main rivers covers 30,662.5 ha (77.6%). Cumulative deforestation (in per cent) is shown for both areas in Figure 8. In this figure the slope of the lines is a measure of the deforestation rate. Especially in the period until 1960 the slope for the <500 m area is steeper than that for the >500 m area. This indicates that deforestation took place preferentially place along the major rivers. Although the percentage of good soils in the <500 m area is slightly higher (86 per cent) than in the >500 m area (79 per cent) this cannot explain the large difference in clearing rate for example in 1960 (Table 4).

After 1972 the slope of the <500 m area line is less steep than the slope of the >500 m area line, because practically no forest area was left close to a river. In 1984 the lines of both areas intersect, indicating that the percentage cleared was equal in both areas.

In the period 1960-1972 the main roads were constructed and access to the forest was shifted from rivers to roads. Forest clearing in the areas <250 m and >250 m from the main roads is shown in Table 5. The <250 m area covers 12,766.5 ha (24.3%); the >250 m area covers 39,733.5 ha (75.7%). The period

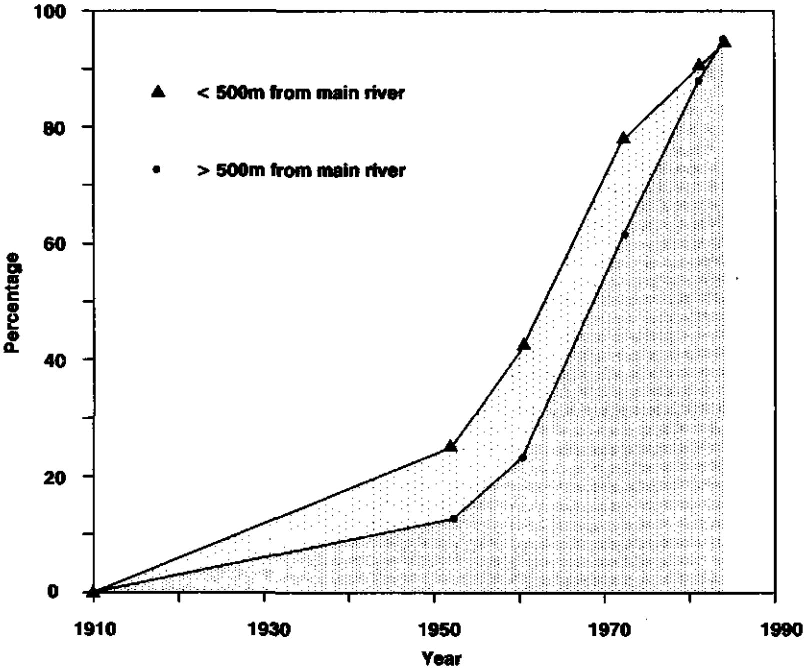

Figure 8. Cumulative deforestation in time of area within 500 m of main rivers and of area more than 500 m from main rivers

until 1960 is not considered as most roads were constructed later. Especially in the year 1972 the deforestation pattern followed the roads. In the <250 m area deforestation was 67.3 per cent whereas in the >250 m area 46.6 per cent of the forest area was cleared. Later the 'road-effect' declined, but was still present in 1984.

The same sequence in the colonization process: access by rivers, followed by access by roads was observed by Brooijmans & van Sluys (1990) for the Cocorí area. Sader & Joyce (1988) also concluded that new road construction into forest regions increased the likelihood of deforestation.

### *Banana plantations*

Between 1960 and 1972 large banana plantations were introduced into the study area. In 1972 the area covered with banana was 8,234 ha. The majority of these plantations (71.5 per cent) was created on newly cleared areas. The remaining 28.5 per cent was situated on areas cleared before 1960. The banana producers selected the locations for their plantations: 92.0 per cent were situated on fertile

**Table 3.** Deforestation area and rate (eq. 2) on different soil classes.

| Year | Fertile soils | | Mod. Fertile soils | | Unfertile soils | |
|---|---|---|---|---|---|---|
| | area ha | rate % $y^{-1}$ | area ha | rate % $y^{-1}$ | area ha | rate % $y^{-1}$ |
| 1910 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| 1952 | 5482.4 | 0.4 | 558.7 | 0.3 | 122.5 | 0.1 |
| 1960 | 3980.8 | 1.9 | 406.3 | 1.2 | 104.7 | 0.5 |
| 1972 | 12785.7 | 4.8 | 1541.7 | 3.3 | 822.6 | 2.7 |
| 1981 | 6990.8 | 8.1 | 1298.0 | 6.0 | 803.8 | 5.2 |
| 1984 | 1542.4 | 19.6 | 487.4 | 14.6 | 370.6 | 13.6 |

soils, while 80.6 per cent of the area consists of fertile soils. Besides the fertile soils, the banana producers planted on 5.1 per cent the moderately fertile and on 2.9 per cent the unfertile soils. Between 1972 and 1981 the changes in banana area showed a shift towards increased utilization of the best soils. Although in some areas plantations were abandoned, in other areas they were enlarged. As a result, in 1981 the plantations were situated almost completely on fertile soils: of the area under banana in 1981 (7,487 ha) the percentages on fertile, moderately fertile and unfertile soils were 94.1 per cent, 4.3 per cent and 1.5 per cent, respectively. In the period 1960-1972 the plantations were not responsible for the bulk of the clearing itself. The secondary effects of the banana plantations were probably more important. An extended infrastructure was constructed, influencing accessability, and the plantations attracted a lot of people, looking for work.

The deforestation rates we calculated in equation 2 are not directly comparable to those of Sader & Joyce (1988), because the intervals of the period studied

**Table 4.** Deforestation in area within 500 m from main rivers (< 500 m) and in area more than 500 m from main rivers (> 500 m).

| | < 500 m | | > 500 m | |
|---|---|---|---|---|
| Year | area ha | rate % $y^{-1}$ | area ha | rate % $y^{-1}$ |
| 1910 | 0.0 | 0.0 | 0.0 | 0.0 |
| 1952 | 2218.3 | 0.6 | 3945.3 | 0.3 |
| 1960 | 1456.3 | 2.7 | 3035.5 | 1.4 |
| 1972 | 3166.3 | 5.1 | 11983.8 | 4.2 |
| 1981 | 1150.8 | 6.3 | 7941.8 | 7.5 |
| 1984 | 325.0 | 12.3 | 2075.3 | 18.4 |

**Table 5.** Deforestation in are within 250 m of main roads (< 250 m) and at more than 250 m from main roads (> 250 m)

| | < 250 m | | > 250 m | |
|---|---|---|---|---|
| Year | ha | cum % | ha | cum % |
| 1972 | 5470.5 | 67.3 | 9679.5 | 46.6 |
| 1981 | 1941.8 | 91.2 | 7150.8 | 81.1 |
| 1984 | 528.3 | 97.7 | 1872.0 | 90.1 |

were different. However, some conclusions can be drawn when looking at Figure 9. For the period until 1960 our rates (1.7% $yr^{-1}$) are lower than the rates as calculated by Sader & Joyce for the two life zones in their study area (4.8 and 3.6 %). Ever since 1972 our rates are comparable to the national deforestation rates, because the bulk of forest clearing in Costa Rica took place in the Atlantic Zone.

### *Evaluation of the use of aerial photographs for deforestation studies.*

Written documentation about forest coverage is often incomplete and does not include detailed maps. Satellite images cover only the most recent period of deforestation. Aerial photography can overcome the aforementioned problems. When available, they usually cover a longer period than satellite images do and have more detail than written records. The availability of aerial photographs is important when our method is used. This is clearly demonstrated by the parts with "incomplete information" in Figure 4.

In the present study the information of the aerial photographs was processed digitally. This process is rather time consuming, but is the only practical quantitative method.

The accuracy of the deforestation maps depends on the scale and quality of the aerial photographs. Interpretation of aerial photographs was done with a 0.5 mm pencil allowing a maximum deviation of 0.5 mm when making the interpretation. Deviations expressed in meters therefore range from 12.5 m at scale 1:25,000 to 40 m at scale 1:80,000. The accuracy of the monoplotting transformations depends on the accuracy of the coordinates of the ground control points. The terrain coordinates of the ground control points of the older aerial photographs were therefore calculated by digitizing them on the 1984 photo's and transforming them.

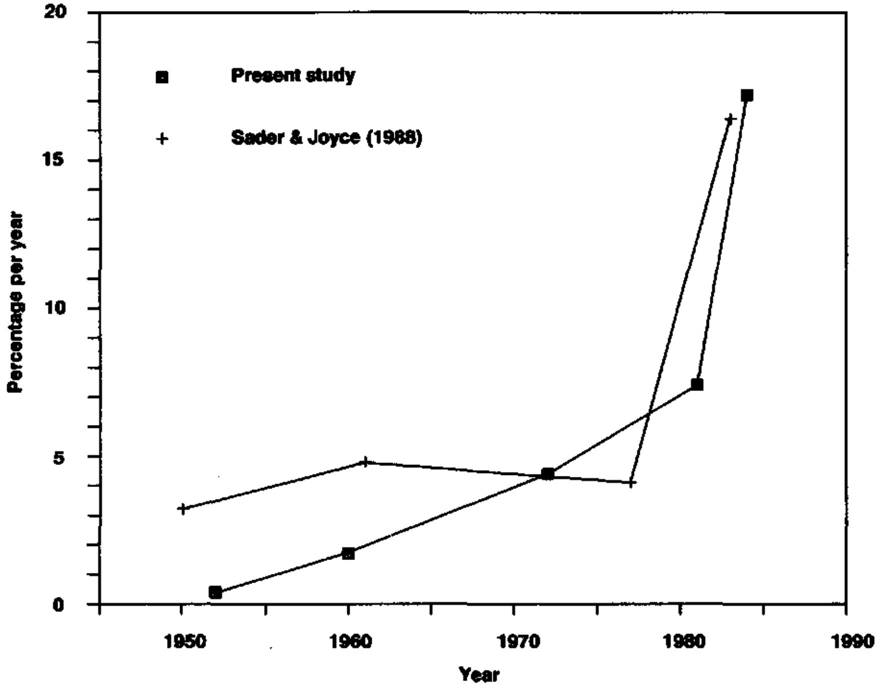

Figure 9. Deforestation rates from our study related to deforestation rates for Costa Rica as presented by Sader and Joyce (1988)

For the 1952 and 1960 aerial photographs it was difficult to find ground control points which were also on the 1984 photo's. Marks chosen were curves of rivers and land parcel limits. As both can change with time the accuracy of these deforestation maps is less than the more recent maps.

## Conclusions

Aerial photographs offer a good means of analyzing deforestation patterns. They usually have more detail and cover a longer period than satellite images do. The method used in this study has good possibilities for use in larger areas and other countries. In Costa Rica aerial photo coverages on national scale exist for 1952, 1960 and 1984. For other years also photo series exist for large areas of the country. We have no detailed information of photo coverages outside Costa Rica, however, in most countries the first systematic flights were made in the 1940s or early- 1950s.

In the study area deforestation took place preferentially along the main rivers, especially in the first stage of deforestation (before 1960), when the forest was only easily accessible by boat. Later, when roads became available the importance of rivers to the process of deforestation decreased sharply.

The search for agricultural land was one of the reasons for forest clearance, as is shown by preferential deforestation on fertile soils. After 1981, relative unfertile soils were also cleared, because hardly any fertile soils were left under forest. The introduction of banana plantations to the study area stimulated forest clearing because a road network was constructed, which made access to the forest easier.

Deforestation trends as described in this study should be extrapolated with great care. The Atlantic Zone of Costa Rica has a completely different climate and colonization history from the rest of Costa Rica. Yet, the general processes described will probably also hold true for other areas, although local conditions will cause considerable variation to occur.

## Acknowledgments

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## *Chapter 3*

# Organic carbon input to soil by grass roots in the humid tropics of Costa Rica.

*Submitted to: Plant and Soil*

E. Veldkamp

# Organic carbon input to soil by grass roots in the humid tropics of Costa Rica.

## Abstract

In the Atlantic Zone of Costa Rica, soil organic matter levels decrease in pastures established after clearing of humid tropical forest, making these pastures a net source of atmospheric $CO_2$. The objective of this study was to estimate the input of organic C to the soil in native, low-yield pastures and to test the hypothesis that improvement of these pastures by introducing better species, will increase input of carbon to the soil. Field labelling with a pulse of $^{14}CO_2$ was done on fully developed plants of two grass species: *Axonopus compressus*, a low-yield species, and *Brachiaria dictyoneura*, an improved grass with high yield.

Annual below-ground dry matter production of Brachiaria (12 Mg ha$^{-1}$ yr$^{-1}$) was about twice that of Axonopus (6 Mg ha$^{-1}$ yr$^{-1}$). During the seasonal decline in above ground production, below ground production increased for both species. In the wet season root growth was mainly concentrated in the top 0.05 m of the soil. In the dry season root growth shifted to deeper layers. Root mass of Brachiaria was about three times the root mass of Axonopus. The discrepancy between root mass and root growth implies a difference in residence time. The residence time of carbon in living roots was 1.3 yr for Axonopus and 2.0 yr for Brachiaria. Residence time increased with depth.

Below ground production is an important factor in the carbon balance of a pasture. Introduction of improved grass species in combination with good management increases the organic carbon input into the soil.

**Key words**: andosol, $^{14}C$, improved pasture, pulse labelling, root growth, root turnover, soil organic matter.

## Introduction

Worldwide, vast areas of tropical rainforest are rapidly converted into low productive, grazed pastures (Fearnside, 1987). As a consequence, between 0.42 and 1.60 Pg C of $CO_2$ may have been released from tropical areas in 1980 (Detwiler & Hall, 1988). Of the latter amount, 0.1-0.3 Pg C was attributed to decreases in soil organic matter content which is 3 to 8 % of the annual increase in atmospheric $CO_2$ (3.6 Pg C). In the Atlantic Zone of Costa Rica, Veldkamp (in press) studied the soil organic carbon dynamics under native pastures on three different soil types using the $\delta^{13}C$ method (Balesdent et al., 1987). Deforestation, followed by 25 years of pasture, resulted in a net soil organic carbon loss of 2 to 18 %.

In the Atlantic Zone of Costa Rica, native grasslands mainly consist of naturalized species with low dry matter production which are geared for survival (e.g. *Axonopus compressus; Ischaeum ciliare*). Technically, yield of these grasslands can be improved by sowing better species (grasses and legumes) and by application of fertilizers. Recently, several grass-legume mixtures, containing *Brachiaria* spec., have been successfully introduced in this area (Ibrahim et al., 1993). The objective of this study was to estimate the input of organic C to the soil by native, low-yield pastures and to test the hypothesis that improvement of these pastures by introducing better species, will increase input of carbon to the soil.

In grasslands, below ground net primary production is a major source of organic carbon to the soil, which is estimated to range from 40 to 85 % of total net primary production. Several methods exist to estimate root production: maxima-minima methods, root ingrowth and root observation, which are all subject to a variety of problems and assumptions (Singh et al., 1984; Milchunas and Lauenroth, 1992). $^{14}C$ tracer techniques have also been used to estimate root production in grasslands. The $^{14}C$ dilution technique, which is based on the reduction of the $^{14}C/^{12}C$ ratio after pulse-labelling when plants assimilate only new $^{12}C$, gave anomalous results, mainly due to nonuniform labelling of tissue age classes (Milchunas and Lauenroth, 1992). Pulse labelling with $^{14}C$ has also been used to trace and quantify the distribution of assimilates and the amount of carbon added to the soil (e.g. Martin and Kemp, 1986; Keith et al., 1986). In their study, Keith et al. (1986) used a mean shoot growth curve and the distribution of the $^{14}C$ over shoots, roots, soil and respired $^{14}CO_2$ to calculate the amount of carbon that was added to the soil by wheat plants. In the present study the same approach was used to compare the root growth and turnover of two different grass species.

## Materials and methods

The experiment was carried out on the Experimental Station "Los Diamantes" at Guápiles in the Atlantic Zone of Costa Rica. The soil is a well-drained loamy Eutric Hapludand (Soil Survey Staff, 1990) or Umbric Andosol (FAO, 1988). The site is located on the foot slopes of the Turrialba volcano at an elevation of 110 m. Conversion from forest to pasture occurred about 25 years ago. The climate is characterized by a constant high temperature (about 26 °C) with relatively small diurnal and annual variations. At the meteorological station La Rita, situated within 1000 m from the experimental site, mean annual precipitation is 3400 mm. In all months, the mean rainfall exceeds the mean evaporation. The total rainfall surplus is about 2600 mm per year. January to April and September are relatively dry months with a rainfall surplus of less than 200 mm.

Two grasses were selected for the experiment. *Axonopus compressus* represents the native pastures of the Atlantic Zone of Costa Rica. Axonopus is a perennial stoloniferous grass of low statue. Dry matter production of Axonopus, measured at "Los Diamantes" on the same soil was 11.6 Mg $ha^{-1}$ $yr^{-1}$ (CATIE, 1989). The other species, *Brachiaria dictyoneura CIAT 6133*, represents the improved pastures. Brachiaria is a perennial stoloniferous, erect branching grass of medium height. Dry matter production of *Brachiaria* at "Los Diamantes" was 25.3 Mg $ha^{-1}$ $yr^{-1}$ (CATIE, 1989). Comparable dry matter yields for *Brachiaria dictyoneura* were measured in a savanna area in Colombia (Grof, 1985).

Thirty PVC tubes (0.159 m ID, 0.50 m long) were pushed into the soil. They were inserted with as little soil profile disturbance as possible. The tubes were arranged on an incomplete grid with a distance of 0.7 m. The area between the tubes was kept free of vegetation. Fifteen columns were planted with young *Axonopus compressus* plants, the rest with young plants of *Brachiaria dictyoneura*. The young plants of both species were obtained from stolones collected in actively grazed pastures. The plants were put into the tubes in May 1990. Three months later both species had completely covered the soil columns. The grass was then cut every four weeks, to simulate intensive grazing. After an establishment period of another four months, the subsequent cuttings were used to estimate the shoot growth curve. To compare the results with CATIE

(1989), we followed the same cutting procedure: cutting was repeated every four weeks throughout one year; Axonopus was cut at a height of 0.05 m above soil surface level and Brachiaria was cut at 0.10 m above soil surface level.

At five times, plants of each species in three columns were pulse-labelled with $^{14}CO_2$. The times of labelling were chosen in such a way that they covered both wet and drier seasons. Labelling was done two weeks after the periodic harvest in the morning of a sunny day when plants were photosynthesizing actively. The plant was completely covered with a clear plastic bag, which was sealed to the PVC tube. In a rubber-sealed vial containing dissolved $Na^{14}CO_3$, $^{14}CO_2$ was released by injecting an excess of hydrochloric acid. After gentle shaking, the vial was connected to the plastic bag and the $^{14}CO_2$ label was released into the bag. Label remaining in the vial was transferred into the bag by injecting extra air through the vial into the plastic bag. Per column 1.95 MBq $^{14}C$ was applied. Two hours later, unlabelled $CO_2$ was added in the same way to maintain the photosynthetic activity of the plant. Four hours after the first labelling the plastic bags were removed and plants were left in the open air and allowed to grow normally. Two weeks after label application, the plants were harvested. First, shoots were cut simulteneously with the four week's harvest. Then, the remaining shoots were clipped of at soil surface level. Next, the columns were dug out and transported to the laboratory.

The harvested material of the 13 four week's cuttings was collected and oven-dried at 75 °C for 24 h to determine dry weight. The amount of shoot material harvested per tube was converted to g $m^{-2}$. After a pulse labelling, the PVC tube was removed in the laboratory and the soil column was cut into 0.05 m layers for the 0 tot 0.2 m depth, and 0.1 m layers for the 0.2-0.5 m depth. The top layer was sliced into 0.0-0.01 m and 0.01-0.05m layers. Each sample was crushed manually and the roots were separated from the soil by hand picking. Dry weights of shoots, roots and soil were determined after drying at 75 °C for 24 h. After weighing, the dried soil and plant was stored in sealed plastic bags and transported to the Netherlands for $^{14}C$ analysis.

The dried root and shoot material was weighed and finely milled (<1 mm). Total C and $^{14}C$ in plant material and soil were determined using a wet-combustion method (Amato, 1983). The combustion mixture contained 1 g $K_2Cr_2O_7$ and 5 ml of a mixture of concentrated $H_2SO_4$ and $H_3PO_4$ (60/40). Digestion was carried out in tubes with a screw cap at 160 °C for 2 h. $CO_2$ evolved was trapped in NaOH in a vial placed just under the cap of the digestion tube. Total C and $^{14}C$ were determined by titration and scintillation counting, respectively.

Root biomass was estimated in the labelled tubes and in adjacent, older pastures which did not grow in tubes. The Axonopus pasture had an age of 25 years. The only available Brachiaria pasture had an age of 2 years. Samples (in duplicate) of about 1.5 kg were taken in the adjacent pastures at 0.05 m intervals in depth. The roots were separated from the soil by hand picking. Total dry weights of roots was determined after drying at 75 °C for 24 h. The root mass per kg soil was converted to root mass per volume soil using the bulk density profiles.

For calculation purposes the assumption was made that all label was released as the acid was supplied in-excess. Furthermore it was assumed, that essentially all applied $^{14}CO_2$ was photosynthesized after 2 hours before extra $CO_2$ was injected, because both grass species are C-4 plants with a $CO_2$ compensation point between 0 and 10 ppm (Waller and Lewis, 1979). Under these assumptions, the difference between the amount $^{14}C$ added and the amount of $^{14}C$ recovered can be considered as the sum of root, shoot and microbial respiration of labile root exudates and other root material during the 14 days between labelling and sampling. This difference is called 'respiration'. Root growth was calculated as the amount of activity (Bq) recovered in the root divided by the specific activity of the newly grown shoot between labelling and harvest. Several assumptions are made in the root growth calculation which will be discussed later. Root exudation was calculated as the multiplication of $^{14}C$ recovered in the soil and the soil mass.

