Forest Ecology and Management 108 (1998) 147–166

Soil carbon changes from conversion of forest to pasture in Brazilian Amazonia

Philip M. Fearnside [a,*], Reinaldo Imbrozio Barbosa [b]

[a] Instituto Nacional de Pesquisas da Amazônia, Coordenadoria de Pesquisas em Ecologia, Caixa Postal 478, 69011-970 Manaus, Amazonas, Brazil
[b] Instituto Nacional de Pesquisas da Amazônia, Núcleo de Pesquisas de Roraima, Caixa Postal 96, 69301-970 Boa Vista, Roraima, Brazil

Received 15 September 1997; accepted 16 December 1997

Abstract

Soils in Brazilian Amazonia may contain up to 136 Gt of carbon to a depth of 8 m, of which 47 Gt are in the top meter. The current rapid conversion of Amazonian forest to cattle pasture makes disturbance of this carbon stock potentially important to the global carbon balance and net greenhouse gas emissions. Information on the response of soil carbon pools to conversion to cattle pasture is conflicting. Some of the varied results that have been reported can be explained by effects of soil compaction, clay content and seasonal changes. Most studies have compared roughly simultaneous samples taken at nearby sites with different use histories (i.e., 'chronosequences'); a clear need exists for longitudinal studies in which soil carbon stocks and related parameters are monitored over time at fixed locations. Whether pasture soils are a net sink or a net source of carbon depends on their management, but an approximation of the fraction of pastures under 'typical' and 'ideal' management practices indicates that pasture soils in Brazilian Amazonia are a net carbon source, with the upper 8 m releasing an average of 12.0 t C/ha in land maintained as pasture in the equilibrium landscape that is established in the decades following deforestation. Considering the equilibrium landscape as a whole, which is dominated by pasture and secondary forest derived from pasture, the average net release of soil carbon is 8.5 t C/ha, or 11.7×10^6 t C for the 1.38×10^6 ha cleared in 1990. Only 3% of the calculated emission comes from below 1 m depth, but the ultimate contribution from deep layers may be substantially greater. The land area affected by soil C losses under pasture is not restricted to the portion of the region maintained under pasture in the equilibrium landscape, but also the portion under secondary forests derived from pasture. Pasture effects from deforestation in 1990 represent a net committed emission from soils of 9.2×10^6 t C, or 79% of the total release from soils from deforestation in that year. Soil emissions from Amazonian deforestation represent a quantity of carbon approximately 20% as large as Brazil's annual emission from fossil fuels.

Keywords: Soil carbon; Amazonia; Deforestation; Tropical forest; Greenhouse gases; Global warming; Pasture; Organic matter; Brazil; Carbon dioxide

* Corresponding author.

0378-1127/98/$19.00
PII S0378-1127(98)00222-9

1. Introduction: the tropical soil carbon controversy

The Intergovernmental Panel on Climate Change (IPCC) has, so far, not encouraged inclusion of carbon fluxes from soils under cleared tropical forests in the national inventories now being compiled under the Framework Convention on Climate Change (FCCC). The reporting instructions state that ''There is no scientific consensus on whether clearing leads to significant soil carbon loss in tropical forests. This calculation is optional for tropical forests'' (IPCC/OECD Joint Programme, 1994; Vol. 2, p. 5.10). The instructions explain that ''The basic calculations allow but do not encourage estimation of soil carbon loss after clearing of tropical forests. There are research results which indicate that conversion of tropical forests to pasture may or may not result in loss of soil carbon'' (IPCC/OECD Joint Programme, 1994; Vol. 3, p. 5.27). This is based on citation of a series of studies, some of which indicate losses of soil carbon (Buschbacher, 1984; Cerri et al., 1991; Fearnside, 1980a, 1986a), and some of which do not (Lugo et al., 1986; Keller et al., 1986), pointing out that the last of these indicates ''that clearcutting of tropical forests does not appear to release soil carbon'' (IPCC/OECD Joint Programme, 1994; Vol. 3, p. 5.45). However, the study in question (Keller et al., 1986; p. 11,798) did not measure soil carbon or draw inferences about it, but rather measured net emissions of CO_2 and other gases from soil under forest and in an adjacent unburned clearcut (without pasture grass). Emissions are not the same as changes in carbon stocks: the emission of carbon can remain unchanged while reduction of the rate of carbon input to the soil results in a drawdown of the carbon stock. In the case of the study by Lugo et al. (1986), the authors concluded that carbon stocks had increased in pasture soils in Puerto Rico as compared to the stocks that had formerly existed at the same sites when under agriculture, *not* that the stocks under pasture were higher than those under mature forest. The stocks under pasture and nearby forest were described as ''similar'' or ''about the same'' (Lugo et al., 1986; pp. 191, 193). The conclusion of Lugo et al. (1986; p. 194) that pasture soils 'may be sinks' of carbon, therefore, is limited to the effects of replacing non-forest land uses with pasture, not the conversion of tropical forests to pasture as is now occurring on a large scale in Brazilian Amazonia.