The root biomass turnover rate ($yr^{-1}$) was calculated, using a steady state mass-balance approach, assuming that annual root biomass production equals annual root turnover. Root biomass turnover rate k ($yr^{-1}$) can then be obtained by dividing root biomass production by average root biomass. The residence time ($\tau$) for roots is 1/k (Frissel, 1981).

Significant effects were investigated with a full factorial analysis of variance. The influence of grass species and precipitation on production and distribution of $^{14}C$ was evaluated. The relation between shoot and root growth was investigated with regression analysis.

## Results

### *Shoot growth*

Shoot production showed a clear seasonal pattern for both Axonopus and Brachiaria (Figure 1). On day number 290, the grass was cut, but dry matter

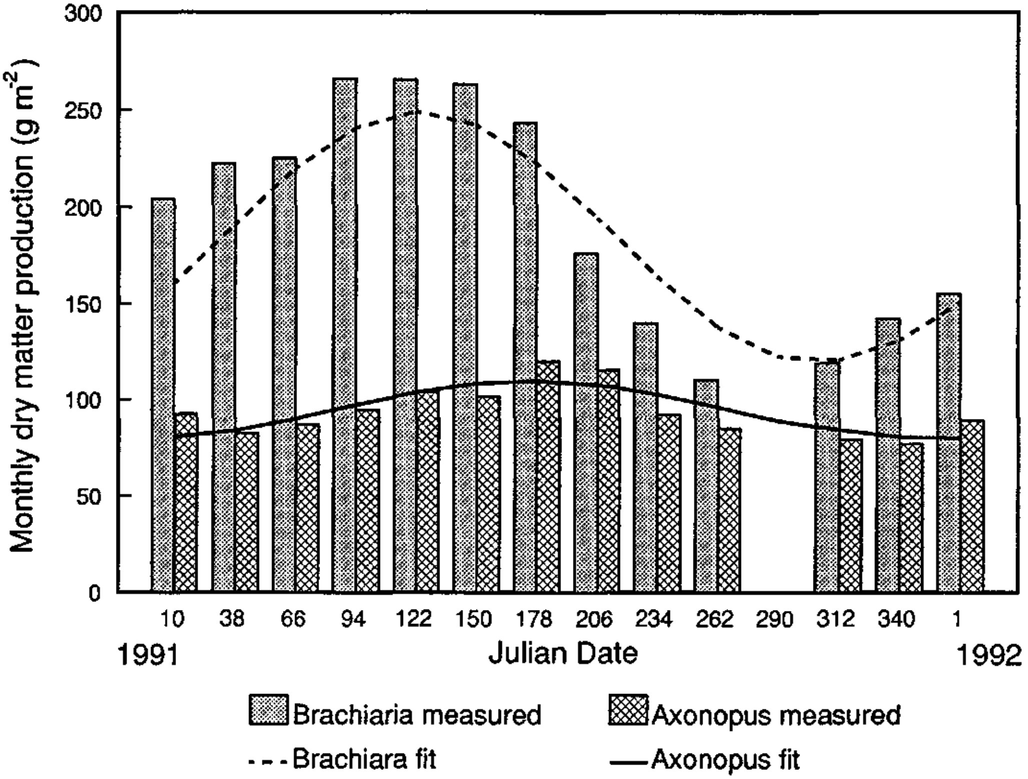

Figure 1. Above ground dry matter production of *Axonopus compressus* and *Brachiaria dictyoneura* in time.

production was not measured. Production of Brachiaria was about twice that of Axonopus, in agreement with observations by CATIE (1989). Brachiaria had a significantly higher shoot production than Axonopus throughout the year (Table 1). For both species production peaked in the beginning of the wet season and was minimal at the end of the wet season. Because production showed a clear seasonality, the dynamics in aboveground dry matter production during the year were described for both species by sinus functions (Figure 1). The sinus function for Axonopus had a $R^2$ of 0.72, the $R^2$ of Brachiaria was 0.67.

### *Distribution of $^{14}C$*

The distribution of added $^{14}C$ was expressed as a percentage of $^{14}C$ applied (Table 2). Losses of $^{14}C$ by respiration were significantly higher for Axonopus (68.2 - 74.9 %) than for Brachiaria (46.3 - 58.8 %). Brachiaria invests a larger fraction of fixed C in the shoot (35.1 - 49.2 %) than Axonopus (19.4 -25.9 %).

**Table 1.** Dry matter production of *Axonopus compressus* and *Brachiaria dictyoneura.*

| Treatment | | Production | | |
|---|---|---|---|---|
| Species | Precip. | Shoot | Root | |
| | | Total | Total | > 0.2 m |
| | (mm) | | (g $m^{-2}$ $month^{-1}$) | |
| A[a] | <100 | 69 | 49 | 16 |
| B[a] | <100 | 225 | 129 | 32 |
| A | 100-200 | 65 | 62 | 21 |
| B | 100-200 | 176 | 86 | 21 |
| A | >200 | 87 | 47 | 8 |
| B | >200 | 299 | 80 | 13 |
| F-test probabilities | | | | |
| Species | | <0.01 | 0.01 | 0.10 |
| Precip. | | 0.02 | 0.30 | 0.04 |
| Spec * Prec | | 0.10 | 0.13 | 0.24 |

[a] A, B refer to Axonopus and Brachiaria respectively

**Table 2.** Distribution of $^{14}C$ (expressed as % of total label applied) for *Axonopus compressus* and *Brachiaria dictyoneura.*

| Treatment | | Distribution of $^{14}C$ | | | | |
|---|---|---|---|---|---|---|
| Species | Precip. | Shoot | Root | Soil | Resp. | Ratio[b] |
| | mm | % | | | | |
| A[a] | <100 | 19.4 | 5.7 | 0.0 | 74.9 | 3.9 |
| B[a] | <100 | 35.1 | 6.0 | 0.1 | 58.8 | 6.3 |
| A | 100-200 | 23.0 | 8.7 | 0.1 | 68.2 | 3.1 |
| B | 100-200 | 35.8 | 5.7 | 0.1 | 58.4 | 6.4 |
| A | >200 | 25.9 | 5.8 | 0.1 | 68.2 | 5.1 |
| B | >200 | 49.2 | 4.5 | 0.0 | 46.3 | 12.6 |
| F-test probabilities | | | | | | |
| Species | | <0.01 | 0.15 | 0.99 | <0.01 | <0.01 |
| Precip. | | <0.01 | 0.21 | 0.17 | 0.03 | 0.01 |
| Species * Prec. | | 0.25 | 0.29 | 0.90 | 0.19 | 0.20 |

[a] A, B refer to Axonopus and Brachiaria respectively. [b] Ratio is Shoot/root production.

Brachiaria appears to invest a smaller fraction in the roots (4.6 - 6.0 % compared to 5.7 - 8.7 %), although the differences are not significant. The Shoot/Root ratio of recovered $^{14}C$ is significantly different for both species ($P < 0.01$) and for the wetter (> 200 mm rain) and drier (< 200 mm) periods ($P < 0.01$). For Axonopus, this ratio is 5.1 in the wet period and 3.1 to 3.9 in the dry period. For Brachiaria the Shoot/Root ratio is 12.6 in the wet period and 6.3 to 6.4 in the dry period. The percentage of fixed $^{14}C$ recovered in the soil is low (about 0.1 %) for both species. Brachiaria caused a relatively large input of $^{14}C$ in the shallow part of the root system, whereas Axonopus transferred a larger part of root $^{14}C$ to deeper layers. For both grasses, more $^{14}C$ is recovered at greater depth in the dry season than in the wet season (Table 3).

### *Root growth, biomass and turnover*

The amount of photosynthetically fixed carbon, transferred to the roots was calculated for all individual labelled plants. From these data a linear relation between above-ground dry matter production and the ratio between above- and below-ground production was calculated for both Axonopus ($R^2 = 0.58$) and Brachiaria ($R^2 = 0.56$). For both species the ratio between above- and below ground production increases when shoot production increases. This means that

**Table 3.** Distribution of $^{14}C$ with depth (expressed as % of total recovered in roots)[a] in the soil columns for *Axonopus compressus* and *Brachiaria dictyoneura* in the wet and dry season.

| | *Axonopus compressus* | | *Brachiaria dictyoneura* | |
|---|---|---|---|---|
| Soil depth | Wet | Dry | Wet | Dry |
| m | % | | % | |
| 0.00-0.01 | 62.1 (11.3) | 34.9 (10.1) | 68.0 (0.4) | 44.8 (4.6) |
| 0.01-0.05 | 14.9 (4.5) | 21.4 (1.1) | 17.5 (2.1) | 24.5 (3.1) |
| 0.05-0.10 | 6.7 (2.8) | 10.8 (4.4) | 3.5 (0.4) | 9.9 (5.4) |
| 0.10-0.15 | 5.1 (2.8) | 8.9 (2.4) | 1.9 (0.5) | 5.5 (1.7) |
| 0.15-0.20 | 2.3 (1.3) | 8.1 (4.2) | 2.1 (0.5) | 3.2 (0.7) |
| 0.20-0.30 | 3.7 (1.2) | 4.8 (1.8) | 1.7 (0.5) | 3.8 (1.0) |
| 0.30-0.40 | 2.6 (0.7) | 4.5 (0.9) | 2.4 (0.9) | 2.3 (0.7) |
| 0.40-0.50 | 2.0 (0.9) | 5.0 (1.2) | 1.8 (0.2) | 4.3 (1.8) |
| 0.50- | 0.7 (0.4) | 1.7 (0.2) | 1.1 (0.4) | 1.6 (0.9) |

[a] Figures represent averages of 3 replicates, standard deviations in parentheses.

both species invest more in their roots when shoot production declines. This effect was more pronounced for Brachiaria than for Axonopus. Root growth of Brachiaria is more than twice the root growth of Axonopus, given the same above ground production, which demonstrates Brachiaria directs a higher percentage of the photoassimilates below ground. The relation between shoot dry matter production and the ratio between above- and below ground production was combined with the monthly shoot dry matter production numbers (Figure 1) to calculate the root growth in time. For Axonopus root production was practically constant throughout the year, with a small increase in the dry period. Monthly root dry matter production ranged from 44 to 48 g $m^{-2}$. For Brachiaria the monthly root dry matter production varied from 87 to 104 g $m^{-2}$. The annual root growth at different depths was estimated by adding up the monthly root growth numbers.

Root biomass in the tubes between 0.10 and 0.20 m depth was smaller than root biomass in adjacent pastures (Table 4). Below 0.4 m depth root biomass in the tubes was larger. For Axonopus, the total biomass in the tube was 770 g $m^{-2}$ compared to 760 g $m^{-2}$ in the field situation. For Brachiaria this was 2270 g $m^{-2}$ and 2590 g $m^{-2}$, respectively.

Using root biomass numbers from adjacent pastures, the residence time, $\tau$, was calculated for both Axonopus and Brachiaria. For Axonopus $\tau$ was 1.3 yr, for Brachiaria $\tau$ was 2.0 yr. Residence time was also calculated at different depths. The $\tau$ in the top 0.00 to 0.05 m layer is for both species lower than in deeper layers (Figure 2).

**Table 4.** Root biomass measured in tubes and in field situation

| Depth | *Axonopus compressus* | | *Brachiaria dictyoneura* | |
|---|---|---|---|---|
| | Tube | Field | Tube | Field |
| (m) | (g $m^{-2}$) | | | |
| 0.00-0.05 | 444 | 387 | 1321 | 1387 |
| 0.05-0.10 | 82 | 94 | 197 | 290 |
| 0.10-0.15 | 52 | 97 | 98 | 203 |
| 0.15-0.20 | 35 | 53 | 75 | 137 |
| 0.20-0.30 | 58 | 60 | 138 | 218 |
| 0.30-0.40 | 43 | 51 | 132 | 199 |
| 0.40-0.50 | 35 | 6 | 142 | 112 |
| 0.50- | 29 | 4 | 164 | 144 |

## Discussion

This experiment was conducted on a fertile, well drained Eutric Hapludand which explains the relatively high dry matter production numbers. Production will probably decrease on soils of lower fertility. However, dry matter production numbers as high as 2000 g $m^{-2}$ $yr^{-1}$ have been reported for *Brachiaria dictyoneura* in a grass-legume mixture on an infertile oxisol in Colombia (Grof, 1985). As is the case with any method which is used to estimate root production and turnover, a number of assumptions are involved in this pulse label experiment. An inherent assumption of the root growth calculations is that there is a 'typical' specific activity for all newly grown parts of the plant. Directly after clipping a considerable amount of C will be translocated from roots to shoots, untill sufficient leaf area is restored. In the experiment it is assumed that two weeks after clipping, this translocation is negligible. If this assumption is not true, this will lead to an over estimation of shoot growth. Another assumption is that two weeks after labelling, allocation of $^{14}C$ has stabilized. Dinwoodie and Juma (1988) found no significant differences in $^{14}C$ distribution between shoots and roots from pulse labelled barley harvested 10, 28, 49 and

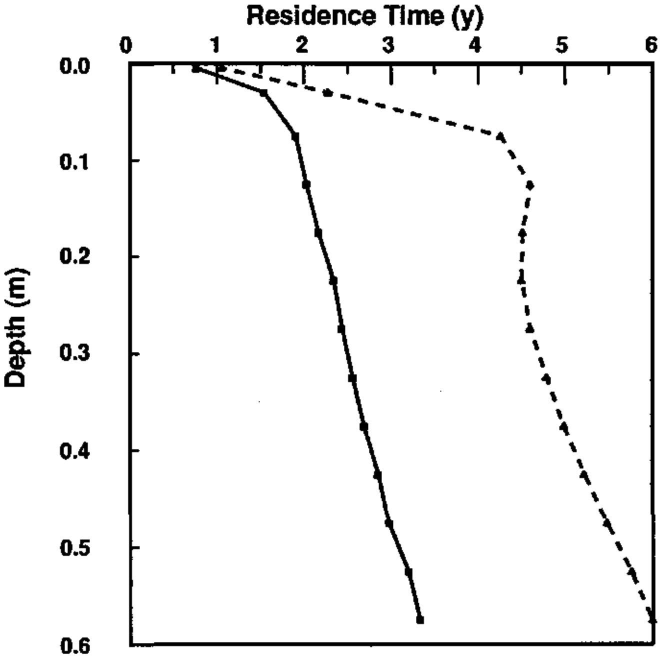

Figure 2. Residence time (τ) of carbon in roots with depth of profile for *Axonopus compressus* (line) and *Brachiaria dictyoneura* (dashed line)

70 days after label application. In contrast, Milchunas and Lauenroth (1992) found that stabilization of labile $^{14}C$ after pulse labelling of a short grass steppe, required one growing season. Other assumptions are: root restriction in tubes will not alter allocation patterns; the labellings are representative for carbon allocation through the year; below ground herbivory is minimal. Bias due to any of these assumptions will be relatively the same between the two species. Therefore, results of this pulse label experiment should be used more in a comparative way than as absolute growth and turnover rates.

The shoot dry matter production, extrapolated from measured production in the tubes to $m^{-2}$ was higher than the production measured by CATIE (1989) in field plots. CATIE collected data on the same soil type at the same experimental station with the same cutting procedure. The higher production in the tubes may have had several causes: the lack of competition for space outside the tubes and the soil cover inside the tube was close to 90 % whereas the ground cover in the pasture is typically about 40 % (Ibrahim, pers. com.). To obtain realistic dry matter production data, in agreement with the data of CATIE (1989), the production data were multiplied by 0.65 to correct for this 'tube effect'. No significant differences were found between shoot weights of labelled and unlabelled plants in tubes (at the 95 % confidence level) which indicates that label application did not affect plant growth. Pulse labelling has been applied to other pasture grasses. Meharg and Killham (1988) applied $^{14}C$ to perennial rye-grass (*Lolium perenne*) and harvested after 8 days. They found that 26.6 % of the label was recovered in the shoot, 6.9 % in the root and 0.7 % in the soil. This is similar to the distribution of $^{14}C$ for Axonopus in the wet season (Table 2). It confirms that a considerable part of the photo-assimilated $^{14}C$ is respired within a few days after labelling. With the present data it is not possible to estimate the contribution of root exudates in the total C balance. However, root exudation seems to be of little direct importance, because only a minor part of the assimilated $^{14}C$ was recovered in the soil.

The main difference between the present experiment and pulse-labelling of annual crops (e.g. Keith et al., 1986) is the monthly harvest of the grass plants. Defoliation strongly influences Shoot/Root ratios. Turner et al. (1993) found that aboveground production of a tallgrass prairie vegetation was maximized by the most frequent mowing treatment. Bushby et al. (1992) found that cutting markedly increases in the Shoot/Root ratio of production in a *Panicum maximum* pasture. In their experiment, the Shoot/Root ratio of labelled $^{14}C$ in an unfertilized treatment increased from 2.4 to 7.0 if the pasture was cut, compared to non-cutting. In the present study, Shoot/Root ratios between 6.3 and 12.6 for Brachiaria and between 3.1 and 5.1 for Axonopus were measured. These results are only valid for intensively grazed pastures and should not be extrapolated to ungrazed conditions. Shoot/Root ratios of both grass species are strongly related to the amount of rainfall. For both pastures the low Shoot/Root ratios occurred in the dry season and the high ratios in the wet season (Table 2). The change in Shoot/Root ratios illustrates that in the dry season the pastures invest more in their root system and less in the shoots to decrease evapotranspiration and increase water uptake.

For the calculation of root turnover of Brachiaria a two year old pasture was used. This pasture was probably not yet in steady state especially at greater depth. At steady state situation, root mass and consequently residence time will probably be larger. The residence time, $\tau$, for roots of Brachiaria may thus be underestimated. The $\tau$ of C in the roots of Axonopus and Brachiaria showed remarkable differences. The $\tau$ of Brachiaria (2.0 yr) is about 50 % higher than the $\tau$ of Axonopus (1.3 yr). The low $\tau$ values in the top layer (Figure 2), may reflect a large contribution of fine roots and root hairs to the total root mass in the top 0.1 m layer. Fine roots and root hairs may have higher turnover rates than larger roots. More herbivores and decomposers in the top layer and greater variability in soil temperature and water may also cause higher turnover rates in the top soil. The contribution of deeper roots to the overall $\tau$ values is relatively small as is reflected by the large difference between $\tau$ values at increasing depth (2-4 yr) and the weighed mean values of $\tau$ (Figure 2).

This study demonstrates the application of labelling with a pulse of $^{14}C$ on two perennial grasses. The method provides useful information, especially on a comparative base. For more absolute information of root growth and root turnover rates, more of the assumptions have to be checked. Input of organic carbon into the soil is about twice as high for the improved species Brachiaria compared to the native species Axonopus. Improvement of the native pastures by introducing better species, will thus increase input of carbon to the soil.

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## *Chapter 4*

# Organic carbon turnover in three tropical soils under pasture after deforestation.

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E. Veldkamp

# Organic carbon turnover in three tropical soils under pasture after deforestation.

## Abstract

Deforestation is one of the main reasons for the global net release of carbon dioxide from soil to atmosphere. Estimates of $CO_2$ emission from soils are highly variable, mainly due to limited data of C dynamics in soils after forest clearing. The objective of this study was to calculate the changes in soil organic carbon (SOC) storage after deforestation in three soil types in the Atlantic Zone of Costa Rica with help of the $\delta^{13}C$ method.

Changes in bulk density, which normally accompany land use changes, had a profound influence on the results of the calculations. Deforestation, followed by 25 years of pasture, caused a net loss of 21.8 Mg $ha^{-1}$ in SOC for an Eutric Hapludand and of 1.5 Mg $ha^{-1}$ for an Oxic Humitropept. SOC changes in time were studied on a deforestation sequence on an Andic Humitropept. In the first years after forest clearing, decomposition of tree roots caused an extra input of soil organic carbon which influenced the $\delta^{13}C$ signal. Decomposition of 'forest C' and increase of 'pasture C' were mathematically described for several depths. A considerable influence of depth on decomposition rates was found. The strong stabilization of organic carbon by Al-organic matter complexes probably caused the relatively small net C loss from SOC since forest clearing.

**Key words**: bulk density, C3, C4, $CO_2$ emission, compaction, Costa Rica, $\delta^{13}C$, deforestation, soil organic carbon.

## Abbreviations:

| | |
|---|---|
| SOC | Soil Organic Carbon |
| $SOC_f$ | Soil Organic Carbon derived from forest (Mg C $ha^{-1}$) |
| $SOC_p$ | Soil Organic Carbon derived from pasture (Mg C $ha^{-1}$) |
| DSOC | Decomposable Soil Organic Carbon (Mg C $ha^{-1}$) |
| PSOC | Passive Soil Organic Carbon (Mg C $ha^{-1}$) |
| TOC | Total Organic Carbon (Mg C $ha^{-1}$) |

# Introduction

Increasing atmospheric concentrations of $CO_2$ and other trace gases will probably affect the global climate in near future. Soils play an important role in the production and consumption of $CO_2$. Soil-vegetation systems can act as a $CO_2$ sink or a $CO_2$ source, depending on decomposition rate and rate of soil organic carbon formation (Van Breemen & Feijtel, 1990). When forest is cleared, the soil turns into a $CO_2$ source.

The annual relative increase in atmospheric $CO_2$ concentration is 0.5 % which corresponds with 3.6 Pg C $y^{-1}$ (Bouwman, 1990). Estimates of global $CO_2$ release caused by deforestation are between 1.0 to 3.2 Pg C $y^{-1}$. Forest clearing as a source of atmospheric $CO_2$, is second only to the release from the burning of fossil fuels (Houghton et al., 1985, Tans et al., 1990). Data from Detwiler & Hall (1988) illustrate the significance of tropical forest clearing. They estimated the $CO_2$ release from non-tropical forest regions in 1980 to be at the most 0.1 Pg C and for tropical forests to be 0.42 to 1.60 Pg C. Of this amount, 0.1 to 0.3 Pg C were attributed to decreases in soil organic carbon content.

The decrease in soil organic carbon content depends on the land use established after forest clearing. Detwiler (1986) estimated that cropping of tropical forest soils reduces their carbon content by 40 %; the use of these soils for pastures reduces it by about 20 %.

The capacity of soils to sequester carbon is considerable. Soils developed in volcanic ash (Andisols) have a very high organic matter storage capacity, due to the amorphous components which stabilize organic matter (Mizota & van Reeuwijk, 1989). Boudot et al. (1986) found a negative correlation between the carbon mineralization rate in Andosols and amorphous Al, Fe and allophane. In spite of the stabilizing effect of amorphous components in Andosols, considerable SOC losses have been found in a secondary regrowth as compared to a mature tropical forest site (Raich, 1983).

Techniques, based on isotope fractionation of plant material and SOC have stimulated quantitative SOC turnover studies (e.g. Balesdent et al., 1987). Plant carbon contains distinctly less $^{13}C$ than atmospheric $CO_2$ and during decomposition of plant material in the soil, further fractionation takes place. To quantify changes in $^{13}C$, carbon isotope ratios defined as $^{13}R = {}^{13}C / {}^{12}C$ can be used. In practice, $\delta^{13}C$ values, which is the relative difference between isotope ratios of sample and standard, are used: $\delta^{13}C \text{ ‰} = ({}^{13}R_{sample} - {}^{13}R_{standard}) / ({}^{13}R_{standard}) \times 1000$. The international standard is V-PDB (Vienna-Pee Dee

Belemnite) which has a $^{13}R$ of 0.01124 (O'Leary, 1981). Atmospheric $CO_2$ has a $\delta^{13}C$ of -7.5 ‰ (Mook, 1986), values of plant C range from -22 ‰ to -34 ‰ in C3 plants, to around -15 ‰ in C4 plants.

During decomposition of plant material a small enrichment in $^{13}C$ takes place. Generally, $\delta^{13}C$ values of soil organic carbon increase with only 2 to 3 ‰ in depth, compared to the 15 ‰ caused by the change between C3 and C4 vegetation (Balesdent et al., 1987). Therefore, changes from C3 to C4 plants, provide an in situ labelling (Balesdent et al., 1987). Quantification requires detailed information on vegetation history.

In the Atlantic Zone of Costa Rica large areas of tropical lowland forest have been cleared in the last 40 years (Veldkamp et al., 1992). More than 50 % of the cleared area has been put under pastures with low dry matter production, mainly consisting of C4 grasses. Because forests are dominated by C3 species, this shift in vegetation makes it possible to apply the $\delta^{13}C$ method.

The objective of this study was to quantify the changes in soil organic carbon storage after deforestation under pastures with help of the $\delta^{13}C$ method. Based on these changes in soil carbon, an estimate will be made of the total C losses caused by deforestation of three different soil types in the wet tropical lowlands of Costa Rica.

## Materials and Methods

### *Study sites*

The study was carried out in the Atlantic Zone of Costa Rica. The climate is humid tropical; mean annual temperature is 26 °C and mean annual rainfall is between 3000 and 6000 mm. Precipitation exceeds evapotranspiration in all months.

The study area is situated on the foot slopes of the Turrialba volcano, and is covered with laharic, fluvio-laharic and fluvial deposits, mostly of andesitic composition. Tropical lowland rain forest is the natural vegetation. Tosi (1969) distinguished the life zones Premontane Wet Forest and Tropical Wet Forest in the area. After deforestation, one of the dominant land use types is pasture. An important assumption of the $\delta^{13}C$ method is that the soils under pasture are directly comparable to those under forest. For that reason, particular attention has been paid to site selection. A deforestation map was made and combined with an available soil map in a GIS (Veldkamp et al., 1992). Corresponding forest and pasture sites were selected within 1 km. On all sites, clearing was done by small farmers without heavy machinery. Valuable stems were removed along a few tracks, other stems were cut and left on the sites. Generally, stumps were not removed. If burning is done, it is restricted to piles. The location of former burning piles is easily identified by the charcoal left in the soil. Only soil profiles were sampled which showed no morphological indications of significant disturbance or burning during clearing. Pasture grasses are normally planted directly after clearing, without any further land preparation treatment.

Three well-drained soil types were sampled: an Eutric Hapludand, an Oxic Humitropept and an Andic Humitropept (Soil Survey Staff, 1990). The studied soil profiles contained only negligible amounts on rock fragments. In Table 1, some important soil characteristics are summarized. Two sites of the Eutric Hapludand were selected: one under natural forest vegetation and the other site was manually cleared about 25 years ago and put under pasture (*Axonopus compressus*). Two Oxic Humitropept sites were selected: one under forest vegetation, where some trees had been removed and the other under 25 year old pasture (*Axonopus compressus*). On the Andic Humitropept, five sites were selected: one site under natural forest, four other sites were pastures, dominated by *Ischaeum ciliare*, of 3, 5, 10 and about 18 years. On the younger pasture sites decomposing stems and tree stumps were still present.

**Table 1.** Bulk density, Organic carbon, pH and CEC of selected samples of three different soil types under forest and pasture.

| | | Forest | | | | Pasture | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Depth (m) | $\rho_b$† Mg m$^{-3}$ | $C_{org}$ % | pH $H_2O$ | CEC cmol kg$^{-1}$ | $\rho_b$ Mg m$^{-3}$ | $C_{org}$ % | pH $H_2O$ | CEC cmol kg$^{-1}$ |
| Eutric Hapludand | 0.05 | 0.58 | 7.40 | 5.6 | 38.2 | 0.82 | 3.95 | 5.5 | 48.4 |
| | 0.35 | 0.83 | 1.54 | 6.0 | 28.0 | 0.92 | 2.17 | 6.1 | 30.2 |
| | 0.65 | 0.93 | 1.42 | 6.2 | 16.0 | 1.04 | 0.83 | 6.4 | 24.8 |
| Oxic Humitropept | 0.05 | 0.62 | 5.32 | 4.3 | 43.5 | 0.74 | 5.07 | 4.8 | 30.2 |
| | 0.35 | 0.83 | 1.26 | 4.6 | 31.1 | 0.81 | 1.27 | 4.9 | 26.4 |
| | 0.65 | 0.89 | 0.79 | 4.7 | 29.2 | 0.81 | 0.92 | 4.9 | 27.5 |
| Andic Humitropept | 0.05 | 0.66 | 4.81 | 3.9 | 31.9 | 0.97 | 3.60 | 5.4 | 40.7 |
| | 0.35 | 0.72 | 0.99 | 4.4 | 29.5 | 0.87 | 1.02 | 5.1 | 27.5 |
| | 0.65 | 0.76 | 0.75 | 4.6 | 24.2 | 0.85 | 0.67 | 5.0 | 27.5 |

† $\rho_b$ = Bulk density

### *Sampling and sample processing*

Soil bulk density was sampled at five depths ($n \geq 8$, depending on variability) using $0.3 \times 10^{-3}$ $m^3$ stainless steel rings. The samples were oven dried for 24 h at 105 °C.

Disturbed soil samples of about 1.5 kg were taken at 0.05 m intervals to 1.0 m. Soil samples were homogenized, air-dried and sieved to pass 2 mm. Litter and root material were separated manually from the soil. Air-dry soil, litter and root samples were stored for $^{13}C$ analysis.

Organic carbon and $\delta^{13}C$ analyses were carried out in the Isotope Physics Laboratory of the University of Groningen, the Netherlands. To measure the Carbon-13 composition, soil samples were first grounded and treated with 1N HCl to remove any $CaCO_3$. The samples were then oven dried at 80 °C. About 1 g of the samples was burned in the presence of Cu- oxide under pure oxygen at 900 °C. Water vapour was trapped on dry ice. The $CO_2$ and $NO_x$ were trapped with help of liquid air (-186 °C). Oxygen was evacuated and $NO_x$ was reduced to $N_2$ in the presence of Cu. The remaining pure $CO_2$ was trapped with help of liquid air and analyzed on a SIRA 9 mass spectrometer. The laboratory reference was calibrated against V-PDB, using the international standard NBS 19. Analytical precision determined as the standard deviation obtained on different combustions of the same homogenized sample is better than 0.05 ‰ $\delta^{13}C$.

Soil pH was measured by glass electrode in supernatant above a 1:2.5 soil/water suspension. Cation-exchange capacity was determined at pH 7 by $NH_4OAc$. Fe and Al were determined on the disturbed soil samples by AAS in pyrophosphate extracts and in acid oxalate extracts (Mizota & van Reeuwijk, 1989). Clay mineralogy was determined by X-ray diffraction on the clay fraction.

### *Calculations*

Carbon contents of soil samples, Cs (g C $g^{-1}$ soil), were converted to Total Organic Carbon, TOC (Mg $ha^{-1}$), based on soil layer thickness, L (m), and bulk density, $\rho_b$ (Mg $m^{-3}$):

$$TOC = Cs \times L \times \rho_b \times 10^4 \qquad (1)$$

Forest clearing and cultivation may cause compaction. On the Andic

Humitropept deforestation sequence bulk density gradually increased in time, which suggest that in this case not clearing practices, but cattle trampling was the main reason for compaction (Keller et al., in press). On all pasture sites, bulk density was higher than for the corresponding forest sites (Table 1). Changes in bulk density may have an important effect on C-balance calculations, because samples of the forest at a certain depth are not directly comparable with samples in the pasture at the same depth. For example: if the bulk density of the first layer in the forest increases from 0.6 Mg m$^{-3}$ to 0.8 Mg m$^{-3}$ in the pasture, the 0.1 m top layer in the forest corresponds to a 0.075 m layer in the pasture. Thickness of layers in the pasture soils were adjusted assuming that the relationship between $\rho_b$ and depth in the pasture was originally the same as presently found in the corresponding forest profiles.

The amounts of soil organic carbon derived from forest (SOC$_f$) and from pasture (SOC$_p$) in soils under pasture, were estimated from the equation (Vitorello et al., 1989):

$$TOC \times \delta^{13}C_p = SOC_f \times \delta^{13}C_f + SOC_p \times \delta^{13}C_r \qquad (2)$$

where:

$\delta^{13}C_p$ = $\delta^{13}$C value of sample from pasture soil, corrected for compaction;

$\delta^{13}C_f$ = $\delta^{13}$C value of corresponding sample from forest soil;

$\delta^{13}C_r$ = $\delta^{13}$C value of pasture residues (litter and roots).

In equation 2 variations in $\delta^{13}$C due to mineralization after change in land use are ignored.

## Results and discussion

### *Organic carbon dynamics in different soil types.*

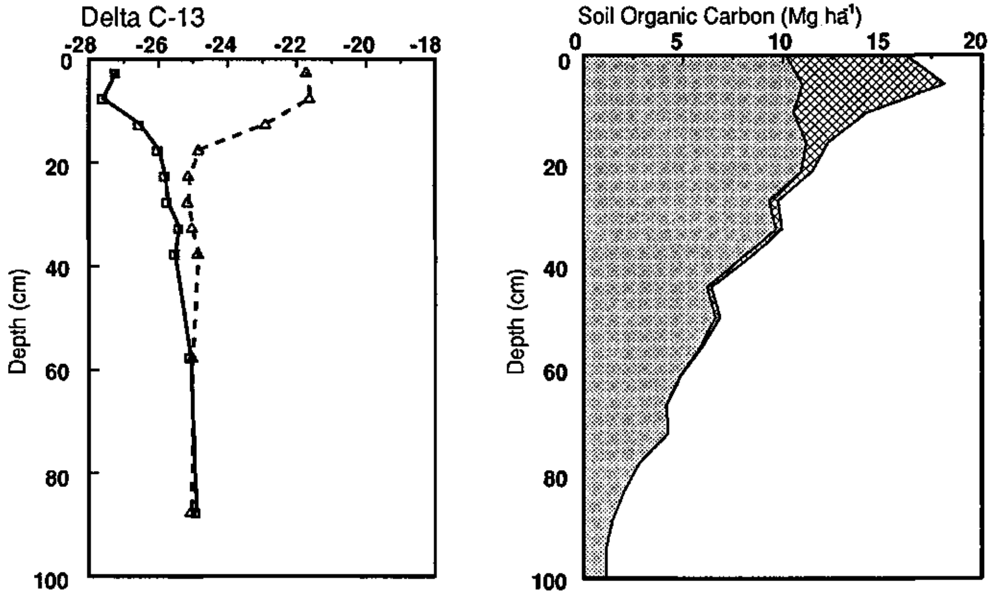
SOC profiles of forest and pasture in the Eutric Hapludand differ considerably (Table 2). TOC has decreased in the top 0.1 m from 48.1 Mg ha$^{-1}$ to 34.3 Mg ha$^{-1}$, if I assume that the soil under forest can be considered as the precursor of the soil under pasture, and if changes in bulk density are ignored. Differences in deeper layers are small. For the Oxic Humitropept under pasture a small

**Table 2.** Soil organic carbon contents for two soil types under forest and under pasture with and without correction for compaction.

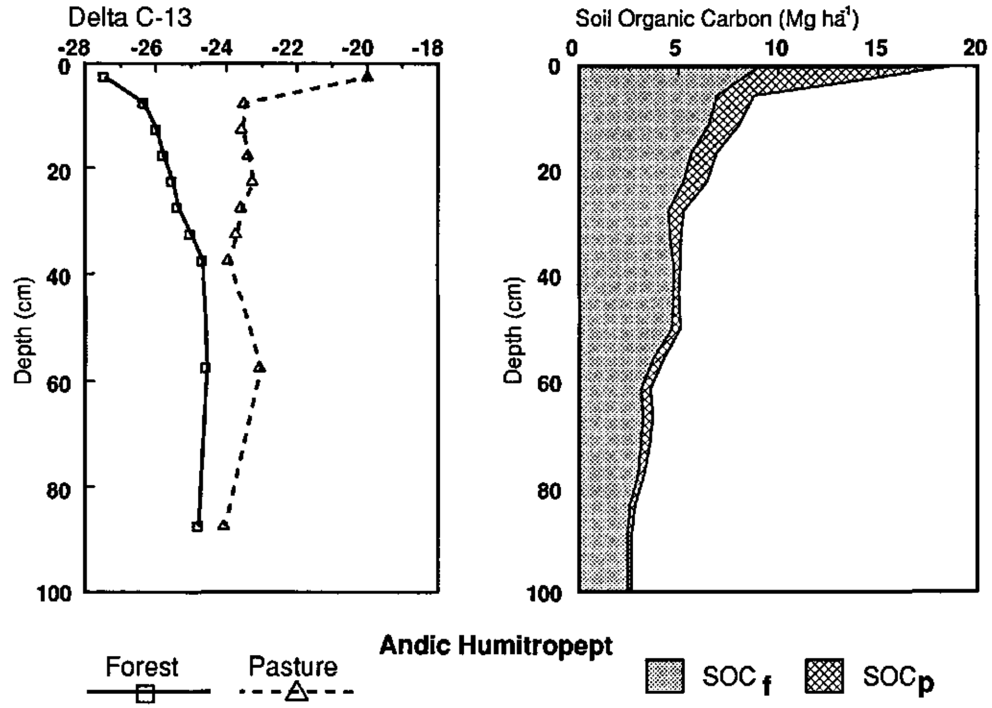
| | Eutric Hapludand | | | Oxic Humitropept | | |
|---|---|---|---|---|---|---|
| | Forest | Pasture correct. | Pasture uncorrect. | Forest | Pasture correct. | Pasture uncorrect. |
| Depth (m) | ----------Mg ha$^{-1}$---------- | | | | | |
| 0.00-0.05 | 21.5 | 11.5 | 16.2 | 16.5 | 15.7 | 18.8 |
| 0.05-0.10 | 26.6 | 13.1 | 18.1 | 9.9 | 9.2 | 8.8 |
| 0.10-0.15 | 15.8 | 13.3 | 14.2 | 7.9 | 6.9 | 8.1 |
| 0.15-0.20 | 11.0 | 11.6 | 12.3 | 7.0 | 7.4 | 6.9 |
| 0.20-0.25 | 13.9 | 10.6 | 11.5 | 6.6 | 5.9 | 6.4 |
| 0.25-0.30 | 8.5 | 10.4 | 9.8 | 6.0 | 5.7 | 5.3 |
| 0.30-0.40 | 12.7 | 17.9 | 18.3 | 9.9 | 10.6 | 10.2 |
| 0.40-0.50 | 14.5 | 14.3 | 13.3 | 10.2 | 11.1 | 10.2 |

increase in TOC can be found in the top 0.1 m (27.6 Mg ha$^{-1}$) as compared to forest (26.4 Mg ha$^{-1}$). When a correction for compaction is made, the TOC under pasture in the top 0.1 m of the Eutric Hapludand appears to have decreased to 24.6 Mg ha$^{-1}$, almost 30% less than without correction. For the Oxic Humitropept the corrected TOC in the topsoil is slightly decreased to 24.9 Mg ha$^{-1}$. Without correction for compaction, the loss of organic C is thus systematically underestimated, especially in top horizons, where compaction is severe. Further calculations in this paper therefore all involved correction for compaction.

In the forest soils $\delta^{13}C$ increased from between -27.01 ‰ and -27.44 ‰ in the top 0 to 0.05 m to about -25.0 ‰ between 0.5 and 0.7 m depth (Figure 1). Similar trends were found by Desjardins et al. (1991) and Volkoff & Cerri (1987) for forest soils in the humid tropics of Brazil. Nadelhoffer & Fry (1988) discussed that extra forest litter inputs to soil lower $\delta^{13}C$ values of top soils. They also linked increasing $\delta^{13}C$ values with depth to soil age and organic matter decomposition. In the forest profiles of the present study, $\delta^{13}C$ values increased with depth to a value of about -25.0 ‰. This suggests that -25.0 ‰ is the $\delta^{13}C$ value typical for the 'passive' soil organic carbon fraction. No differences occurred in $\delta^{13}C$ values in different top soils, suggesting that forest litter at the different sites had similar isotopic composition.



Eutric Hapludand



Andic Humitropept

Figure 1. $\delta^{13}C$ profiles and corresponding SOC$_f$ (Soil Organic Carbon derived from forest) and SOC$_p$ (Soil Organic Carbon derived from pasture) profiles of Eutric Hapludand and Oxic Humitropept under 25 years of pasture.

### *Decomposable and passive soil organic carbon*

Soil organic carbon was divided in two fractions: a 'decomposable' SOC pool (DSOC) which consists of the SOC which decomposes within 25 years (Parton et al., 1987), and a 'passive' SOC pool (PSOC), which consists of SOC which decomposes at slower rates. After 25 years of pasture the decomposable SOC fraction has been mineralized and the remaining soil organic carbon derived from forest ($SOC_f$) consists of passive SOC. This passive soil carbon fraction is much larger for the Hapludand than for the Humitropept (Figure 1). In the Hapludand, the passive carbon fraction remains about constant to a depth of 0.3 m, whereas for the Humitropept it decreases with depth from more than 7 Mg $ha^{-1}$ in the top 0.0 to 0.05 m to less than 5 Mg $ha^{-1}$ in the 0.25 to 0.30 m layer. In the forest profiles, the increase in $\delta^{13}C$ values during decomposition can be used to quantify the decomposable and passive SOC pools. From the following equation both fractions can be calculated under the assumption that the passive soil organic carbon has a typical $\delta^{13}C$ value:

$$TOC_f \times \delta^{13}C_{fs} = DSOC \times \delta^{13}C_D + PSOC \times \delta^{13}C_P \qquad (3)$$

where:

$\delta^{13}C_f$ = $\delta^{13}C$ value of forest soil,
$\delta^{13}C_D$ = $\delta^{13}C$ value of Decomposable Soil Organic C,
$\delta^{13}C_P$ = $\delta^{13}C$ value of Passive Soil Organic C.

The maximum $\delta^{13}C$ value in each profile was used as $\delta^{13}C_P$. The $\delta^{13}C$ value for decomposable organic C must lie between that of fresh plant material ($\delta^{13}C \approx -32.0$ ‰) and SOC in the surface soil ($\delta^{13}C \approx -27.5$ ‰). $\delta^{13}C_D = -28.9$ ‰ was used, which was measured for decomposing litter in the forest. In the two Humitropepts decomposable SOC occurred only in the top 0.5 m (Table 3). In the Hapludand DSOC occurs to a depth of 0.7 m. The total DSOC pool in the top 0.3 m of the Hapludand is about 53 Mg $ha^{-1}$ and in the Andic and Oxic Humitropepts 28 and 23 Mg $ha^{-1}$, respectively. The PSOC pool is also larger in the Hapludand (45 Mg $ha^{-1}$) than in the Humitropepts (23 and 30 Mg $ha^{-1}$, resp.).