2. Land-use conversion in Brazilian Amazonia

The cumulative area deforested in Brazilian Amazonia (Fig. 1) is estimated to have reached 470×10^3 km^2 by 1994 (Brazil, Instituto Nacional de Pesquisas Espaciais, 1996; see Fearnside, 1997a). Of this total, approximately 45% (assuming 1990 values) was occupied by actively grazed cattle pastures from conversion of tropical forests, 28% was post-1970 secondary forest derived from abandoned pasture, and 2% was degraded pasture (Fearnside, 1996). The remainder was farmland (5%), post-1970 secondary forest derived from farmland (2%), and pre-1970 secondary forest (17%). As long as pasture is maintained for cattle, it is subject to periodic fires that can affect biogeochemical cycles and climate due to their emissions of trace gases such as CH_4, CO, N_2O and NO_x (CO_2 is also emitted from above-ground biomass, but, unlike the trace gases, the amount emitted is later reabsorbed when the pasture regrows).

The conversion of forest to pasture causes large net releases of carbon (mostly from biomass) in the form of CO_2, in addition to trace gases (Fearnside, 1997b). Emissions for 1990 are of particular interest

Fig. 1. Brazil's Legal Amazon region and locations mentioned in the text.

because this is the base year for the national inventories now being prepared under the United Nations Framework Convention on Climate Change (UN-FCCC). Carbon stocks in replacement vegetation biomass and original forest remains that are present in the converted system have received attention in a variety of estimates of carbon dynamics (Buschbacher, 1984; Uhl and Jordan, 1984; Uhl et al., 1988; Uhl and Kauffman, 1990; Fearnside et al., 1993; Kauffman et al., 1995; Barbosa and Fearnside, 1996). In addition to biomass, soils in the tropics also represent an important storage compartment and potential source of carbon release to the atmosphere (Houghton et al., 1983). The upper 1 m of the world's soils has been estimated to contain 1220 Gt of organic carbon (Sombroek et al., 1993). Tropical soils account for 11–13% of all carbon stored in the world's soils (Brown and Lugo, 1982; Post et al., 1982). Soils under the original vegetation in the Brazilian Legal Amazon to a depth of 1 m are estimated to have contained 47 Gt C, of which 21 Gt C (45%) were in the top 20 cm (Moraes et al., 1995).

Carbon below a depth of 1 m has traditionally been regarded as inert, and presumed to be unaffected by changes in land use. However, Nepstad et al. (1994) have presented isotopic evidence indicating that up to 15% of the deep soil (1–8 m depth) carbon stock turns over on annual or decadal time scales, and that the loss of deep roots by replacement of forests with pastures could release substantial amounts of carbon over time scales as short as a decade. The study by Nepstad et al. (1994), near Paragominas, Pará, indicated that including the carbon stock between 1 and 8 m depth increases the total stock by a factor of approximately 2.9 relative to the stock in the top meter. A previous estimate based on extrapolation to 5 m depth of a 3 m profile near Manaus had estimated the total stock to be increased by a factor of 1.8 relative to the stock in the top meter (Cerri and Volkoff, 1987; p. 37). The results of Nepstad et al. (1994) imply that the soil to 8 m depth in the Legal Amazon contained (prior to changes in the portion of the region that is now cleared) a total stock of soil carbon of 136 Gt. This value undoubtedly overestimates the true one, as not all soils in the region reach 8 m in depth. Soils on the Içá formation of Central Amazonia and many soils on the Brazilian and Guianan shields are less deep than those of the Paragominas area (W.G. Sombroek, personal communication, 1997). Estimates of soil depth and deep layer carbon stocks are needed for each soil type in order to derive a better estimate for the soil carbon stock in the region as a whole. However, the carbon stock is clearly sufficiently large that even small percentage changes in these stocks would translate into fluxes large enough to be climatically significant, indicating the importance of reducing uncertainty concerning the amount, and even the direction, of soil carbon changes resulting from deforestation.