**Table 3.** Decomposable Soil Organic Carbon (DSOC) and Passive Soil Organic Carbon (PSOC) pools in three soil types under forest.

| | Eutric Hapludand | | Oxic Humitropept | | Andic Humitropept | |
|---|---|---|---|---|---|---|
| | DSOC | PSOC | DSOC | PSOC | DSOC | PSOC |
| Depth (m) | Mg $ha^{-1}$ | | | | | |
| 0.00 - 0.05 | 13.8 | 7.7 | 11.2 | 5.3 | 10.8 | 5.1 |
| 0.05 - 0.10 | 19.2 | 7.4 | 4.2 | 5.6 | 6.4 | 4.6 |
| 0.10 - 0.15 | 7.9 | 8.0 | 2.8 | 5.1 | 4.7 | 3.6 |
| 0.15 - 0.20 | 4.2 | 6.8 | 2.2 | 4.8 | 3.4 | 2.9 |
| 0.20 - 0.25 | 4.7 | 9.2 | 1.7 | 4.9 | 1.7 | 3.8 |
| 0.25 - 0.30 | 2.8 | 5.8 | 1.3 | 4.7 | 1.1 | 3.5 |
| 0.30 - 0.40 | 3.4 | 9.3 | 1.0 | 8.9 | 1.3 | 5.8 |
| 0.40 - 0.50 | 3.8 | 10.6 | 0.1 | 8.5 | 0.8 | 4.3 |

## *Organic C dynamics after forest clearing*

$\delta^{13}C$ profiles under pastures of different age in an Andic Humitropept were studied to follow the changes in SOC with time after deforestation. In the forest profile $\delta^{13}C$ increases from -27 ‰ at the top to about -24 ‰ at 1.0 m depth. The $\delta^{13}C$ profiles under pasture differ considerably from the forest profile (Figure 2). When a C3 forest vegetation is replaced by C4 grassland vegetation, normally an increase in $\delta^{13}C$ is expected. At 3 and 5 years after deforestation, however, $\delta^{13}C$ at a depth between 0.2 and 0.4 m was distinctly lower than in the forest (Figure 2). An extra input of forest-derived carbon, from decomposing tree roots, is probably responsible for this decrease. When calculations based on the $\delta^{13}C$ profiles of young pastures are performed, this extra input of carbon derived from tree roots has to be taken into account. Pasture-derived organic carbon will be underestimated for the young pastures, if the soil organic carbon profile under forest is taken as reference for calculations, as is usually done (Vitorello et al., 1989).

For root biomass and litter on the forest floor data from Raich (1983) were used who studied the carbon cycle in a forest in the same area on a very similar soil. A $\delta^{13}C$ value of -31.98 ‰ was used for forest roots, which was measured for fresh tree roots. For calculation purposes, litter on the forest floor was added to the top 0.05 m layer. Results of the calculations are summarized in Table 4.

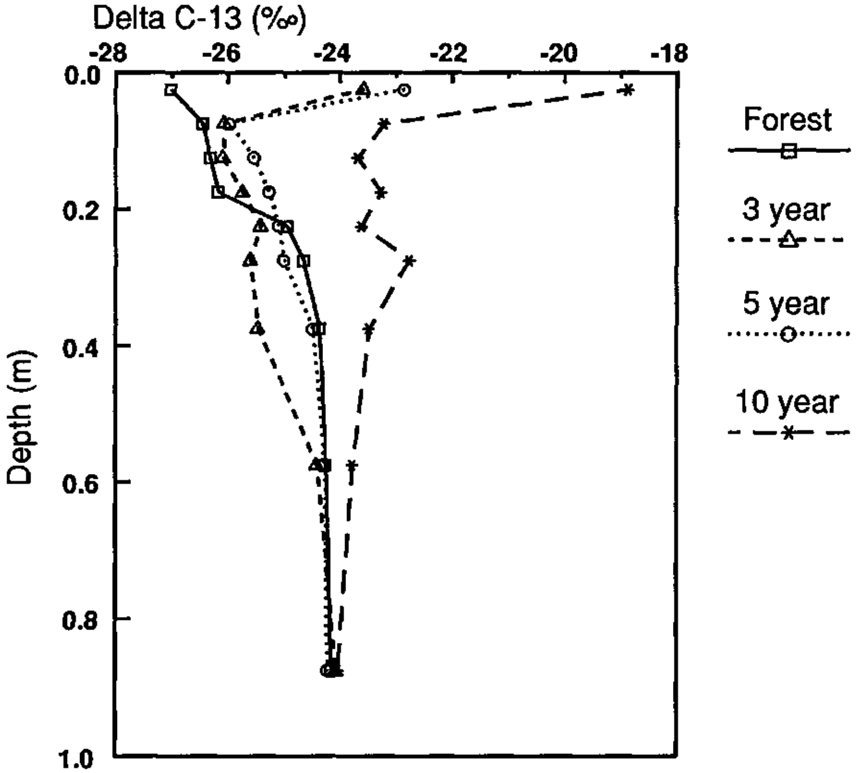


Figure 2. $\delta^{13}C$ profiles in an Andic Humitropept under forest and pastures of 3, 5 and 10 years after deforestation.

For the 0 to 0.1 m layer, changes in time of $SOC_f$ and $SOC_p$ were calculated (Figure 3). An exponential increase of $SOC_p$ was assumed according to:

$$SOC_p = TP \times (1 - \exp(-r_{TP} \times t)) \tag{4}$$

with t = time (in years), TP is the maximum (steady state) level of $SOC_p$ and $r_{TP}$ is the relative growth rate ($y^{-1}$). Best fit ($R^2 = 0.99^{**}$) was obtained with: TP = 12.2 Mg ha$^{-1}$ and $r_{TP} = 0.19$ y$^{-1}$. With these parameters, the maximum value of $SOC_p$ will be reached in about 30 years.

At time = 0 (forest) the sum of organic carbon in the soil (0 to 0.1 m) plus the above-ground litter in the forest floor (24.7 Mg C ha$^{-1}$) and the root biomass (2.3 Mg C ha$^{-1}$) in the top 0.1 m layer is used. A double exponential function was fitted to the $SOC_f$ observations, assuming a decomposable (DF) and a passive (PF) fraction of the soil organic carbon :

$$SOC_f = DF \times \exp(-k_{DF} \times t) + PF \times \exp(-k_{PF} \times t) \qquad (5)$$

with t = time (in years) and $k_{DF}$ and $k_{PF}$ are the relative decomposition rate constants. Parameters with the best fit ($R^2 = 0.99^{**}$) were:
DF = 41.5 Mg ha$^{-1}$; $k_{DF}$ = 0.46 y$^{-1}$; PF = 12.2 Mg ha$^{-1}$ and $k_{PF}$ = 0.003 y$^{-1}$.
Gonzalez & Sauerbeck (1982), who estimated the decomposition rate constants for fresh maize straw on a site in the Atlantic Zone of Costa Rica found values of 3.4 y$^{-1}$ and 0.15 y$^{-1}$. Their overall rate constant corresponds to the rate constant for the decomposable fraction ($k_{DF}$) in the present study.

**Table 4.** Soil organic carbon derived from forest ($SOC_f$) and soil organic carbon derived from pasture ($SOC_p$) in a deforestation sequence on an Andic Humitropept.

| | 0 | | 3 | | 5 | | 10 | | 18 | |
|---|---|---|---|---|---|---|---|---|---|---|
| Depth (m) | $SOC_f$† | $SOC_p$ | $SOC_f$ | $SOC_p$ | $SOC_f$ | $SOC_p$ | $SOC_f$ | $SOC_p$ | $SOC_f$ | $SOC_p$ |
| 0.05 | 41.8 | 0.0 | 12.3 | 4.4 | 8.4 | 5.5 | 6.5 | 7.3 | 3.9 | 6.3 |
| 0.10 | 12.0 | 0.0 | 8.1 | 1.5 | 9.7 | 1.7 | 5.7 | 3.5 | 4.6 | 5.0 |
| 0.15 | 9.1 | 0.0 | 6.0 | 0.4 | 7.5 | 0.6 | 4.6 | 1.6 | 5.2 | 3.7 |
| 0.20 | 7.0 | 0.0 | 5.4 | 0.4 | 5.5 | 0.6 | 3.3 | 1.1 | 5.4 | 2.1 |
| 0.25 | 6.1 | 0.0 | 4.4 | 0.3 | 4.1 | 0.4 | 2.0 | 0.8 | 4.0 | 1.5 |
| 0.30 | 5.1 | 0.0 | 4.1 | 0.1 | 3.5 | 0.2 | 3.0 | 0.6 | 3.8 | 1.0 |

† Numbers include litter on the forest floor and roots in profile.

Extrapolation in time of the equations should be done with care, because only 5 observations in time were used. After the first years the amount of $SOC_f$ seems to stabilize at about 12 Mg ha$^{-1}$ for the 0 to 0.1 m layer. Of the SOC originally present (about 26.9 Mg ha$^{-1}$ for the 0 to 0.1 m layer) about 45 % consists of 'passive' soil organic carbon and the rest of 'decomposable' soil organic carbon. Total soil C ($SOC_p$ + $SOC_f$) rapidly decreases after deforestation and stabilizes after about 5 year (Figure 3). This rapid initial decrease is mainly caused by decomposing litter and roots. If less litter would be left after clearing, the initial TOC value would be lower, but after about 5 year TOC would not differ significantly. After 30 y TOC stabilizes at about 24 Mg ha$^{-1}$ for the 0 to 0.1 m layer. Comparing this number with the TOC pool in the forest (26.8 Mg ha$^{-1}$, without taking litter and roots into account) reveals that in the Andic Humitropept under low productivity pasture the soil organic carbon content in the top layer tends to decrease but only very slowly.

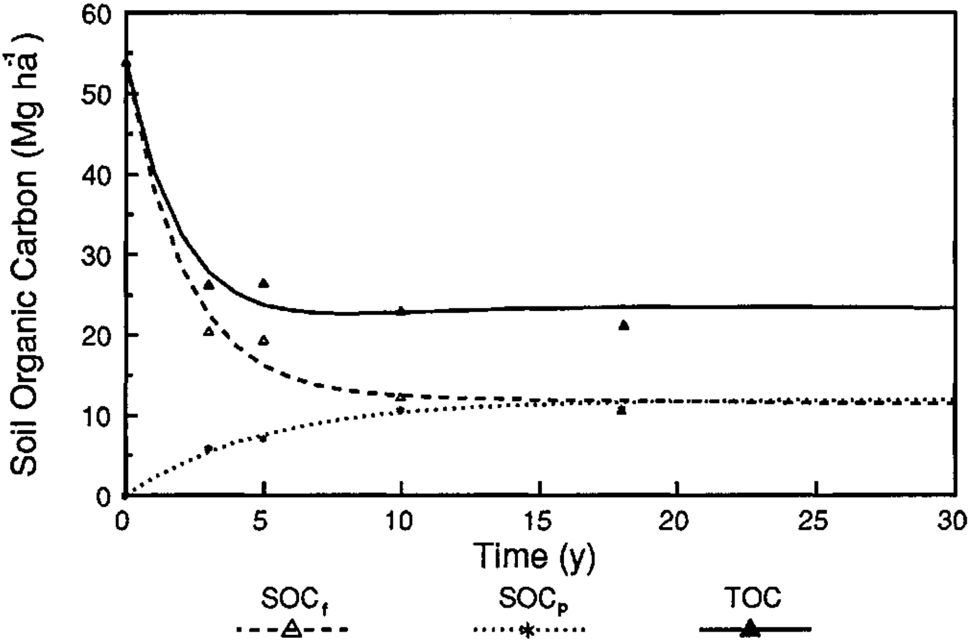


Figure 3. Soil organic carbon changes in time in the top 0 to 0.1 m. Markers are observations, lines are fitted curves.

### *Organic C decomposition with depth*

Equation 5 was fitted to the observations at three depths, to evaluate the relation between changes in SOC and depth in profile. Parameters of the best fits are given in Table 5. The relative decomposition rate of the decomposable $SOC_f$ pool ($k_{DF}$) is higher in the 0 to 0.1 m layer than in deeper layers. The high concentration of relatively easy decomposable litter in the top layer is probably responsible for this effect. In the deeper layers, the decomposable pool (DF) probably consists mainly of tree roots, left in the soil after forest clearing. The rate at which the passive $SOC_f$ pool is decomposed is higher in the top layers than in the deepest layer.

### *Mechanisms stabilizing soil organic C*

Extraction with acid oxalate dissolves Al in allophane and organically bound Al ($Al_o$). Pyrophosphate is supposed to dissolve only humus-Al ($Al_p$). Substraction of $Al_o$ with $Al_p$ results in the Al present in allophane ($Al_{al}$) (Mizota & van Reeuwijk, 1989). Stepwise multiple regression was done to correlate the Al and

**Table 5.** Parameters describing the exponential increase in soil organic carbon derived from pasture ($SOC_p$) and the exponential decrease in soil organic carbon derived from forest ($SOC_f$) at three depth in a deforestation sequence on an Andic Humitropept.

|  | $SOC_p$ |  | $SOC_f$ |  |  |  |
|---|---|---|---|---|---|---|
| Depth (m) | TP† | $r_{TP}$† | DF‡ | $k_{DF}$‡ | PF§ | $k_{PF}$§ |
| 0.0 - 0.1 | 12.2 | 0.19 | 42.5 | 0.44 | 11.1 | 0.003 |
| 0.1 - 0.2 | 8.2 | 0.05 | 7.0 | 0.23 | 9.0 | 0.003 |
| 0.2 - 0.3 | 3.6 | 0.05 | 5.6 | 0.24 | 5.7 | 0.001 |

† TP and $r_{TP}$ steady state level and relative growth rate of soil organic carbon derived from pasture.

‡ DF and $k_{DF}$ decomposable soil organic carbon pool derived from forest and decomposition rate constant.

§ PF and $k_{PF}$ passive soil organic carbon pool derived from forest and decomposition rate constant

Fe extractions to the passive SOC fraction. Changes in passive SOC could be explained by sesquioxide fractions ($R^2 = 0.96^{**}$) in the Eutric Hapludand with the relationship:

$$SOC_f = -2.95 + 14.98 \times Al_p + 3.24 \times (Al_o - Al_p) \qquad (6)$$

Stable Al-organic matter complexes ($Al_p$) thus play a dominant role in the stabilization of SOC. The contribution of ($Al_o$-$Al_p$) indicates that allophane ($Al_o$-$Al_p$) also plays a role in stabilizing organic C.

For the Oxic Humitropept, the explained variance of the regressions using passive SOC as dependent variable was also high ($R^2 = 0.97^{**}$):

$$SOC_f = 0.36 + 6.9 \times Al_p + 3.6 \times Fe_o \qquad (7)$$

So, in the Oxic Humitropept Al-organic matter complexes are closely correlated to the magnitude of the stabile $SOC_f$ pool (Mizota & van Reeuwijk, 1989).

## Conclusions

For a correct application of the $\delta^{13}C$ method, detailed information of changes in bulk densities, accompanying changes in land use, is critical. In the first years after clearing, decomposing tree roots cause a decrease in $\delta^{13}C$ instead of an increase, which would be expected under pastures. Deforestation, followed by 25 years of pasture caused a net loss of 21.8 Mg C ha$^{-1}$ for the Eutric Hapludand and of 1.5 Mg C ha$^{-1}$ for the Oxic Humitropept. Due to the strong stabilization of SOC in Al-organic matter complexes in soils of volcanic origin and the continuous cover of grassland, this decline in soil organic carbon after forest clearing was less than usually reported in other studies.

## Acknowledgements

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## *Chapter 5*

# Uncertainty analysis of the $\delta^{13}C$ method in soil organic matter studies.

*Submitted to Soil Biology & Biochemistry*

E. Veldkamp and A.M. Weitz

# Uncertainty analysis of the $\delta^{13}C$ method in soil organic matter studies.

## Abstract

The $\delta^{13}C$ method is becoming increasingly popular to calculate turnover rates in soil organic matter studies. The method requires a lot of input data, all of which exhibit a natural variability. We performed an uncertainty analysis on the calculations in the $\delta^{13}C$ method to evaluate the influence of uncertainty in the input data on the results of the calculations. Based on field measurements, we estimated the frequency distributions of the input data required for the $\delta^{13}C$ method (soil organic carbon, bulk density and $\delta^{13}C$). From these frequency distributions, two hundred randomly chosen data combinations were sampled, taking into account their mutual correlations. The data combinations were used to calculate the soil organic carbon pool in the forest and pasture and the carbon loss since forest clearing. The uncertainty in output was described using frequency distributions. Uncertainty in the output was high, especially for net carbon loss (ranging between -79.3 to +63.5 Mg $ha^{-1}$). An estimation of the carbon pools within ± 10 % of the estimated mean at the 90 % confidence level requires 5 measurements in the forest and 7 in the pasture. To estimate the carbon loss with the same precision requires 170 combinations of forest and pasture measurements. Heterogeneity in $\delta^{13}C$ values of soil organic matter was the main reason for the higher sample number requirement in the pasture, compared to the forest.

Spatial variability of organic carbon and bulk density in forest and pasture was described by semivariograms. The semivariograms of forest and pasture differed mainly by the distance to which organic carbon and bulk density displayed spatial dependence (the range). The larger range of the pasture semivariograms was explained by the absence of trees. Spatial variability was the main source of the uncertainty in input data. However, variations due to sampling error and short scale variability (the nugget of a semivariogram) are considerable and should not be ignored.

**Keys words:** $\delta^{13}C$, Monte Carlo sampling, sample number requirement, soil organic matter, spatial variability, uncertainty analysis.

# Introduction

Decomposition of soil organic matter plays a critical role in the supply of inorganic nutrients for plant production and can be a major source of atmospheric $CO_2$. Knowledge of soil organic matter dynamics is important if we want to improve management of cropping systems and decrease emissions of soil borne trace gasses.

A relatively new tool used to study organic carbon dynamics in the soil is the stable-isotope analysis of $^{13}C$ (Balesdent *et al.*, 1987). Plant carbon contains distinctly less $^{13}C$ than atmospheric $CO_2$. While atmospheric $CO_2$ has a $\delta^{13}C$ of -7.5 ‰ (Mook, 1986), values of plant C ranges from -22 ‰ to -34 ‰ (average about -26 ‰) in C3 plants, to around -11 ‰ in C4 plants. During decomposition of plant material only a small enrichment in $^{13}C$ takes place. As a result, variations in $^{13}C$ of soil organic carbon (SOC) with soil depth are small (2 to 3 ‰) compared to variations caused by a change in vegetation between C3 and C4 types (about 15 ‰). Growing C4 plants on a soil, which has previously been under C3 vegetation can thus be considered as an *in situ* labelling of soil organic matter (Balesdent *et al.*, 1987). The method requires changes in vegetation from C3 to C4 plants or vice versa, and knowledge about vegetation history and the time of vegetation change.

The availability of the $\delta^{13}C$ method has stimulated quantitative studies of Soil Organic Carbon (SOC) turnover. Balesdent *et al.* (1987) used the method to quantify turnover of total SOC for a forest soil that had been planted continuously to corn (*Zea mays* L.). Vitorello *et al.* (1989) measured SOC turnover in a 12 and 50-yr-old cane (*Saccharum officinarum* L.) field. Martin *et al.* (1990) applied the method in a savanna soil in Ivory coast. Cambardella and Elliott (1992) used the method to characterize organic fractions. Veldkamp (1993) applied the $\delta^{13}C$ method to the bulk SOM of a deforestation - grassland sequence of 0, 3, 5, 10 and 18 yr. These studies are all based on the comparison of two or more sites, characterized by one or a few sample points. Often a 'representative' sample is used, without explaining to what extent the sample is representative and without taking into account the natural variation of the data. Uncertainty analysis is used to get insight into the behaviour of dynamic simulation models and to relate the origins and effects of model uncertainties (Janssen *et al.*, 1993). Uncertainty analysis can also be used to evaluate calculations based on a large number of input data, e.g. the $\delta^{13}C$ method. We used uncertainty analysis to reveal how variability in input data affected the results of the $\delta^{13}C$ method.

The objectives of this study were (i) to determine the uncertainty in the carbon loss from a pasture soil, applying the $\delta^{13}C$ method; (ii) to determine sample number requirements for various degrees of precision using the information of the frequency distributions as well as information on spatial variability; and (iii) to compare the structure and magnitude of the spatial variability of soil organic carbon and bulk density in a humid tropical forest and a 25 yr old pasture.

## Materials and Methods

### *Study sites*

The study was performed in the Atlantic Zone of Costa Rica. The climate is humid tropical; average annual temperature is 26°C and mean annual rainfall is between 3000 and 6000 mm. The natural vegetation is tropical lowland rain forest. One of the main land use types after deforestation is low-input pasture. An important assumption of the $\delta^{13}C$ method is that the situation now found in an adjacent forest site was originally also found on the pasture sites. For that reason, particular attention has been paid to site selection. A deforestation map was made and combined with an available soil map in a GIS (Veldkamp *et al.*, 1992) to select forest and pasture sites on the same soil type.

This study was carried out on a well-drained Eutric Hapludand (Soil Survey Staff, 1990). Two sites were selected. The first sampling site was under natural forest vegetation. The second site was manually cleared and put under pasture (*Axonopus compressus*) about 25 years ago. The sites lie about 1 km apart. A detailed soil description and chemical analysis of the soil types is given in Veldkamp and Weitz (in prep.).

### *Sampling and Sample Processing*

At each site, sampling was done according to an incomplete grid design. The sampling distance ranged from 2 to 130 m in the forest and from 2 to 200 m in the pasture. We choose for an incomplete grid design to cover a wide range of sampling distances with a limited number of sample points. At each sampled grid point samples were taken at three depths: 0-0.1 m, 0.1-0.2 m and 0.2-0.3 m. In the forest, 55 points were sampled for C analysis; at 44 of these points also bulk density was determined and at 10 of these points also $\delta^{13}C$ was analyzed. In the grass plot 64 points were sampled for C analysis, at 39 of these points bulk density was measured and at 10 of these points $\delta^{13}C$ was analyzed. Undisturbed soil samples for determining soil dry bulk density were taken with help of $0.3 * 10^{-3}$ $m^3$ stainless steel rings. The samples were oven dried for 24 h at 105°C. Disturbed soil samples for C and $\delta^{13}C$ analyses were taken with a gauge auger (0.02 m diameter). Soil samples were homogenized, air dried and sieved to less than 2 mm. Litter and root material were removed from the soil manually. Total carbon contents of samples was determined by combustion in a CHN Carlo Erba elemental analyzer. The $\delta^{13}C$ analyses were carried out at the Isotope Physics Laboratory of the University of Groningen, the Netherlands. To measure Carbon-13, soil samples were first treated with 1 N HCl to remove any $CaCO_3$. Next, the samples were then oven dried at 80°C. About 1 g of the soil was burned in the presence of Cu oxide under pure oxygen at 900°C. The resulting $CO_2$ was purified by trapping water vapour on dry ice. The $CO_2$ and $NO_x$ were trapped with help of liquid air (-186°C). Oxygen was evacuated from the purification line. Then $NO_x$ was reduced to $N_2$ in the presence of Cu. The remaining pure $CO_2$ was trapped with help of liquid air and analyzed on a SIRA 9 mass spectrometer. The laboratory reference has been calibrated against V-PDB, using the international standard NBS 19. Analytical precision determined as the standard deviation obtained on different combustions of the same homogenized sample is better than 0.05 ‰ $\delta^{13}C$.

### *Calculation of carbon budgets*

Carbon contents of soil samples were converted from g C $g^{-1}$ soil (Corg) to Mg $ha^{-1}$ (TOC), based on soil layer thickness (L in m) and bulk density ($\rho_b$ in Mg $m^{-3}$):

$$TOC = Corg \times L \times \rho_b \times 10^4 \qquad (1)$$

The grassland soil had higher bulk density than adjacent forest soils as a result of compaction during and after deforestation. To relate the same amount of soil (on mass base) in forest and grass, we corrected our calculations for compaction. The decrease in thickness of the top 0.3 m of pasture soil due to compaction was estimated assuming that the bulk density profiles in forest and grass sites were equal before deforestation (Veldkamp, 1993).

The amounts of soil organic carbon derived from forest ($SOC_f$) and from pasture

($SOC_p$) in soils under pasture, were estimated from the equations (Vitorello *et al.*, 1989):

$$TOC \times \delta^{13}C_{ps} = SOC_f \times \delta^{13}C_{fs} + SOC_p \times \delta^{13}C_p \qquad (2)$$

and

$$TOC = SOC_f + SOC_p \qquad (3)$$

where:

$\delta^{13}C_{ps}$ = $\delta^{13}C$ value of sample from pasture soil;
$\delta^{13}C_{fs}$ = $\delta^{13}C$ value of corresponding sample from forest soil;
$\delta^{13}C_p$ = $\delta^{13}C$ value of pasture residues (litter and roots).

In equations (2) and (3) variations in $\delta^{13}C$ due to mineralization after change in land use are ignored.

Equations 1, 2 and 3 show that calculation of the loss of forest carbon since pasture establishment ($TOC_{fs}$ - $SOC_f$) for one soil horizon, requires the following data: $\delta^{13}C_p$, $\delta^{13}C_{ps}$, $\delta^{13}C_{fs}$, $C_{ps}$, $C_{fs}$, $\rho_{ps}$ and $\rho_{fs}$ where ps and fs refer to pasture and forest soil, respectively. For every horizon more we need information of the same variables with the exception of $\delta^{13}C_p$, which we assume constant with depth. So, the calculation of the forest carbon loss of a soil profile with three horizons requires 19 different input data.

## Statistical Methods

### *Uncertainty analysis.*

Each input variable has its own accuracy and variability. Using these variables in calculations leads to uncertainty in the output. The quality of the calculated estimate will depend not only on random and spatial variations in input data, but also on the assumptions underlying the calculations. In this paper we will concentrate on uncertainty in the input data. We evaluated the influence of these uncertainties on the output of calculations by means of an uncertainty analysis, using the UNCSAM programme (Janssen *et al.*, 1993). Uncertainty in input data was described by specifying the probability distributions (normal and log-normal) and mutual correlations. Random sampling from these distributions ('Monte Carlo' sampling), resulted in 200 sets of values for the various input variables. We choose for the relatively large number of 200, because in this case the calculations are simple and do not take too much time. The 200 sampled sets of input data were used to calculate the carbon loss (equations 1,2 and 3). The distribution of the output was specified to evaluate the total influence of the uncertainties in the various input data on the output.

### *Frequency distributions*

The frequency distributions for the bulk density and carbon measurements were evaluated using the Kolmogorov-Smirnov one-sample test to determine how well a random sample of data or log transformed data fits a normal distribution (Snedecor and Cochran, 1967). The mean of a log-normal distributed variable X is equal to $X = \exp(Y + s_y^2/2)$, where Y and $s_y^2$ are the sample mean and variance of the log-transformed variable (Figure 1). The variance of a log-normal distributed variable X is given by: $s_x^2 = X[\exp(s_y^2) - 1]$.

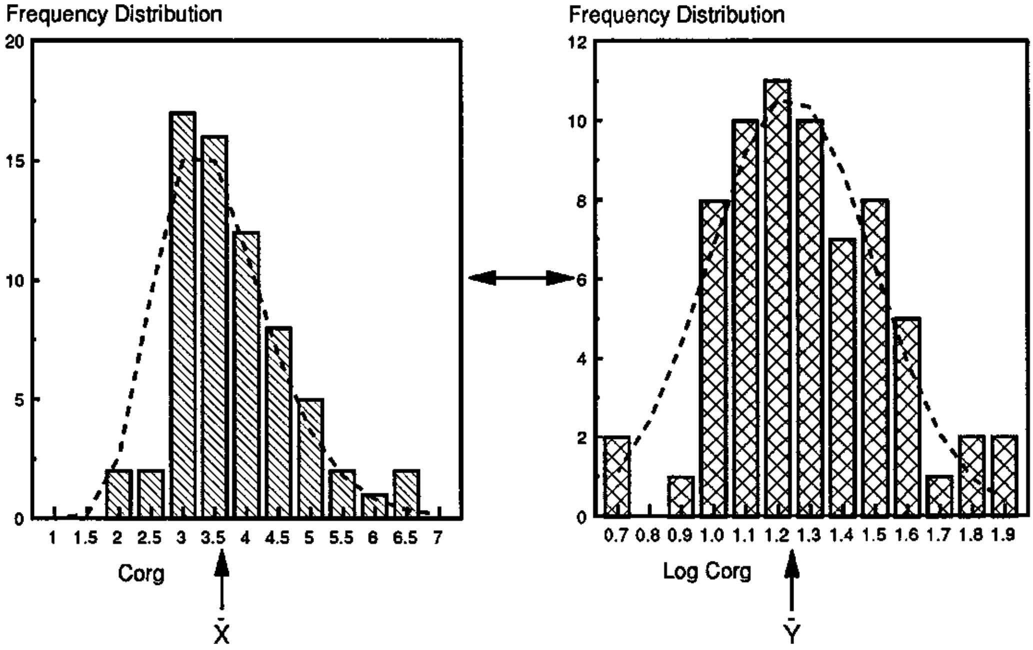

Figure 1. Frequency distributions of a log-distributed variable (Corg: Organic Carbon in pasture) with indicated mean value (X) and the log-transformed distribution of the same variable (Log Corg) with indicated mean value (Y).

### *Sample number requirement*

If we want to calculate the number of samples required to calculate the population mean we have to meet several demands. The population must be normally distributed and the samples must be independent. Furthermore we must specify the relative error that we are willing to tolerate in the estimate and we have to specify the level of significance. If we have met these requirements, the sample number requirement (n) can be approached by (Cochran, 1977):

$$n = ((t_{\alpha/2} \times s_y) / (r \times \bar{Y}))^2 \quad (4)$$

Where r is the relative error, Y is the sample mean, s is the standard deviation, $\alpha$ is the level of significance (e.g. $\alpha = 0.05$ or $\alpha = 0.01$), and t is the t-statistic (number of degrees of freedom = n-1).

### *Spatial variability*

The spatial variability of the input data was described using semivariograms. We will briefly discuss the concept of the semivariogram, for more information we refer to standard geostatistical literature (e.g. Isaaks and Srivastava, 1989). A semivariogram describes the relationship between half the average squared difference between data values paired by their spatial distances. Important features of the semivariogram are:

-Range: the distance at which an increase in the sampling distance no longer causes a corresponding increase in the average squared differences between pairs of values.

-Sill: the plateau that the semivariogram reaches at the range (= variance)

-Nugget: Discontinuity at the origin of the semivariogram caused by sampling error and short scale variability.

A semivariogram is defined as $\tau(h) = \sigma_n^2 + m(h)$, where $\sigma_n^2$ is the Nugget variance, independent of the distance h, and m(h) is the spatially dependent part. The function m(h) is continuous and increasing as a function of h. $\sigma_n^2$ and m(h) can be separated to distinguish the non-spatial variation $\sigma_n^2$ from the spatial variation m(h).

Usually a model equation is fitted to the calculated semivariances. Several model types exist, e.g. linear, spherical and exponential equations. In this study, we calculated the semivariogram for Corg and bulk density on the two sites at three depths. A spherical model equation, one of the most commonly used semivariogram models, was fitted to the calculated semivariogram.

## Results and Discussion

### *Measured Frequency Distributions and Correlations*

The measured Corg and bulk density numbers showed in most cases a normal distribution. Only the Corg in the pasture showed a log- normal distribution. Table 1 lists the Kolmogorov-Smirnov probability, the mean and variance. The probability that the observed frequency distributions for Corg and $\rho_b$ are normal or log-normal is high (P between 0.65 and 0.99).

The correlations between the measured variables are given in Table 2. As we had only 10 observations of $\delta^{13}C$ per site, per depth, their correlations were left out of Table 2. In the forest $\delta^{13}C$ at 0.1-0.2 m only correlated significantly with $\delta^{13}C$ at 0.2-0.3 m ($r = 0.83$). In the pasture $\delta^{13}C$ at 0.1-0.2 m correlated significantly with $\rho_b$ 0.0-0.1 m ($r = -0.72$), $\delta^{13}C$ 0.0-0.1 m ($r = 0.79$) and $\delta^{13}C$ at 0.2-0.3 m ($r = 0.94$). At 0.2-0.3 m $\delta^{13}C$ correlated significantly with $\rho_b$ 0.0-0.1 m ($r = -0.80$).

An evaluation of the mean values for bulk density and Corg of Table 1 show that in the forest bulk density is lower than in the pasture. Furthermore, bulk density in the pasture reaches the highest values in the 0.1 to 0.2 m layer. Both features demonstrate compaction in the pasture. Corg values are generally higher in the forest than in the pasture. The variance of $\delta^{13}C$ is an order of magnitude higher in the pasture than in the forest. This demonstrates that the $\delta^{13}C$ is much more heterogeneous in the pasture. The variance of bulk density is higher in the pasture and the variance of Corg is higher in the forest. In both cases the difference in variance must be explained by differences in heterogenity because the same method (and sampling error) was used for Corg and bulk density in pasture and forest.

### *Carbon budgets*

The two hundred randomly sampled data combinations were used to calculate several dependent variables: the total amount of carbon in the forest soil ($TOC_{fs}$), the total amount of carbon in the pasture soil ($TOC_{ps}$), the loss of forest

**Table 1.** Information on measured frequency distributions of bulk density ($\rho_b$), organic carbon (Corg) and $\delta^{13}$C value of forest and pasture soil in the Atlantic Zone of Costa Rica.

| | Depth (m) | Distrib[a]. | KS[b] Prob. | Mean | Variance |
|---|---|---|---|---|---|
| | | Forest | | | |
| $\rho_b$ (Mg m$^{-3}$) | 0.0-0.1 | normal | 0.65 | 0.652 | 0.005 |
| | 0.1-0.2 | normal | 0.90 | 0.705 | 0.002 |
| | 0.2-0.3 | normal | 0.97 | 0.719 | 0.007 |
| Corg (g C g$^{-1}$) | 0.0-0.1 | normal | 0.98 | 7.155 | 1.563 |
| | 0.1-0.2 | normal | 0.84 | 5.782 | 1.543 |
| | 0.2-0.3 | normal | 0.86 | 4.286 | 0.977 |
| $\delta^{13}$C[c] | 0.0-0.1 | normal | 0.99 | -27.10 | 0.102 |
| | 0.1-0.2 | normal | 0.96 | -26.17 | 0.221 |
| | 0.2-0.3 | normal | 0.59 | -25.58 | 0.067 |
| | | Pasture | | | |
| $\rho_b$ (Mg m$^{-3}$) | 0.0-0.1 | normal | 0.88 | 0.745 | 0.010 |
| | 0.1-0.2 | normal | 0.78 | 0.847 | 0.005 |
| | 0.2-0.3 | normal | 0.76 | 0.828 | 0.010 |
| Corg (g C g$^{-1}$) | 0.0-0.1 | log-normal | 0.81 | 5.714 | 1.573 |
| | 0.1-0.2 | log-normal | 0.82 | 4.367 | 1.301 |
| | 0.2-0.3 | log-normal | 0.99 | 3.508 | 0.997 |
| $\delta^{13}$C[c] | 0.0-0.1 | normal | 0.99 | -20.51 | 1.543 |
| | 0.1-0.2 | normal | 0.99 | -22.17 | 0.726 |
| | 0.2-0.3 | normal | 0.69 | -23.20 | 0.683 |

[a] Distribution
[b] Kolmogorov Smirnov Probability
[c] Based on 10 samples only

carbon since clearing equal to $TOC_{fs}$ - $SOC_f$ (C-loss$_f$), the increase of pasture carbon since clearing ($SOC_p$) and the Net Difference between $TOC_{fs}$ and $TOC_{ps}$ (Net Dif.)(equations 1, 2 and 3). Based on the ten observations of $\delta^{13}$C per soil horizon per site we used normal distributions for $\delta^{13}$C. The calculated distributions of dependent variables are summarized in Table 3.

The calculated $TOC_{fs}$ and C-loss$_f$ both have normal distribution. The $TOC_{ps}$, $SOC_p$ and Net Dif. have a log-normal distribution. The large spread in results is evident, as is illustrated by a plot of $TOC_{fs}$ and Net Difference between $TOC_{fs}$

**Table 2.** Correlations between bulk density ($\rho_b$) and organic carbon (Corg) at three depths for forest and pasture soils in the Atlantic Zone of Costa Rica.

| | Forest | | | | | |
|---|---|---|---|---|---|---|
| | $\rho_b$ 00 | $\rho_b$ 10 | $\rho_b$ 20 | Corg 00 | Corg 10 | Corg 20 |
| $\rho_b$ 00 | 1.00 | 0.35 | 0.14 | -0.32 | $-0.58^{**}$ | $-0.37^{**}$ |
| $\rho_b$ 10 | | 1.00 | $0.74^{**}$ | 0.03 | -0.34 | $-0.51^{**}$ |
| $\rho_b$ 20 | | | 1.00 | -0.02 | -0.39 | $-0.59^{**}$ |
| Corg 00 | | | | 1.00 | $0.63^{**}$ | 0.19 |
| Corg 10 | | | | | 1.00 | $0.69^{**}$ |
| Corg 20 | | | | | | 1.00 |

| | Pasture | | | | | |
|---|---|---|---|---|---|---|
| | $\rho_b$ 00 | $\rho_b$ 10 | $\rho_b$ 20 | Corg 00 | Corg 10 | Corg 20 |
| $\rho_b$ 00 | 1.00 | $0.57^{**}$ | 0.32 | $-0.73^{**}$ | -0.30 | -0.12 |
| $\rho_b$ 10 | | 1.00 | $0.82^{**}$ | $-0.80^{**}$ | $-0.67^{**}$ | -0.34 |
| $\rho_b$ 20 | | | 1.00 | $-0.76^{**}$ | $-0.69^{**}$ | $-0.46^{*}$ |
| Corg 00 | | | | 1.00 | $0.70^{**}$ | $0.51^{**}$ |
| Corg 10 | | | | | 1.00 | $0.61^{**}$ |
| Corg 20 | | | | | | 1.00 |

$^{*}$ and $^{**}$ : significant at P = 0.01 and 0.001 level.

00, 10 and 20: Depth = 0.0-0.1 m; 0.1-0.2 m and 0.2-0.3 m.

and TOC$_{ps}$ (Figure 2). The best estimator for the Net Difference is -23.0 Mg ha$^{-1}$, but based on one random observation in the forest and one in the pasture, the result can vary between a carbon loss of 79.3 Mg ha$^{-1}$, to a carbon increase of 63.5 Mg ha$^{-1}$. It is obvious that a conclusion based on only one observation in forest and pasture is not very reliable and should be interpreted with great care.

### *Sample number requirement*

To calculate the sample number requirement, we have to meet the demands of a normally distributed population and independent samples. We already showed that the sampled populations were normally or log-normally distributed. The log-normal distributions were log-transformed to normal distributions (Figure 1). De Gruijter and Ter Braak (1990) showed that whether n samples are stochastically

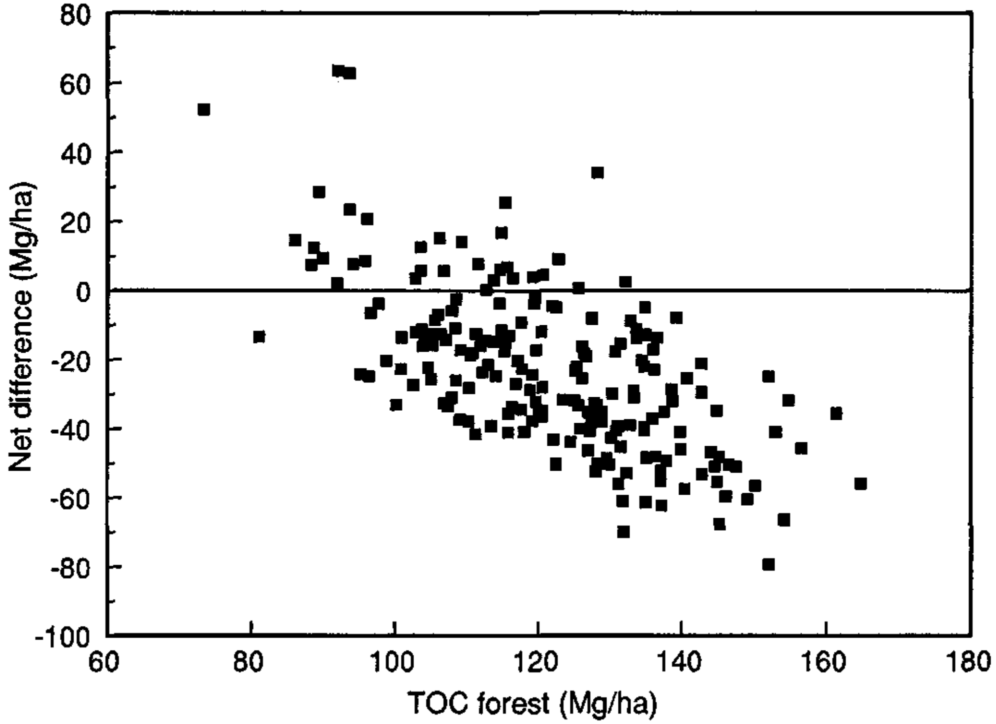

Figure 2. Plot of calculated Total Organic Carbon in Forest Soil ($TOC_{fs}$) and the net difference between calculated $TOC_{fs}$ and the Total Organic Carbon in the Pasture Soil ($TOC_{ps}$) for 200 sets of input data, randomly sampled from their distributions and mutual correlations.

independent or not is completely determined by the sampling design and not by the spatial variation in the sampling area. This means that the following calculations of sample number requirement are only valid if random sampling is applied.

Using equation 4, we calculated the number of observations needed to estimate the mean of the normal or log-normal distributions at the 90 and 95 % confidence level for 5 and 10 percentages of the true mean. Table 4 gives the sample number requirement for the calculated distributions. The number of observations required strongly depends on the precision and confidence level desired. To increase the confidence level from 90 % to 95 %, with the same precision about 50 % more samples are required. To enlarge precision from 10 % to 5 % ± the true mean requires three to four times more observations.

If we accept a precision of 10 % ± the 'true' mean at the 90 % confidence level, the number of observations required for $TOC_{fs}$ and $TOC_{ps}$ are 5 and 7

**Table 3.** Information on calculated distributions of carbon pools and changes after forest clearing.

| | Distrib.[a] | KS[b] prob. | Mean | Variance | Minimum (Mg ha$^{-1}$) | Maximum (Mg ha$^{-1}$) |
|---|---|---|---|---|---|---|
| $TOC_{fs}$ | normal | 0.76 | 121.5 | 276.3 | 73.2 | 164.9 |
| $TOC_{ps}$ | log-normal | 0.77 | 98.5 | 311.6 | 62.0 | 162.5 |
| C-loss$_f$ | normal | 0.93 | 59.0 | 388.1 | -3.3 | 102.5 |
| $SOC_p$ | log-normal | 0.99 | 36.0 | 66.3 | 18.2 | 64.1 |
| Net Dif. | log-normal | 0.75 | -23.0 | 559.1 | -79.3 | 63.5 |

[a] Distribution

[b] Kolmogorov Smirnov Probability

respectively, for C-loss$_f$ we need 19 observations, for $SOC_p$ we need 10 observations and for Net Difference we need 170 observations. This does not necessarily mean that 170 measurement have to be performed in forest and pasture, because 5 observations in the forest and 7 in the pasture can be combined to 5 × 7 = 35 calculations of C-loss$_f$ and C-increase$_p$. These 35 loss and increase numbers can be combined to 35 × 35 = 1225 calculations of Net Difference. This combining should however be done with care because the resulting 1225 calculations are not randomly selected. The higher number of observations required in the pasture was mainly caused by the heterogenity of the $\delta^{13}C$ values in the pastures (Table 1).

These results show that calculation of C-loss based on the $\delta^{13}C$ method exhibits large variances. A large number of calculations will be required to make a precise estimate. The number of observations of TOC in forest and pasture which are necessary, turns out to be considerably less than the number of calculations required, because the observations can be combined in numerous ways to calculate the C-loss. The calculated number of observations may depend upon the size of the sample taken. Use of larger samples might result in fewer observations due to the possible integration of the short scale variability by the larger sample. However only the short scale variability will decrease by larger samples. The spatial variability will not decrease when taking larger samples.