3. Soil carbon changes

3.1. Short-term effects of conversion

The carbon stock in any soil is the result of a balance between inflows and outflows to the pool; in the case of tropical soils, the rates of both inflow and outflow are substantially higher than is the case in other parts of the world, making tropical soil carbon stocks respond rapidly to any changes in the flux rates. Small increases in temperature can substantially increase the rate of soil respiration in the tropics, leading to rapid drawdown of carbon pools (Townsend et al., 1992). The conclusion of Townsend et al. (1992) that large carbon emissions from tropical soils could result from the temperature increases expected from global warming apply equally well to the soil temperature increases that result when forests are converted to pastures.

Burning generally does not produce soil temperatures sufficient to oxidize carbon directly (Sánchez, 1976; p. 373). However, where heavy logs burn or vegetation is piled up, the soil temperature increases substantially (Zinke et al., 1978; p. 144; see also Nye and Greenland, 1960; p. 70). Soil sampling can easily miss this, either by deliberate avoidance of these 'atypical' microsites, or by a low number of samples (sometimes relying on a single soil pit). Studies are lacking that estimate the percentage of area of each type under sample points and stratify soil sampling among microsites of differing burn intensity.

Despite the small long-term effect of burning on soil carbon, short-term changes can result from the effect of the burn on the microbial biomass of the soil. Laudelot (1961) and Suares de Castro, 1957 (cited by Sánchez, 1976; p. 373) report sharp microbial biomass decreases following burns in shifting cultivation systems. Cerri et al. (1985) found that deforestation followed by burning eliminated all of the microbial biomass from the top 10 cm of soil, and two-thirds of the total in the soil profile. Any decrease in soil carbon as a direct result of killing microbial biomass would be small: the carbon stock in this compartment under natural forest in Capitão Poço, Pará has been found to be only 1.3% of the total carbon stock in the top 15 cm, where all of the microbial biomass is concentrated (Cerri et al., 1985). During the period when the microbial biomass remains depressed, the net rate of carbon accumulation in the soil should increase because soil respiration would be reduced. Following the initial decrease, the microbial population is believed to rebound to a level higher than it was prior to the burn (Nye and Greenland, 1960; p. 72).

Recent studies in Amazonia have found pasture burning to cause no significant instantaneous changes of carbon in the upper layers of soil. Barbosa (1994) examined the carbon stock in the top 20 cm of soil in a pasture in Apiaú, Roraima, and found a non-significant decrease (−2.6 t C/ha) between pre-burn samples and those taken seven days after the burn. Kauffman et al. [1] also found no difference in the stocks of carbon pre-burn (36.2 t C/ha) and post-burn (36.4 t C/ha) in the top 10 cm of soil under a pasture with typical management in Marabá, Pará. However, other studies in Amazonia have found net reduction in carbon stocks in the soil in different systems in measurements made within a short time after burning (< 1 year). For example, in burning secondary forests of different ages in Altamira, Pará, Guimarães (1993) found a mean loss of 36 t C/ha from the top 20 cm of soil after burning. Cerri et al. (1991; p. 254), working near Manaus, Amazonas, reported a 9-t/ha soil carbon release in the 0–20 cm layer from conversion of primary forest to burned forest, and an additional 3 t/ha release by the time the pasture was one year old. Kauffman et al. (1995) found a reduction of 7.2 t C/ha near Marabá, Pará and 5.8 t C/ha in Santa Barbara, Rondônia in the top 10 cm of soil in an initial burn.

3.2. Establishment of a new equilibrium

The instantaneous release of soil carbon at the time of burning is followed by establishment of a new equilibrium over the medium to the long term. Studies of medium and long-term changes (> 1 year) indicate mixed results on the increase or decrease of soil carbon when tropical forests are converted.