**Table 4.** Number of observations required to calculated Total Organic Carbon in the forest ($TOC_{fs}$), Total Organic Carbon in the pasture ($TOC_{ps}$), Loss of forest carbon since clearing (C-loss$_f$), Soil Carbon derived from pasture ($SOC_p$) and Net Difference between $TOC_{fs}$ and $TOC_{ps}$ for two degrees of precision at the 90 and 95% confidence levels.

|  | ± % of estim. mean | ---Number of observations--- |  |
|---|---|---|---|
|  |  | 90% | 95% |
| $TOC_{fs}$ | 5 | 14 | 22 |
|  | 10 | 5 | 8 |
| $TOC_{ps}$ | 5 | 23 | 36 |
|  | 10 | 7 | 11 |
| C-loss$_f$ | 5 | 75 | 118 |
|  | 10 | 19 | 32 |
| $SOC_p$ | 5 | 35 | 58 |
|  | 10 | 10 | 16 |
| Net Dif. | 5 | 693 | 1108 |
|  | 10 | 170 | 287 |

### *Spatial variability*

Semivariograms were made for Corg and bulk density for forest and pasture at the three depths. An example of a semivariogram with the nugget, sill and range is given in Figure 3. Table 5 gives the nugget, sill and range of all semivariograms. In all cases a sill is reached indicating that there was no trend in the studied area. Both Corg and bulk density show a clear spatial structure. There are, however, clear differences between forest and pasture sites. In the pasture, the range of the semivariograms, is considerably larger. It is likely that the spatial structure in the forest is strongly influenced by the presence of trees, causing a range of about 10 m for both bulk density and Corg. In the pasture trees are absent and the spatial structure probably shows more relation with the parent material, resulting in a larger range.

### *Sample number requirement without spatial variability*

We have calculated the sample number requirement for a situation including spatial variability (which is normally the case if we compare two different sites). When we sample in the forest, cut the forest down and come back after some

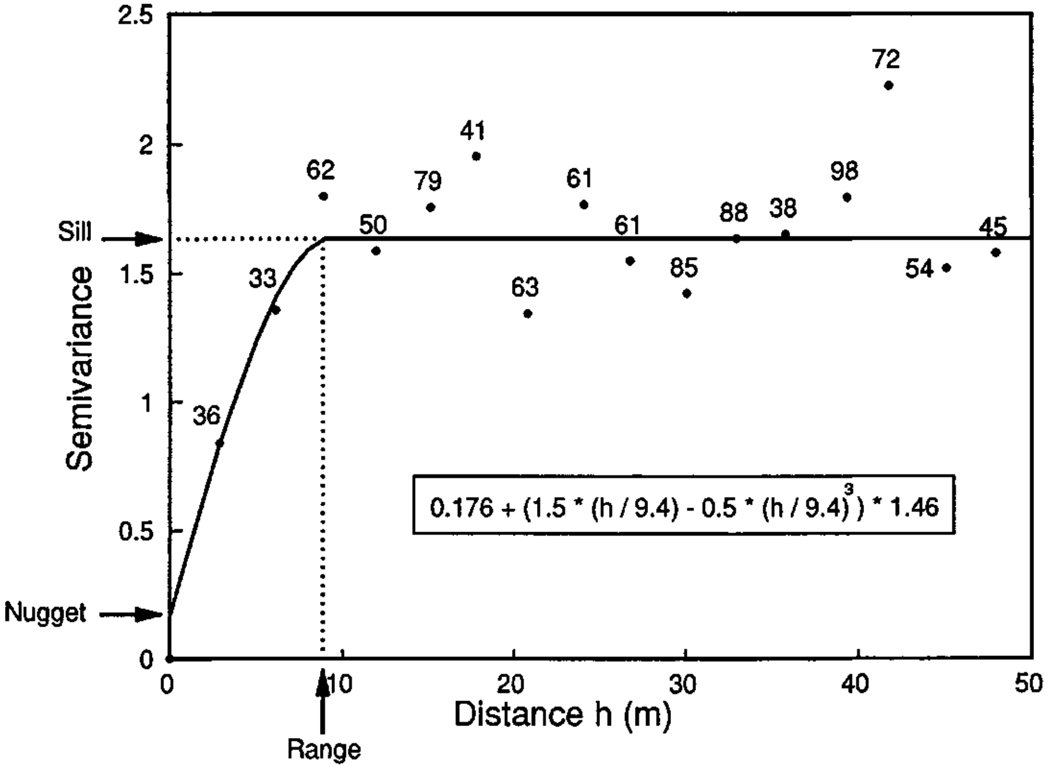

Figure 3. Example of calculated and fitted semivariogram of the Corg in the forest between 0.1 and 0.2 m depth. The range, sill and nugget are indicated. Data labels indicate number of data pairs used to calculated semivariance at particular distance interval.

years to sample in the pasture on exactly the same place, we minimize spatial variability. In this case the only causes of variability are sampling error and short scale variability, which are quantified by the nugget variance of the semivariograms. To calculate the sample number requirement when spatial variability is minimized, we repeated the calculations on the carbon budgets with the assumption that the variance is equal to the nugget effect.

Compared to the original results (Table 3) the standard deviation of $TOC_{fs}$ and $TOC_{ps}$ decrease with 51 and 60 %, the standard deviations of C-$loss_f$ and $SOC_p$ decrease with 61 and 66 % and the standard deviation of the Net Difference decreases with 63 %. The results for the sample number requirement are displayed in Table 6. A precision of 10 % ± the 'true' mean at the 90 % confidence level requires three samples in the forest and 3 in the pasture. These three samples should be taken close to each other.

**Table 5.** Some characteristics of semivariograms of bulk density ($\rho_b$) and organic carbon (Corg) in forest and pasture soils in the Atlantic Zone of Costa Rica.

| | Depth (m) | Nugget | Sill | Range(m) |
|---|---|---|---|---|
| | | Forest | | |
| $\rho_b$ | 0.0-0.1 | $1.33\ 10^{-4}$ | $4.69\ 10^{-3}$ | 11.3 |
| (Mg m$^{-3}$) | 0.1-0.2 | $8.73\ 10^{-4}$ | $2.47\ 10^{-3}$ | 13.7 |
| | 0.2-0.3 | $2.09\ 10^{-3}$ | $7.34\ 10^{-3}$ | 11.8 |
| Corg | 0.0-0.1 | 0.473 | 1.586 | 9.2 |
| (g C g$^{-1}$) | 0.1-0.2 | 0.176 | 1.636 | 9.4 |
| | 0.2-0.3 | 0.567 | 0.953 | 10.7 |
| | | Pasture | | |
| $\rho_b$ | 0.0-0.1 | $1.28\ 10^{-3}$ | $9.89\ 10^{-3}$ | 25.2 |
| (Mg m$^{-3}$) | 0.1-0.2 | $6.91\ 10^{-4}$ | $5.42\ 10^{-3}$ | 35.1 |
| | 0.2-0.3 | $2.10\ 10^{-3}$ | $9.91\ 10^{-3}$ | 49.7 |
| Corg | 0.0-0.1 | 0.222 | 1.545 | 64.8 |
| (g C g$^{-1}$) | 0.1-0.2 | 0.251 | 1.285 | 66.3 |
| | 0.2-0.3 | 0.309 | 0.988 | 89.3 |

### *Sampling Strategy*

We showed that the uncertainty in the results of the calculations can be due to measurement errors, short scale variability and spatial variability. Measurement errors can be minimized e.g. by taking larger samples. Spatial variability can only be controlled if the $\delta^{13}$C method is applied in an experiment where we can sample on exactly the same location at different times. In most cases, however, this is not a realistic option. Short scale variability (and in most cases spatial variability) cannot be controlled.

Because 'representative' samples conceal the inherent natural variability, we are no supporter of this kind of method. We found that spatial variability is the principal reason for the large spread in results, but it is not the only one as the remaining standard deviations (caused by sampling error and short scale variability) are still considerable. The only way to cope with natural variability is to adapt the sampling strategy to the observed variability. The sampling strategy will be the result of balancing the required precision, the required confidence level and the available time and money.

If we compare our results with previous studies, we do not think that these studies must be discarded because they took only one or a representative sample

**Table 6.** Number of observations required to calculated Total Organic Carbon in the forest ($TOC_{fs}$), Total Organic Carbon in the pasture ($TOC_{ps}$), Loss of forest carbon since clearing (C-$loss_f$), Soil Carbon derived from pasture ($SOC_p$) and Net Difference between $TOC_{fs}$ and $TOC_{ps}$ for two degrees of precision at the 90 and 95% confidence levels, when minimizing spatial variability.

| | | Nr. of observations | |
|---|---|---|---|
| | ± % of est. mean | 90% | 95% |
| $TOC_{fs}$ | 5 | 5 | 7 |
| | 10 | 3 | 3 |
| $TOC_{ps}$ | 5 | 5 | 8 |
| | 10 | 3 | 4 |
| C-$loss_f$ | 5 | 13 | 21 |
| | 10 | 5 | 7 |
| $SOC_p$ | 5 | 6 | 9 |
| | 10 | 3 | 4 |
| Net Dif. | 5 | 97 | 162 |
| | 10 | 26 | 41 |

for each site. Especially when more than two sites were compared in a time sequence as was e.g. done by Balesdent *et al.* (1988), Vitorello *et al.* (1989) and Veldkamp (1993) it is likely that the observed trends were interpreted in a correct way. If, however, only two sites were compared the results should be interpreted only in a qualitative way.

## Conclusions

Application of uncertainty analysis on the $\delta^{13}C$ method has demonstrated that the output of the $\delta^{13}C$ method in soil organic matter studies is highly variable due to variations in the input data. Uncertainty can be caused by measurement errors and natural variability. We have demonstrated that in this study spatial variability is the most important source of uncertainty. However variations due to sampling error and short scale variability are considerable and should not be neglected. If the $\delta^{13}C$ method is applied in soil organic matter studies, the variability of the input data should be quantified. The results of the calculations should be accompanied by the degree of precision and confidence level of the output. The number of observations required will be the result of balancing the required precision, the required confidence level and the available time and money.

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*Chapter 6*

# Improved pastures reduce net soil $CO_2$ emissions after deforestation in the Atlantic Zone of Costa Rica.

*Submitted to Climatic Change*

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# Improved pastures reduce net soil $CO_2$ emissions after deforestation in the Atlantic Zone of Costa Rica

## Abstract

The clearing of forest for cattle pastures is one of the main reasons for the present large contribution of Latin America to the global net release of carbon dioxide from soil to the atmosphere. At our study sites in the Atlantic Zone of Costa Rica, we calculated the net $CO_2$ release from soil to atmosphere for some cattle pastures and explored possibilities to decrease these emissions by introducing other pasture types. We used a simple structured soil organic carbon (SOC) model which includes carbon isotope fractionation during decomposition and depth dependent decomposition and humification rates. With this model, the observed changes in soil organic carbon and corresponding $\delta^{13}C$ levels during the conversion from a humid tropical forest (C3 vegetation) to a cattle pasture (C4 vegetation) were simulated successfully for two contrasting soil types. The cumulative net release of $CO_2$ for pastures with low productive grass species (*Axonopus compressus*), varied from 31.5 (Humitropept) to 60.5 Mg C $ha^{-1}$ (Hapludand) in the first 20 years after forest clearing. These cumulative emissions can be reduced to 12.0 and 24.7 Mg C $ha^{-1}$ respectively, if higher productive grass species (e.g. *Brachiaria dictyoneura*) will be introduced into the area. Modelling of isotope fractionation during decomposition will improve the calibration of 'passive' or 'biologically inert' pools, and therefore the performance of SOC models in long-term studies.

**Key words:** C3, C4 vegetation, $CO_2$ emission, $\delta^{13}C$, improved pasture, isotope fractionation, model, soil organic matter.

## Introduction

Currently, rates of deforestation in the tropics are higher than ever before in the past. Forest clearing not only has important impacts on the local physical environment such as soil degradation and nutrient loss, but increasing atmospheric concentrations of $CO_2$, $N_2O$ and other biotic trace gases reveal that changes in land use, caused by deforestation are of global importance. Net release of carbon from the world's tropics due to deforestation was recently estimated to be 0.42 to 1.60 Pg C $yr^{-1}$, with 0.1 to 0.3 Pg C $yr^{-1}$ attributed to decreases in soil organic matter content (Detwiler and Hall, 1988). This carbon release from tropical areas is second only to the global release from the burning of fossil fuels, which is about 5.3 Pg C $yr^{-1}$.

In Latin America, the total net release of carbon between 1850 and 1985 due to changes in land use was estimated to be 30 Pg C (Houghton *et al.*, 1991). This $CO_2$ was mainly emitted from an increased area of pastures. In 1985, Latin America had the largest net flux of biotic carbon (0.67 Pg C $yr^{-1}$), approximately one-half of the total biotic $CO_2$ flux for the world's tropics. The largest contribution to this flux resulted from increases in the area of degraded lands, which mainly consists of degraded pastures (Houghton *et al.*, 1991). Although loss of soil organic carbon after deforestation is less in pastures than in croplands (Detwiler, 1986), pastures play a key role in the release of biotic carbon to the atmosphere, due to the large area they cover.

In the Atlantic Zone of Costa Rica large areas of tropical lowland forest have been cleared in the last 40 years (Veldkamp *et al.*, 1992). In 1986 more than 50 % of this cleared area was covered with pastures (Huising, 1993). In these pastures, a study on the dynamics of soil organic carbon (SOC) showed that deforestation, followed by 25 years of low productive pasture, caused a net loss of soil organic carbon of between 1.5 Mg C $ha^{-1}$ and 21.8 Mg C $ha^{-1}$ depending on the soil type (Veldkamp, 1993a). In an additional study, the below-ground dry matter production of a low productive (*Axonopus compressus*) and high productive 'improved' pasture (*Brachiaria dictyoneura*) were compared, using pulse labelling with $^{14}C$ (Veldkamp, 1993b). The below ground dry matter production of Brachiaria (12 Mg $ha^{-1}$ $yr^{-1}$) was about twice the below ground production of Axonopus (6 Mg $ha^{-1}$ $yr^{-1}$).

Soil organic carbon (SOC) levels are the result of production and decomposition processes. The best way to integrate these dynamic processes is by dynamic simulation modelling. We used a simple soil organic carbon model to describe the changes that take place in soil organic carbon levels after forest clearing and during pasture establishment. This modelling approach had several objectives. The first was to simulate the observed organic matter levels in two soil types under low productive pastures, following forest clearing in the Atlantic Zone of Costa Rica and to calculate the resulting net emission of $CO_2$. The second objective was to evaluate the effect of replacing the low productive pasture by an 'improved' pasture on the soil organic matter levels and the corresponding net $CO_2$ efflux between soil and atmosphere. The third objective was to demonstrate the use of stable C isotopes for calibration of SOC models and to demonstrate the concept of depth dependent decomposition.

## Description of important model characteristics

The model we developed simulates the dynamics of soil organic carbon before and after the conversion from C3-forest to C4-pasture. As the model is developed according to common SOC model theories, no detailed description of the model will be given, but we will concentrate on the main differences between our model and the CENTURY model (Parton *et al.*, 1987). Our model uses the concept of three discrete SOC pools with increasing turnover time, which is frequently used in models of soil organic matter dynamics (Van Veen *et al.*, 1985; Parton *et al.*, 1987; Jenkinson, 1990). The structure of the SOC model was kept as simple as possible (Figure 1), to reduce the number of parameters which must be calibrated. Litter was not divided into 'structural' and 'metabolic' pools, but enters directly into the 'rapid' soil organic carbon pool. Soil carbon fluxes are supposed to be uni-directional (Figure 1). The biomass was separated into different pools which mainly differ in residence time. In the forest, the above ground biomass pool is divided into leafs, branches and stems; below ground biomass is subdivided into coarse, medium and fine roots. In the pasture no subdivision was made between fine, medium and coarse roots. Instead, we used a residence time of carbon in roots which varies with depth (Veldkamp, 1993b). The model contains three soil organic carbon pools ('Rapid', 'Active' and 'Passive'). Decay of the pools is by first order kinetics. The first-order decay rates are designated as $k_{Rt}$, $k_{At}$ and $k_{Pt}$. For the 'rapid' and 'active' pools the overall decay rate is envisaged as the sum of respiration and humification rate ($k_{Rt} = k_{Rr} + k_{Rh}$, and $k_{At} = k_{Ar} + k_{Ah}$). The 'passive' SOC pool is only decomposed by respiration ($k_{Pt} = k_{Pr}$). Temperature and moisture were not included into the model, because these variables are almost always optimal

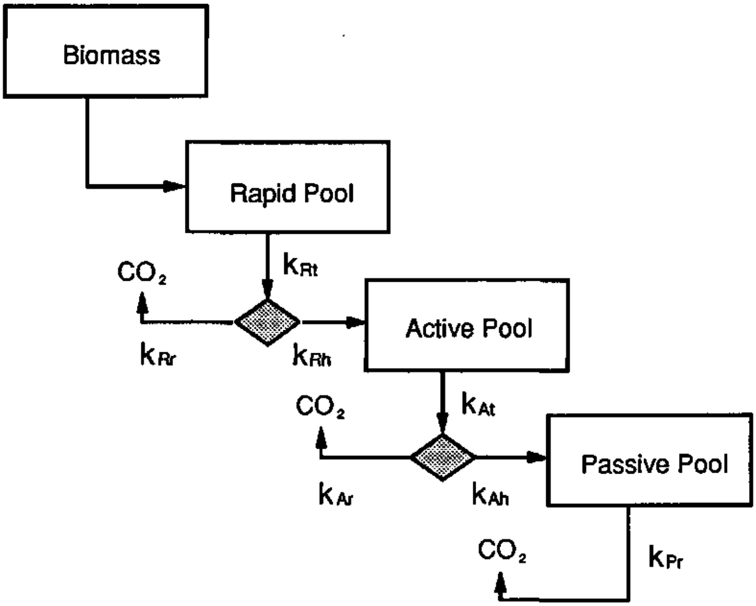

Figure 1. Structure of soil organic carbon model.

for decomposition in the study area. Compared to CENTURY or the ROTHAMSTED model (Jenkinson, 1990), we made some important additions: 1) we included $^{13}C$ in the model, with discrimination against the heavier isotopes during organic matter decomposition in the model, 2) we considered multiple soil layers with depth-dependent rates of decomposition and humification, and 3) we included conversion of C3-forest vegetation to C4-pasture vegetation into the model.

### *Isotope fractionation during decomposition*

$^{13}C$ was included into the model together with discrimination against $^{13}C$ during organic matter decomposition. This was done to get more insight into the processes taking place after a change of vegetation with different $^{13}C$ signatures, and to consider isotope fractionation during decomposition. Measured $\delta^{13}C$ profiles under forest and pastures of different age were used for the calibration of the model.

Plant carbon contains distinctly less $^{13}C$ than atmospheric $CO_2$. Further fractionation takes place during decomposition of plant material in soil (Nadelhoffer & Fry, 1988). To quantify isotope effects, carbon isotope ratios defined as:

$$^{13}R = {}^{13}C\,/\,{}^{12}C$$

can be used. In practice, instead of isotopic ratios, δ values are used, which is the relative difference between isotope ratios of sample and a standard:

$$\delta^{13}C\ ^{o}/_{oo} = ({}^{13}R_{sample} - {}^{13}R_{standard})\ /\ ({}^{13}R_{standard}) \times 1000$$

The international standard is V-PDB (Vienna-Pee Dee Belemnite) which has a $^{13}$R of 0.0112372 (O'Leary, 1981). While atmospheric $CO_2$ has a $\delta^{13}$C of -7.5 *‰* (Mook, 1986), values of plant C range from -22 *‰* to -34 *‰* (average about -26 *‰*) in C3 plants, to around -15 *‰* in C4 plants. In well drained soils under forest, normally a small enrichment in $^{13}$C can be observed in deeper layers (Balesdent *et al.*, 1987; Veldkamp, 1993a). This enrichment is a reflection of isotope discrimination taking place during respiration of SOC and stabilization of the $^{13}$C enriched, non-respired substrate in carbon pools with a longer residence time. There are several other mechanisms which could lead to $^{13}$C enrichment with depth: differential preservation of lignin, changes in litter source, and illuviation of $^{13}$C enriched dissolved organic C (Nadelhoffer & Fry, 1988). We did not include these processes in the model for several reasons: 1) Residual lignin is the C fraction which is most likely to be preserved during organic carbon decomposition because lignin decomposes relatively slowly. Lignin preservation would however lead to $^{13}$C depletion (Benner *et al.*, 1987) rather than enrichment, and therefore cannot account for the $^{13}$C enrichment in depth. 2) We have no reason to suspect that in the forest sites the isotopic composition of primary production has changed systematically over recent centuries. 3) Nadelhoffer & Fry (1988) have shown that illuviation is unlikely to be an important process in the enrichment of $^{13}$C in deeper layers, because even without illuvial input they measured an enrichment in $^{13}$C in an incubation study.

In the model isotope fractionation during decomposition is treated as a dynamic process. Fluxes of $^{13}$C are modelled with similar process formulations as for total organic carbon (TOC= $^{12}$C + $^{13}$C), taking into account the isotope ratio's IR = $^{13}C/(^{13}C + ^{12}C)$ in various pools and $\delta^{13}$C in the input. IR is calculated for each pool after each time step. Fluxes of total C (FluxTOC) are calculated first and without isotope fractionation, concurrent fluxes of $^{13}$C might simply be calculated as Flux $^{13}$C = FluxTOC × IR.

However, isotope fractionation takes place, because the heavier isotope usually has a lower reaction rate due to a higher binding energy, and diffuses more slowly (resulting in a lower collision frequency). For a given first order reaction, the isotope fractionation factor is defined as $\beta = k_2/k_1$, in which $k_1$ and $k_2$ are the first order reaction rate constants for $^{12}C$ and $^{13}C$ respectively. The resulting isotope discrimination is $\alpha = 1 - \beta$, and thus $k_2 = k_1\ (1-\alpha)$. The overall reaction rate ($k_t$) for TOC ($^{12}C$ + $^{13}C$) differs negligibly from that for $^{12}C$ ($k_1$), because IR and $\alpha$ are both sufficiently small. This is illustrated in the following example where the ratio between $^{13}C$ and TOC is as high as 0.012 and the isotope discrimination against $^{13}C$ is 0.005. In practice IR and $\alpha$ will be lower. For a first order reaction the resulting fluxes are:

$$\begin{aligned} Flux^{12}C &= 0.988k_1 \times TOC \\ Flux^{13}C &= 0.012k_1 \times TOC\ (1 - 0.005) \\ FluxTOC &= 0.99989 \times TOC = k_t \times TOC \end{aligned}$$

Therefore, in the model the value for $k_1$, i.e. the reaction rate of $^{12}C$ was assumed to be equal to $k_t$, the overall reaction rate of ($^{12}C$ + $^{13}C$). In Table 1 the procedure followed to calculate isotope fractionation is presented. First, an auxiliary value (FLC = $k_t$ × TOC) for the total flux of $^{12}C$ + $^{13}C$ is calculated. Then, the fluxes of $^{12}C$ and $^{13}C$ are calculated as
FLC × (1-IR) and FLC × $\beta$ × IR, respectively. The resulting total flux of C is recalculated as: FluxTOC = FLC × (1 - $\alpha$ × IR). After each time step a new value for IR is obtained, calculated as:

$$IR = (POOL^{13}C + Flux^{13}C_i - Flux^{13}C_0)\ /\ (POOLC + FluxTOC_i - FluxTOC_0)$$

In the model, isotope fractionation was simulated during respiration of organic carbon for all three SOC pools in the model. Calibration of the model resulted in an isotope discrimination factor ($\beta$) of 0.0028 which is close to $\beta = 0.003$, calculated by Bertram (1986) for soil respiration.

In their study Nadelhoffer & Fry (1988) showed that not only isotope fractionation was an important control of isotopic composition, also new litter inputs were an important control. In the model, litter from the above ground biomass not only enters the rapid organic matter pool of the top layer, but a small percentage also enters into the layers to a depth of 0.2 m. This would simulate the effect of bioturbation. Both earthworms (*Pontocolex corethrurus*) and leaf-cutter ants (*Atta cephalotes*) are involved in bioturbation in the Atlantic Zone of Costa Rica (Alvarado *et al*, 1981).

Table 1. Procedure to calculate isotope ratios in SOC model.

| | Pool | Rate constant | Flux |
|---|---|---|---|
| 1 | $TOC = {}^{12}C + {}^{13}C$ | $k_t$ | $FLC = k_t \times TOC$ |
| 2 | ${}^{12}C = (1 / IR - 1) \times {}^{13}C$ | $k_1 = k_t$ | $k_t \times (1 / IR - 1) \times {}^{13}C$ |
| | | | $= k_t \times (1 / IR - 1) \times IR \times TOC$ |
| | | | $= FLC \times (1 - IR)$ |
| 3 | ${}^{13}C = IR \times ({}^{12}C + {}^{13}C)$ | $k_2 = k_1 \times (1-\alpha)$ | $k_t \times \beta \times IR \times ({}^{12}C + {}^{13}C)$ |
| | | $= k_t \times \beta$ | $= k_t \times \beta \times IR \times TOC$ |
| | | | $= FLC \times \beta \times IR$ |
| 2+3 | TOC | | $k_t \times IR \times (1 / IR - 1 + \beta) \times TOC$ |
| | | | $= k_t \times IR \times (1 / IR - \alpha) \times TOC$ |
| | | | $= FLC \times (1 - \alpha \times IR)$ |

IR = ratio between ${}^{13}C$ and $({}^{12}C + {}^{13}C)$, $\beta$ = isotope fractionation factor and $\alpha = 1 - \beta$ is the isotope discrimination factor of ${}^{13}C$ with respect to ${}^{12}C$, FLC = auxiliary value.

### *Depth dependent decomposition*

In the available SOC models, usually only the dynamics of soil organic carbon in the top 0.2 m is considered (Parton *et al.*, 1987; Jenkinson, 1990). As the topsoil contains the highest soil organic carbon levels, this seems logical. However, a considerable fraction of total soil organic carbon occurs deeper in the soil profile. Brown and Lugo (1990) found that the C content in the top 0.3 m represented only 40-52% of total carbon to 1 m depth in a humid tropical area. Veldkamp (1993a) showed that less than 50 % of the total organic carbon to a depth of 1.0 m was concentrated in the top 0.2 m. In contrast, 80 % of the carbon input to the soil by grass roots was concentrated in the top 0.2 m (Veldkamp, 1993b). A higher proportion of total carbon input than of the total carbon pool in the surface soil strongly suggests that decomposition rates in the top soil are higher than in deeper layers. This is confirmed by observations that the apparent age of soil organic carbon, as revealed by ${}^{14}C$ dating, increases with depth (Jenkinson & Rayner, 1977; Scharpenseel *et al.*, 1989). If decomposition and humification rates would be the same throughout the profile, there would be no difference in apparent age throughout the profile. A higher apparent age in deeper layers can only be achieved when the contribution of the slower pools increases with depth. Increasing ratios between SOC and microbial biomass (Insam *et al.*, 1991) might also be interpreted as evidence for decreasing

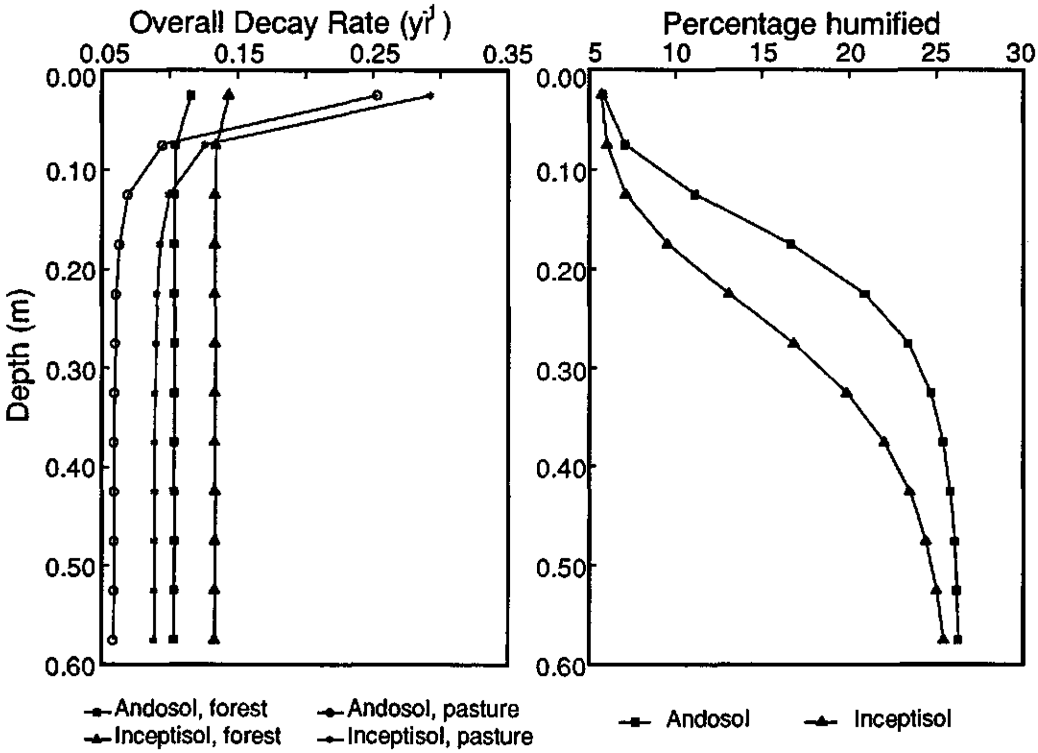

Figure 2. Left part: depth dependence of overall decay rate of rapid pool for two soil types under forest and pasture. Right part: depth dependence of percentage of decomposed material from rapid pool, which enters the active pool (is humified).

turnover rates of SOC with depth. The only decomposition model which simulates changing decomposition rates with depth was developed by Hunt (1977). Hunt (1977) discussed that he had to incorporate depth dependent decomposition rates because temperature and moisture could not explain the large differences in decomposition rates that he found. He suspected that the changing decomposition rates were related to the decreasing oxygen concentration or changes in physical structure.

In our soil organic carbon model, we divided the soil into twelve 0.05 m layers and for every separate layer decomposition was calculated following the model structure in Figure 1. Both decomposition rates and humification rates were supposed to vary with depth. We use a so-called 'biological switch' function (Thornley and Johnson, 1990) to describe the depth dependency because slope and extremes are easy to adapt independently, which facilitates calibration.

### *Conversion of forest to pasture*

When forest is cleared and replaced by pasture several changes take place regarding the carbon balance. We assumed that all stems and 90 % of branches and leaves were removed during clearing. Furthermore we assumed that all litter of the forest floor remained on the site and was thoroughly mixed with the top layer. All forest roots remained in the soil. After clearing, medium and coarse roots decomposed at rates which were 10 respectively 20 times faster than the turnover rates of their living predecessors. For carbon in leaves we used a $\delta^{13}C$ value of -31.5 ‰, for the rest of the tree carbon -30.75 ‰ was used. For the above ground pasture carbon a $\delta^{13}C$ value of -15.0 ‰ was used, for the pasture roots we used a $\delta^{13}C$ of -14.75 ‰. Vitorello *et al.* (1989) found slightly more negative numbers for leaves than for wood in a humid tropical forest in Brazil. Also $\delta^{13}C$ values of leafs from sugar cane were slightly more negative than their roots.

## Results and Discussion

### *Calibration of the model for forest and pasture sites*

The model was calibrated for two soil types: a fertile Eutric Hapludand and an infertile Andic Humitropept (Soil Survey Staff, 1990). For calibration of the carbon cycle in the forests we used data by Raich (1980, 1983) who worked in the same area of Costa Rica, completed by soil organic carbon and $\delta^{13}C$ data by Veldkamp (1993a). For the forest we assumed that the SOC levels are near steady state. On the Andic Humitropept we used a net primary production of 8.5 Mg C $ha^{-1}$ $yr^{-1}$. On the Eutric Hapludand this number was 14 Mg C $ha^{-1}$ $yr^{-1}$. Simulated steady state values of the sizes of different carbon pools in the forest for both soils are summarized in Table 2.

For calibration of the carbon cycle in the low productive pastures we used data by Veldkamp (1993a, 1993b) and Veldkamp & Weitz (1993). The net primary

**Table 2.** Simulated amount of C and corresponding turnover rates of different pools for forest in steady state situation on two different soil types in the Atlantic Zone of Costa Rica.

| | Andic Humitropept | | Eutric Hapludand | |
|---|---|---|---|---|
| | C in pools ($Mg\ C\ ha^{-1}$) | Turnover rate ($yr^{-1}$) | C in pools ($Mg\ C\ ha^{-1}$) | Turnover rate ($yr^{-1}$) |
| Leafs | 5.8 | 0.56 | 10.4 | 0.56 |
| Branches | 27.2 | 0.05 | 39.2 | 0.05 |
| Stems | 153.0 | 0.01 | 210.0 | 0.01 |
| Coarse roots | 4.1 | 0.01 | 6.3 | 0.01 |
| Medium roots | 3.1 | 0.11 | 4.7 | 0.11 |
| Fine roots | 1.5 | 1.67 | 2.3 | 1.67 |
| Forest litter | 18.7 | * | 35.1 | * |
| Rapid SOC | 33.8 | * | 61.1 | * |
| Active SOC | 19.2 | * | 44.4 | * |
| Passive SOC | 18.9 | * | 50.3 | * |

* Turnover rate variable, see Figure 2.

production of the pasture was 5 Mg C $ha^{-1}$ $yr^{-1}$ on the Inceptisol and 6 Mg C $ha^{-1}$ $yr^{-1}$ on the Andisol. For each soil type the model was calibrated simultaneously for TOC and $\delta^{13}C$ values. First, calibration was done for the forest sites, running the model for 10000 years. For this near steady state situation $\delta^{13}C$ increases gradually with depth, caused by isotope discrimination against $^{13}C$ during respiration and a resulting preferential stabilization of the heavy isotope in pools with a longer residence time. Hereafter the model was calibrated for pasture sites of 3, 5, 10 and 18 years (Andic Humitropept) and 25 years (Eutric Hapludand). Calibration was mainly done by adapting the extremes and the slope of the depth dependent total decay and humification functions until an acceptable fit of both TOC and $\delta^{13}C$ profiles was reached. In Figure 2 the depth dependent $k_{Rt}$ curve for the rapid SOC pool and the percentage of the decomposed rapid carbon which is humified to active carbon is displayed for forest and pasture sites of both soil types.

Under forest as well as under pasture sites, decomposition rates appear to be slower in the Andisol than in the Inceptisol. This could be attributed by the presence of allophane in the Andisol, which slows down decomposition of organic carbon. Decomposition rates in the topsoil of the pasture exceeded the decomposition rates in the topsoil of the forest, but below 0.1 m the $k_t$ values of the pasture are lower than those of the forest. Furthermore, the percentage of carbon which is humified increases for all sites from about 5 % in the topsoil, to about 25 % in the subsoil. There may be several explanations for these changes with depth. Depth dependency may be related to the presence of roots. Microorganisms may prefer root derived material as energy source above native soil organic matter (Kuikman *et al.*, 1992). If more root derived material is available, i.e. if the input of carbon from roots to the soil is higher, decomposition of native soil organic matter could be reduced. This would explain why decomposition rates in the pasture are higher in the topsoil, but lower in the subsoil. Another explanation may be an adverse effect of the increasing $CO_2$ concentration in the soil atmosphere at depth on decomposers. The topsoil in the pastures is usually strongly compacted (Veldkamp, 1993a), which would further decrease gas exchange between atmosphere and soil, contributing to the low decomposition rates in pastures. Below 0.1 m depth compaction may also diminish the accessibility of decomposable organic carbon for the microbial biomass, which would further decrease the rate of decomposition. Changing decomposition rates with depth have been reported before (Hunt, 1977) for decomposition in temperate grasslands. Surprisingly there has been no systematic research to this effect, which could be important if we consider long term SOC dynamics.

Model outputs and measured TOC and $\delta^{13}C$ profiles fit well, as is demonstrated for the Andic Humitropept (Figure 3). The largest differences between measurements and simulations are found in the topsoil for TOC and in the subsoil for $\delta^{13}C$. The model indicates a rapid increase in TOC in the top layer during the first years after deforestation, due to decomposition of litter of the forest floor (e.g. decomposing stems which were already present on the forest floor before clearing) which remained on the sites after forest clearing. Because stems and coarse roots were not sampled, we could not expect to observe the modelled increase in TOC after forest clearing. In the subsoil, we find the largest systematic differences between measured and simulated $\delta^{13}C$ values. This may have two causes. First, the modelled contribution of the rapid pool at this depth may be too high (Table 3). The second reason may be that the percentage of organic carbon which is humified (Figure 2) is too low in the subsoil. Veldkamp (1993a) has described a decrease in $\delta^{13}C$ values in the first years after forest clearing due to decomposing forest roots. This effect was also simulated

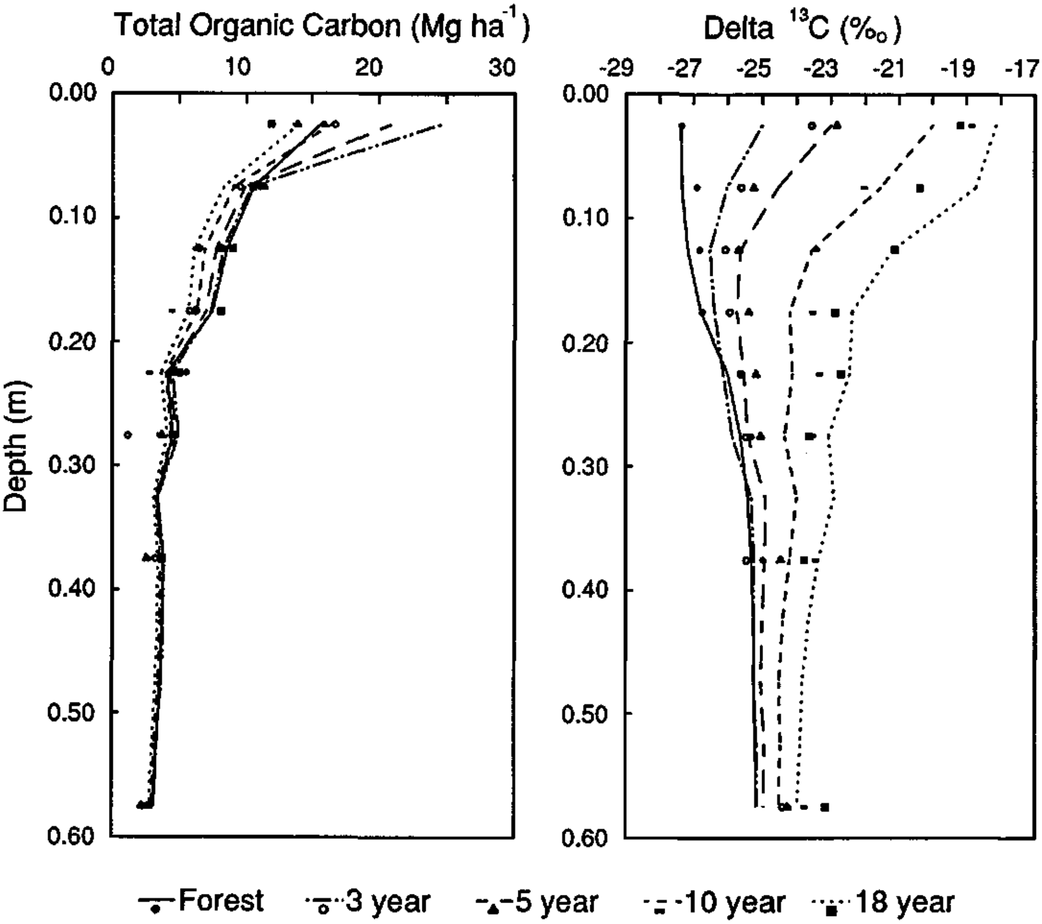

Figure 3. Left part: total organic carbon profiles for Andic Humitropept before and after conversion from forest to low productive pasture. Markers are observations, lines are simulations. Right part: $\delta^{13}C$ profiles for Andic Humitropept before and after conversion from forest to low productive pasture. Markers are observations, lines are simulations.

by the model (Figure 3) at depths between 0.2 and 0.3 m. $\delta^{13}C$ of SOC under forest at any depth invariably increases by about 5 ‰ from the 'rapid' to the 'stable' pool (Table 3). In a given pool $\delta^{13}C$ always slightly decreases with depth between 0 and 0.6 m by about 0.3 ‰. This is caused by a smaller substrate depletion at greater depth, due to slower decomposition. However, $\delta^{13}C$ of the bulk SOC increases with depth due to the increasing contribution of the more stable pools with depth.

### *Calculated $CO_2$ emissions from pastures after forest clearing*

We calculated the net $CO_2$ loss for the years following deforestation, assuming that all carbon is lost by $CO_2$ emission. We calculated $CO_2$ emissions for the calibrated sites (with *Axonopus compressus* as low productive pasture), as well as a hypothetical high-productive pasture (in this case: *Brachiaria dictyoneura*) planted after forest clearing. Information on the root and shoot production of both pasture types is given by Veldkamp (1993b). For both pastures above ground production was obtained from sequential harvesting, and below ground production from pulse labelling with $^{14}C$. The net primary production for the high productive pasture we used was 8.5 Mg C $ha^{-1}$ $yr^{-1}$ on the Inceptisol and 12 Mg C $ha^{-1}$ $yr^{-1}$ on the Andisol. We assumed that the decomposition and humification rates in the soil were the same for the natural and improved pasture. The calculated $CO_2$ emissions (Figure 4) peak in the first years after forest clearing, especially for soils under the low productive pasture. The annual $CO_2$ emissions slowly decrease asymptotically until at least 20 years after establishment. All pastures remain a $CO_2$ source. The net $CO_2$ emissions are higher for the Andisol than for the Inceptisol. Furthermore the emission for the low productive pasture is much higher than for the high productive pasture. Table 4 summarizes the calculated cumulative net $CO_2$ emission over 20 years. The Andisol emits about twice as much $CO_2$ as the Inceptisol under a given grass species. Introducing an improved pasture with Brachiaria, may reduce the

**Table 3.** Simulated $\delta^{13}C$ values and contribution of different pools to TOC after 10000 yr in Andic Humitropept under forest for different soil organic carbon pools at different depths.

| | Rapid pool | | Active pool | | Passive pool | | TOC |
|---|---|---|---|---|---|---|---|
| Depth (m) | $\delta^{13}C$ | % of total | $\delta^{13}C$ | % of total | $\delta^{13}C$ | % of total | $\delta^{13}C$ |
| 0.00-0.05 | -28.50 | 67 | -25.87 | 23 | -23.51 | 10 | -27.41 |
| 0.10-0.15 | -28.59 | 61 | -25.96 | 26 | -23.66 | 13 | -27.26 |
| 0.35-0.40 | -28.62 | 26 | -26.02 | 31 | -23.72 | 43 | -25.53 |
| 0.55-0.60 | -28.71 | 19 | -26.12 | 29 | -23.82 | 52 | -25.41 |

net $CO_2$ emission by about 60 %. Of course this is only valid if a constant high dry matter production is maintained. Gross $CO_2$ emission shows no relation with net $CO_2$ emission (Table 4). The highest gross emission is found on the Brachiaria pasture on the Andisol, but the highest net emission is found for the Axonopus pasture on the Andisol. Net $CO_2$ emission is not only a result of the amount of $CO_2$ produced, but also of the input of carbon into the soil. So $CO_2$ emission measurements from soil would not say much about the net $CO_2$ emission. As is demonstrated by Figure 4, net $CO_2$ emission is highly variable with pasture age. Especially the first years after conversion from forest to pasture emissions are high, but after 10 to 15 years, net emissions drop to values which in most cases are less than 1 Mg C $yr^{-1}$. The analogy with $N_2O$ and NO emissions which were measured on the same Inceptisol deforestation sequence (Keller *et al.*, 1993) is striking and demonstrates the close link between the carbon and nitrogen cycle in this soil.