Some authors have considered this stock to be stabilized below the level in primary forest (Cunningham, 1963; Falesi, 1976; Hecht, 1982; Sánchez et al., 1983; Allen, 1985; Mann, 1986). Fearnside (1997b) (using data from Paragominas, Pará, and Suiá Missu, Mato Grosso by Falesi, 1976 and Hecht, 1981) found that soil carbon levels under 10–11-yr-old pastures imply an average net release of 3.92 t C/ha (a value revised in this paper based on alternative bulk density values). Veldkamp (1994) found a net loss of 2–18% of carbon stocks in the top 50 cm of forest-equivalent soil after 25 years under pasture in lowland Costa Rica. Others have found the reverse (Moraes et al., 1996; Neill et al., 1996). In studies of two 20-yr chronosequences at Fazenda Nova Vida, Rondônia, Moraes et al. (1996) found increases of 19% and 17% to a depth equivalent to 30 cm of forest soil (22% and 5%, respectively, if not corrected for clay content). In two other 20-yr chronosequences in the same part of Rondônia, Moraes et al. [2] (cited by Moraes et al., 1995; p. 77) found an increase in one chronosequence and no significant change in the other. Another study of the layer equivalent to 30 cm of forest soil in the same area (M. Grzebyk, unpublished data cited by Moraes et al., 1996; p. 77), found a decrease of 5% in the carbon stock in 3-yr-old pasture, followed by increases of 5% in 9-yr-old pasture and 10% in 20-yr-old pasture. Neill et al. (1996) found an increase in

[1] Kauffman, J.B., Cummings, D.L., Ward, D.E., Babbitt, R., no date. Fire in the Brazilian Amazon: 2. Biomass, nutrient pools and losses in cattle pastures. *Oecologia* (in press).

[2] Moraes, J.F.L. de, Neill, C., Volkoff, B., Cerri, C.C., Melillo, J., Lima, V.C., Steudler, P.A., no date. Soil carbon and nitrogen stocks following conversion to pasture in the western Brazilian Amazon Basin. (manuscript).

soil carbon concentration to a depth of 30 cm in pastures five or more years old in very well-managed pasture at Fazenda Nova Vida, Rondônia. Some studies have found no significant change (Buschbacher, 1984; Choné et al., 1991; [1]).

It is logical to expect that conversion of forests to pasture would decrease soil carbon stocks, as the temperature of the soil increases markedly when exposed to the sun in a pasture, a factor known to shift the equilibrium between formation and oxidation of organic carbon to a lower plateau (Cunningham, 1963; Greenland and Nye, 1959; see also Sánchez, 1976; pp. 164–172). In addition, annual burning in tropical savannas is known to reduce raw organic matter additions to the soil (Sánchez, 1976; p. 170), and the same effect can be expected to apply to planted cattle pastures.

Conversion of forest to pasture reduces the water storage capacity of the soil (Chauvel et al., 1991) and confines the distribution of inputs of carbon from the roots to the surface layers (Nepstad et al., 1994). Both of these factors, together with high rates of decomposition (oxidation) of soil carbon (especially near the surface), could result in a decline in the soil carbon stock (Post et al., 1995).

There may be additional effects on the soil mesofauna. Termites (Isoptera), leaf-cutter ants (*Atta* spp.), cicadas (Cicadidae) and springtails (Collembola) are important in incorporating litter and half-burned plant remains into the soil (at greater depth). These populations can be expected to be affected both by the transformation of forest to pasture and by the periodic impacts of fire. Termite populations take approximately six years to increase their populations in cleared areas in response to the increased availability of dead wood; termite populations then decline to low levels by the tenth year after deforestation as the dead wood in the original forest remains disappears through decomposition and burning (Martius et al., 1996). Leaf-cutter ant populations increase in pasture, especially degraded pasture with invading woody plants (the leaves of which the ants prefer over those of grasses); these ants carry substantial quantities of plant matter to their nests in the soil at depths sometimes exceeding 5 m (Nepstad et al., 1995a).

The sources of soil carbon can be identified using isotopic fractionation techniques. Forest trees fix carbon by the C_3 photosynthetic pathway. The carbon fixed by pasture, which uses the C_4 pathway, has a higher fraction of the ^{13}C isotope, as compared to ^{12}C. The isotopic composition, expressed as $\delta^{13}C$ (a measure of change in the ratio of $^{13}C/^{12}C$ relative to an international standard), allows the rates of supply and degradation to be estimated in pasture, even if no difference is apparent between forest and pasture carbon stocks based on carbon concentration and soil bulk density (e.g., Desjardins et al., 1994; Trumbore et al., 1995; Moraes et al., 1995; Neill et al., 1996). Changes in ^{14}C accumulation from nuclear weapons testing also provide a means of obtaining statistically significant indicators of carbon turnover even when traditional carbon inventory measurements produce results too variable to detect differences with statistical significance (Trumbore et al., 1995).

When conversion of forest to pasture results in a new equilibrium with lower soil carbon stocks, the losses occur in small amounts over time (Davidson et al., 1993; Barbosa, 1994). These losses can occur due to incomplete substitution of the organic carbon source that had been provided by the forest. Although pasture provides a strong source of carbon to the soil, these inputs are not enough to compensate for losses of initial forest soil carbon (Desjardins et al., 1994; p. 113). This leads to a net loss of carbon from the ecosystem due to humification of the pasture roots (stock) not exceeding the emission from oxidation during and/or after burning. Repeated burning, change from a high- to a low-biomass system, trampling by hooves of the cattle, and exposure to rain and direct sunlight can cause small local effects that, under the most common management regime in Brazilian Amazonia, accumulate over the long term to transform the landscape from productive pasture into a degraded system and a net source of carbon to the atmosphere.

3.3. Effects of pasture management regimes

Over the long term, global calculations have shown losses of carbon stocks when forests are converted to other land uses. Detwiler (1986) and Detwiler and Hall (1988) suggested that the stock of soil carbon in a 40-cm profile in the tropics could be reduced by 20% by conversion of primary forest to pasture, and used this value in modeling the effects of deforestation on the global carbon budget. Houghton et al. (1983, 1987) assumed that converting tropical forest to pasture would release 25% of the soil carbon stock to a depth of 1 m. These are reviews derived from a variety of studies, and a wide range of values has been obtained by different authors. However, a study of the 0–30 cm layer in soils in Rondônia by Moraes et al. (1996) indicated that, over the long term, there is an increase in the stock of carbon in well-managed pasture as compared to nearby primary forest. In an extreme case, carbon in the top 20 cm of soil under very well-managed pasture at an agricultural station near Manaus returned to approximately the same level present under original forest eight years after clearing (90 t C/ha under forest vs. 96 t C/ha under pasture, without correction for soil compaction), following a decline by 21.4% reached two years after clearing (Cerri et al., 1991; Choné et al., 1991). Teixeira (1987; p. 59) found a similar pattern for carbon content in the same well-managed pasture near Manaus. Under more commonly occurring conditions, the opposite result was found by Serrão and Falesi (1977) and by Eden et al. (1991); these authors found, respectively, a decline of about 50% in carbon concentration in pasture soil with 11 years of use in Fazenda Suiá Missu, Mato Grosso, and a 15% decline in concentration in pasture with 12 years of use near Maracá Island, Roraima. Fearnside (1978; p. 254) found mixed results: a gain in carbon concentration upon forest conversion that maintained its new level through 1–2 years of pasture, and carbon loss with conversion that only partially regained forest soil levels after 6–16 years of use. Well-managed pastures are not typical of Amazonia, and are unlikely to reflect regional means for the increase or decrease of carbon stocks caused by land-use change (Trumbore et al., 1995).