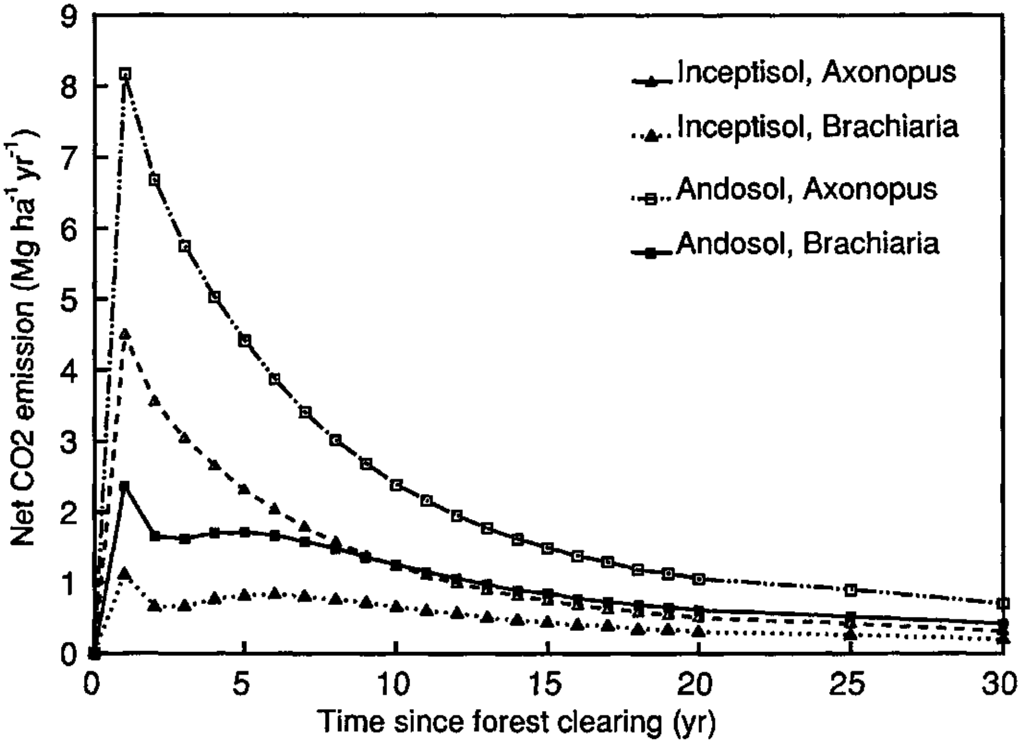

Figure 4. Simulated net $CO_2$ emissions for two soil types under a low productive (Axonopus) and high productive (Brachiaria) pasture.

### *Using C isotopes to improve simulation of long-term SOC dynamics*

Recently, attempts have been made to simulate large-scale and long-term consequences of climate and management changes on soil organic matter dynamics with the CENTURY model (Parton *et al.*, 1987; Schimel *et al.*, 1990) and the ROTHAMSTED model (Jenkinson *et al.*, 1991). Critical parameters for long-term simulations in the CENTURY model are the decomposition rates of the 'slow' and 'passive' soil organic matter fractions which have turnover times of decades to millennia respectively. In ROTHAMSTED the main source of error in long term simulations is the presence of 'biologically inert' carbon. Calibration of the turnover times and extent of these pools in CENTURY is based on 'long-term' incubation studies (5 to 10 years). But even ten years of incubation is a short period to calibrate pools with a turnover time of several decades to millennia. In case of ROTHAMSTED $^{14}C$ data have been used to adjust the amount of the 'inert' pool to more realistic values, but neither CENTURY nor ROTHAMSTED has dynamically included $^{13}C$ and $^{14}C$. Carbon isotopes may, however, be very helpful in the calibration of 'slow' and 'passive' pools. This can be illustrated with the changes in SOC and $\delta^{13}C$ profiles in time (Figure 3a, b). In the left part of this figure we see that especially below 0.3 m there have been no clear changes in the SOC levels in the 18 years which have passed since forest clearing. In contrast, the $\delta^{13}C$ values have changed considerably (Figure 3b) and we used these changes to adjust the humification rates at shallower depth. After a change from C3 to C4 vegetation, carbon isotope signals in most cases change more rapidly than the soil organic matter levels. If the long term consequences of climatic change are to be simulated, calibration of pools with long term turnover times is critical. Including C

**Table 4.** Cumulative net $CO_2$ emission, gross $CO_2$ emission and loss of soil organic carbon in the first 20 y after conversion of forest to pasture with *Axonopus compressus* or *Brachiaria dictyoneura*.

| | *Brachiaria dictyoneura* | | *Axonopus compressus* | |
|---|---|---|---|---|
| | Andisol | Inceptisol | Andisol | Inceptisol |
| | --------Mg C $ha^{-1}$-------- | | | |
| Cum. net $CO_2$ emission | 24.7 | 12.0 | 60.5 | 31.5 |
| Cum. gross $CO_2$ emission | 250.0 | 172.2 | 166.1 | 122.3 |

isotopes into SOC simulation models will improve calibration. Finally, an important part of the 'passive' or 'biologically inert' pools is found below 0.2 m. Layers, deeper than 0.2 m should therefore also be considered in simulations of long term soil organic carbon dynamics.

## Conclusions

The net $CO_2$ emissions from pastures established after forest clearing can be reduced by about 60 % if a grass species with a higher dry matter production will be introduced. Including carbon isotopes and considering deeper layers in dynamic soil organic carbon models will improve the performance of these models, especially for simulations which go beyond a few decades.

## Acknowledgements

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## *Chapter 7*

# Epilogue

# Epilogue on carbon budgets, isotope- and modelling techniques in soil organic matter studies

The objective of this thesis was to quantify the changes in soil organic carbon (SOC) storage and the resulting release of $CO_2$ after the conversion of tropical rain forest to pasture on two contrasting soil types in the humid tropics of Costa Rica. For this study deforestation sequences were selected on an Eutric Hapludand and an Andic Humitropept, for which a deforestation map was made. The methods that I used to quantify changes in soil organic carbon and input from carbon to the soil, were based on carbon isotope techniques. Integration of the results of the different studies was done with help of a dynamic simulation model. In this closing chapter I use the main conclusions to elaborate on aspects of carbon budget calculations, isotope- and modelling techniques.

## Main conclusions.

* Aerial photographs offer a good means of analysing deforestation patterns.
  -Rates of deforestation in the Atlantic Zone of Costa Rica demonstrate a close relation with accessibility by human settlers and soil quality.

* Pulse labelling with $^{14}CO_2$ is a relatively simple method to obtain quantitative data about carbon input into the soil by grass roots.
  -In the Atlantic Zone of Costa Rica, root dry matter production of an improved pasture like *Brachiaria* is about twice the root production of a low-productive species like *Axonopus*. Root biomass of *Brachiaria* is about three times the root biomass of *Axonopus* due to higher residence time of carbon in the root biomass of *Brachiaria* as compared to *Axonopus.*
  -Root exudates of grass plants do not contribute appreciably to the longer term carbon storage, either because exudation rate is small or because exudates decompose quickly and completely.

* The $\delta^{13}C$ method is very valuable in soil organic matter studies. For a correct application of the method, detailed information of changes in bulk densities accompanying changes in land use is vital.
  -An uncertainty analysis of the $\delta^{13}C$ method demonstrated that the output of the $\delta^{13}C$ method in soil organic matter studies is highly variable due to variations in the input data. Spatial variability is the main source of the uncertainty in input data. Variations due to sampling error and short-scale

variability are considerable and should not be ignored.

* The performance of dynamic soil organic carbon models will be improved, especially for long term simulations, if carbon isotope fractionation and deeper layers will be included into these models.

-Net $CO_2$ emissions peak in the first years after deforestation, but drop to lower values after about 10 to 15 years.

-The net $CO_2$ emissions from pastures established after forest clearing can be reduced by about 60 % if a grass species with higher dry matter production (e.g. *Brachiaria*) will be introduced.

## Carbon budgets and bulk density.

In most published studies on soil organic matter dynamics bulk density does not get the attention it should have. When calculations on carbon budgets are performed, reliable bulk densities are of crucial importance to convert soil organic carbon data from g C $g^{-1}$ to Mg $ha^{-1}$. In many cases a change in land use is accompanied by a change in bulk density as was demonstrated for the conversion from forest to pasture, where compaction of the top soil was caused by cattle trampling (Chapter 4). If we assume that compaction is the result of a decrease in porosity, then the logical consequence is that the soil horizon has decreased in thickness. If no correction is made for this decrease, then the result will be an overestimation of soil organic carbon storage in the soils which have a higher bulk density due to the land use. This was clearly demonstrated in Chapter 4. In too many soil organic matter studies, bulk densities and changes in bulk densities are ignored. The present confusion about the effect of pastures on soil organic carbon levels may be partly explained by this 'compaction effect'.

## The use of carbon isotopes in soil organic matter studies.

Soil organic matter levels are the result of complex production and decomposition processes. In soil organic matter studies, traditionally decomposition processes are emphasized, but decomposition is only part of the story. The input of carbon from vegetation is one of the key factors in the carbon cycle as was demonstrated in Chapter 3. Roots play an important direct role in the production of soil organic carbon. However, roots may play a considerable indirect role in decomposition processes (Kuikman et al., 1992). One of the best direct ways to reveal the role of vegetation and roots in the study of carbon input into the soil is by means of $^{14}$C tracers. For practical reasons most $^{14}$C labelling experiments are conducted in laboratory conditions. More field experiments are necessary to obtain data which can be better used for the calibration and validation of soil organic carbon models.

A relatively new method in soil organic matter studies is the use of differences in natural $^{13}$C isotope levels in vegetation (C3 and C4 vegetation) and soil organic matter (Balesdent et al., 1987). This method is increasingly popular in soil organic matter studies where a change from C3 to C4 vegetation has occurred (or vice versa), because $^{13}$C provides a natural tracer, which can be used to distinguish the source of soil organic matter. So far, isotope fractionation which occurs during decomposition has been neglected as 'noise'. In these conventional studies the $^{13}$C profile under forest is always used as original situation, with which changes in $^{13}$C are compared. Chapter 4 demonstrated that this may cause considerable errors, especially in the first years after forest clearing, when an extra input of organic matter with a low $^{13}$C value is caused by decomposing tree roots.

In chapter 6 we illustrated the difference in $^{13}$C levels among soil organic carbon pools with different turnover rates. The soil organic carbon pool with the higest decomposition rate ('rapid pool' in chapter 6) is the one with the lowest $^{13}$C level. If forest is cleared and is not being replaced by another vegetation, input of organic carbon stops and the contribution of the rapid pool to the soil organic carbon will decrease rapidly. Consequently, $^{13}$C levels in the topsoil will increase, even without C4 vegetation. In the conventional $^{13}$C studies the observed increase in $^{13}$C levels is explained by the input from C4 organic matter which contains higher levels of $^{13}$C. Disregarding the increase in $^{13}$C as a result of ageing of soil organic carbon may lead to an overestimation of the soil organic carbon derived from C4 vegetation. In isotope budget calculations the $^{13}$C values deeper in the forest profile could be used as $^{13}$C value of the soil organic carbon derived from the original forest, to correct for this effect. Conventional isotope budget calculations assume that changes in $^{13}$C levels of soil organic carbon are caused exclusively by a change in vegetation. As was discussed, several other processes exist, which cause changes in isotope levels. The best way to integrate these dynamic processes is by dynamic simulation.

## Spatial variability and sampling strategy.

Spatial variability is a major source of uncertainty in the output of the $^{13}C$ method, as was demonstrated in chapter 5. Therefore, if the $\delta^{13}C$ method, or simple budget calculations are applied in soil organic matter studies, the variability of the input data should be quantified. In most budget studies, not spatial information is required and the minimum number of samples can be calculated from 'traditional' random sampling techniques. Only if information on spatial structure is required, a more extensive sampling scheme is necessary. Ideally, the results of carbon budget calculations are accompanied by the degree of precision and confidence level of the output. The number of observations to calculate changes in SOC will be the result of balancing the required precision, the required confidence level and the available time and money.

## Scenario studies and scales in time and space.

Soil organic carbon models like CENTURY and ROTHAMSTED have been used for scenario studies on global scale. However, extrapolation of model simulations in the future should be done with care. In chapter 6, we discussed that calibration of pools with long-term turnover times is critical especially for scenario calculations which go beyond a few decades. Inclusion of C isotopes ($^{13}C$ and $^{14}C$) into SOC simulation models will improve calibration of SOC pools with slow turnover times, because levels of C-isotopes change more rapidly than SOC levels. Furthermore, an important part of the 'passive' or 'biologically inert' pools is found below 0.2 m. Layers, deeper than 0.2 m should therefore also be considered in simulations of long term soil organic carbon dynamics.

Dynamic soil organic matter models like CENTURY or ROTHAMSTED are one-point models which also have been used for global scenario studies (Schimel et al., 1988; Jenkinson et al., 1991), using one simulation per climatic zone. It would be more realistic to do a set of simulations per climatic zone, which includes the natural variations in input data characteristical for that zone. Using so-called Monte Carlo techniques, a distribution of e.g $CO_2$ fluxes for one climatic zone would be obtained.

In the present scenario studies, input is based on natural vegetation in the climate zones plus an additional 'cultivated land'. This approach may be acceptable as a first step, but future scenario studies will need a different approach. Recent studies show that the largest emissions of gases ($CO_2$ and

$N_2O$) from the soil take place during the first years after land use changes (Keller et al., 1993; Veldkamp et al., 1993). After some time (in most cases 10 to 15 years), the system approaches a new equilibrium and the emissions drop to relatively small values. Future scenario studies therefore must emphasize the transient periods, including land use changes in a dynamical way. Relations between e.g. soil quality, accessibility and deforestation rates, as was discussed in chapter 2, will probably play an important role in such an approach.

## Implications for future research.

As usually, this study ends with a lot of unanswered questions. One of the main unsolved topics is the processes behind the depth-dependency of soil organic carbon turnover rates. Many mechanisms could cause changes in decomposition rates with depth. The most likely candidates were discussed in chapter 6, but there are other processes which could cause this effect. Getting more insight in SOC dynamics in the subsoil is crucial to make more realistic scenario studies. Research to isotope fractionation during decomposition is needed. In our model we include isotope fractionation during respiration, but more research is needed in the processes which control discrimination against heavier isotopes.

The quality of scenario studies will also depend strongly on feedbacks between vegetation and soil organic matter. We did not include any of this mechanism in our modelling study (chapter 6) but if we want to simulate realistic future scenarios, changes in litter quality, increased photosynthesis potential, increased decomposition rates in the rhizosphere, etc., will play a key role.

In chapter 3 and 6 we showed that quantification of carbon input from plant roots is essential to complete the picture of soil organic carbon dynamics. Although this point has been stressed during several recent workshops, I would like to emphasize that no soil organic matter budget study is complete without a study of above and below ground carbon input from vegetation.

As was mentioned before, calibration of soil organic matter pools with slow turnover times will be essential in long-term studies. The best way to achieve this is the dynamic inclusion of isotopes in soil organic carbon models.

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# Samenvatting

# Samenvatting

Het tropisch regenwoud wordt gekapt met een snelheid die waarschijnlijk hoger is dan ooit tevoren. Naast lokale verschijnselen zoals bodemdegradatie en nutrienten uitspoeling heeft dit consequenties die veel verder reiken zoals blijkt uit de stijgende concentraties in de atmosfeer van zogenaamde 'broeikas gassen' zoals kooldioxide en lachgas. Deze gassen komen versneld vrij uit de bodem nadat een tropisch regenwoud wordt gekapt. De bijdrage van ontbossing aan de stijgende concentraties aan $CO_2$ in de atmosfeer wordt in recente studies geschat tussen de 0.42 en 1.60 Pg C $yr^{-1}$ (1 Pg = $10^{15}$ g). Hiervan is 0.1 tot 0.3 Pg C $yr^{-1}$ afkomstig van de afbraak van bodem organische stof. De hoeveelheid aan $CO_2$ die vrijkomt door ontbossing wordt alleen overtroffen door de hoeveelheid $CO_2$ die ontstaat bij de verbranding van fossiele brandstoffen (op dit moment circa 5.3 Pg C $yr^{-1}$).

Het doel van deze studie was het kwantificeren van veranderingen in bodem organische stof en de bijbehorende emissie van $CO_2$ die optreden na ontbossing van het tropisch regenwoud. Hiervoor werden onderzoekslokaties vergeleken die op verschillende tijdstippen in het verleden ontbost zijn (zgn. ontbossingssequenties). Omdat het tijdstip van ontbossing zo nauwkeurig mogelijk bekend moest zijn, werd een ontbossingskaart gemaakt van een deel van de Atlantische Zone van Costa Rica. Hiervoor werden luchtfoto series uit de periode 1952 tot 1984 gebruikt. Met behulp van GIS technieken werd de ontbossingskaart gecombineerd met een beschikbare bodemkaart om de onderzoekslokaties te selecteren. Uit een analyse van de ontbossingskaart bleek dat er een duidelijke relatie bestaat tussen bodemkwaliteit, bereikbaarheid en ontbossingssnelheid.

De hoeveelheid aan organische stof in een bodem wordt bepaald door dynamische input en afbraak processen. De input van organische koolstof in de bodem van twee grassoorten werd bepaald met behulp van een zogenaamd pulselabelling experiment met $^{14}C$ als tracer. Uit dit experiment volgde dat de ondergrondse droge stof productie van een verbeterd, hoog productief gras (*Brachiaria dictyoneura*) ongeveer het dubbele was van de droge stof productie van een lokaal, laag productief gras dat veel in de Atlantische Zone van Costa Rica voorkomt (*Axonopus compressus*). De wortelmassa van Brachiaria was echter drie maal zo hoog als de wortelmassa van Axonopus. Dit verschil kon worden verklaard door een verschil in verblijftijd van koolstof in de wortels van beide grassoorten. Uit het pulselabel experiment bleek bovendien dat de directe bijdrage van wortelexudaten aan de koolstof dynamiek op langere termijn te verwaarlozen is. De verklaring hiervoor was dat of de exudatie snelheid klein was, of de afbraak van wortelexudaten snel en vrijwel compleet was.

De afbraak van bodem organische stof werd gemeten met behulp van de zogenaamde $\delta^{13}C$ methode. Bij deze methode wordt gebruik gemaakt van het natuurlijk verschil aan $^{13}C$ tussen C3 en C4 vegetatie om aan te tonen wat de bron van de bodem organische stof is. De methode is toepasbaar na een verandering van C3 naar C4 vegetatie, of omgekeerd. Met behulp van deze methode kon zeer waardevolle informatie worden verkregen over de afbraaksnelheid van bodem organische stof. Een analyses van de veranderingen in de specifieke dichtheid van bodemhorizonten na ontbossing bleek cruciaal te zijn om deze methode op een correcte manier toe te passen. Uit een onzekerheidsanalyse van de $\delta^{13}C$ methode bleek dat de resultaten van de methode zeer gevoelig zijn voor variaties in de gebruikte gegevens. De belangrijkste oorzaak van deze variaties was de ruimtelijk variabiliteit. Echter, variaties door meetfouten en variaties op korte afstand waren aanzienlijk en konden niet worden verwaarloosd. De verzamelde informatie over de input van organische stof in de bodem en de afbraaksnelheid van bodem organische stof werd geintegreerd in een organisch stof model. Het gebruikte model simuleert onder andere isotoop fractionering tijdens afbraak van organische stof, en beschrijft de dynamiek van organische stof in diepere lagen. Met dit model werden de waargenomen veranderingen in bodem organische stof en de corresponderende veranderingen $\delta^{13}C$ die optreden na de overgang van bos naar gras nader gekwantificeerd. Gedurende 20 jaar na ontbossing bedroeg de cumulatieve netto $CO_2$ emissie 31.5 Mg C $ha^{-1}$ voor een Andic Humitropept en 60.5 Mg C $ha^{-1}$ voor een Eutric Hapludand. Beide voor het geval een laag productief gras (Axonopus) wordt gezaaid. Uit modelberekeningen blijkt verder dat deze netto emissies gereduceerd kunnen worden met ongeveer 60 % indien een hoog productief gras (Brachiaria) geintroduceerd zou worden in plaats van het laag productief gras.

# Curriculum vitae

Edzo Veldkamp werd geboren op 14 januari 1965 in Sittard. In 1983 behaalde hij het Athenuem-ß diploma aan de Scholengemeenschap St. Michiel te Geleen. In datzelfde jaar begon hij aan de studie Bodemkunde aan de Landbouw Universiteit Wageningen, waar hij in 1989 cum laude afstudeerde. De doctoraalvakken waren Bodeminventarisatie en Landevaluatie, Bodemvorming en Ecopedologie en Geomorfologie.

Vanaf september 1989 tot augustus 1993 was hij als Onderzoeker in Opleiding (OIO) werkzaam bij de stichting voor wetenschappelijk onderzoek in de tropen (WOTRO) en gestationeerd bij de vakgroep Bodemkunde en Geologie van de Landbouwuniversiteit.

Vanaf september 1993 werkt hij als Postdoc binnen het onderzoeksproject: 'Effect of land use change on soil emissions of $N_2O$ and NO' gefinancierd door de U.S. Environmental Protection Agency (EPA) en is gestationeerd op het biologisch onderzoeksstation 'La Selva' in Costa Rica.