The quality of management of tropical pastures is critical to the conclusions drawn about whether the soils under this land use represent a source or a sink of atmospheric carbon. In well-managed pastures in formerly forested areas, the root system of the pasture grass can redistribute carbon to deeper layers (Nepstad et al., 1991), where it is less susceptible to decomposition. This could be used as a strategy for carbon sequestration in soils in the areas that have already been converted to pasture (Batjes and Sombroek, 1997). The same is true of well-managed pastures planted in former natural savanna areas (Fisher et al., 1994). However, this conclusion is wholly dependent on the pastures being very well managed, such as those on the agricultural experiment station in Colombia where the work on former savannas was carried out (see Nepstad et al., 1995b). Unfortunately, the vast majority of cattle pastures in Brazilian Amazonia is poorly managed: grass productivity declines within a decade, regardless of whether the pastures are on formerly forested or on former savanna land. Under typical (i.e., minimal input) management conditions at Ouro Preto do Oeste, Rondônia, measurements of grass productivity indicate that a 12-year-old pasture produces only about half as much above-ground dry weight of pasture grass annually as compared to a three-year-old pasture (Fearnside, 1989; p. 50).

4. Effects masking soil carbon changes

4.1. Fine-scale spatial variability

Fine-scale spatial variability in soil properties, including carbon concentration, clay content and bulk density, could mean that the results of many studies that substitute space for time to compare soil samples taken at about the same time from a series (chronosequence) of pastures of different ages could produce the varied results that they do merely because of soil quality differences that have nothing to do with the effect of time under pasture. The presence of trees in the forests with which pasture soils are compared results in spatial variability in both carbon concentration and bulk density being significant over shorter distances in forest samples than pasture samples (Veldkamp and Weitz, 1994). Studies vary in the methodology used to collect the samples (ranging from point samples at a single pit or core to composite samples representing wider areas), making them correspondingly more or less subject to the effect of fine-scale spatial variation.

4.2. Correction for soil compaction

Corrections applied (or not applied) by different authors to soil carbon measurements can make a substantial difference in the conclusions reached. Correction for the effect of soil compaction is needed for valid results. If one compares, for example, the top 20 cm of soil under pasture with the top 20 cm of soil under forest, it is possible to conclude that carbon has increased when, in reality, it has declined. This is because the soil in the top 20 cm under pasture has been compacted from a thicker layer under the forest, and the comparison of the stock of carbon (the concentration of carbon multiplied by the bulk density and the volume of soil) must be done between equivalent weights of soil in the two land uses. Correction for compaction has been made in a growing number of soil carbon studies since it was first introduced in 1985 (Fearnside, 1985), but studies without the correction (or without a clear explanation of what corrections have been applied) continue to contribute to confusion over the effect of deforestation on emissions from the soil. Measurements of soil bulk density are much less frequently made than are measurements of carbon concentration, perhaps because of the considerable additional work that density measurement adds to the demands of field sampling. Bulk density results are also subject to high variability, stemming from slight differences in methodology, for example, among different investigators.

4.3. Correction for clay content

Moraes et al. (1996) introduced a correction for soil clay content, based on the finding of Feller et al. (1991) that soil organic matter is closely related to the clay content of the soil, such that the variation in clay content over fine spatial scales could mask the effect of land-use changes. This is consistent with what is known about the reaction of organic matter with iron and aluminum oxides. These oxides react with organic radicals to form complexes that remain relatively resistant to mineralization, possibly because the oxides may physically block access of microorganisms to the organic particles, thereby inhibiting their decomposition (Sánchez, 1976; p. 164).

Applying a correction for clay content is not easily done without masking the land-use change effects that need to be estimated for use in greenhouse gas emission calculations. The clay content correction by Moraes et al. (1996; p. 68) assumes that there is no change in the clay content of the soil resulting from land-use change. Moraes et al. (1996) calculate changes over 20 years, and Cerri et al. (1996) use these data to calculate changes over 35 years; both periods are sufficient for appreciable granulometric changes to occur. By correcting the measured carbon values in the pasture to reflect what the carbon values would be if clay were the same as that under undisturbed forest, one hides any change in soil carbon that may have resulted from deforestation's effect on soil clay content.

A decrease in clay content is one of the changes known to result from conversion of forest to pasture. This occurs by two mechanisms. First, soil erosion preferentially removes the smaller soil particles (i.e., clay), leaving the coarser particles (such as sand) behind (Lal, 1977; p. 54). Clay content is the most important determinant of erodibility (the susceptibility of a soil to erosion), an effect only partially offset by a negative relationship between organic matter and erodibility (Mitchell and Bubenzer, 1980; pp. 32–33). Soil erosion on clay soils on the Transamazon Highway has been found to be more rapid than on sandy soils with the same slope and land use (Fearnside, 1980b, 1986b), and significant rates of soil erosion have been measured under cattle pasture (Barbosa, 1991; Fearnside, 1989). The second mechanism is the transport of clay particles downward in the soil profile, leaving the coarser particles at the surface (Scott, 1975, 1978). Removal of clay to the deeper layers would lead to a decline in the total carbon inventory, even though the transported clay remains in the soil column. This is because the deeper soil layers have lower rates of supply of organic carbon due to the lower density of roots in these soil layers. Relationships between clay content and soil carbon are summarized in Fig. 2.

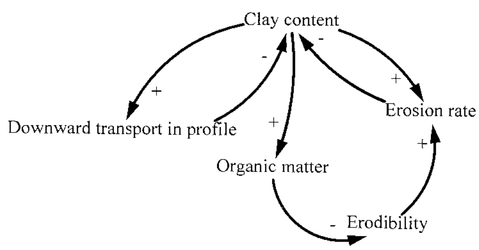

Fig. 2. Relationships between clay content changes and soil carbon. The sign by each arrow represents the direction of change in the quantity at the head of the arrow given an *increase* in the quantity at the tail of the arrow.

The portion of the soil most affected by alteration of clay content is that closest to the surface, which is precisely where organic matter is most highly concentrated and dynamic. All of these clay content changes would tend to decrease the carbon stock under pasture soils, but this decrease would be hidden when a correction is applied for clay content between the forest and the pasture or between the pastures of different ages (as was done by Moraes et al., 1996). In the study by Moraes et al. (1996), the clay correction did not alter the direction of the change in carbon stock, but it did increase the estimated rise in carbon stock from 13.5% $\pm$ 8.5 to 18% $\pm$ 1.

It is possible that the values found for stocks and emissions of soil carbon depend more on the soil type (including clay content) than they do on the management system that replaces forest (e.g., Allen, 1985). Clayey soils tend to have more carbon per unit area (Cerri et al., 1992). This could lead to greater releases of carbon, as the amount of carbon lost is positively related to the amount initially present (Allen, 1985; Fearnside, 1986b, p. 194; Mann, 1986).

4.4. Humified versus non-humified material

Nye and Greenland (1960; p. 70) pointed out that much confusion over the effects of burning on soil organic matter has been caused by failure to distinguish between humified and non-humified material. This confusion continues today both with regard to the effect of burning and with regard to the overall effect of forest-to-pasture conversion. At the time of burning, almost all litter present is combusted, but this is not humus or soil organic matter (which is little changed). The classification of roots as included or not in soil organic matter can have a dramatic effect on conclusions. In a recently implanted cattle pasture, the dead (but unhumified) coarse roots remaining from the original forest would appear as a tremendous jump in soil organic matter if this stock were considered to be part of the soil carbon pool. By the same token, the greater number of fine roots in the surface layers of pasture soil also would appear to be a huge increase in soil carbon if these roots were considered to be part of the soil rather than part of the biomass. Since soil analyses frequently define whatever passes through a 2-mm sieve as part of the soil, these fine roots can be an important component of reported increases in soil carbon as a result of conversion to pasture.

4.5. Seasonal cycles

A seasonal cycle of soil carbon stocks complicates assessment of the effects of conversion of forest to pasture. A persistent problem in interpreting published results on soil carbon changes with burning is that authors rarely report the dates of the pre- and post-burn samples, making it impossible to tell if the increases or decreases reported are due to the burning or to a seasonal cycle of soil carbon stocks.

The content and composition of the soil organic matter can be influenced as a result of seasonal changes (Andreux et al., 1990), resulting in different phases of accumulation and loss of carbon in the soil under the system. The peak carbon stock in tropical soils that the few available data suggest occurs during the rainy season appears likely to be linked to the increase in volume of fine roots during this period in pastures. Measurements taken in an Oxisol near Manaus indicate that this component almost doubles in weight during the rainy season, as compared to the dry season, in a young *Brachiaria humidicola* pasture, especially in the 0–5 cm layer (Luizão et al., 1992). A similar result was found by Veldkamp (1993; p. 40) in fertile soils (Andisols and Inceptisols) in Costa Rica in *B. dictyneura* pasture. The seasonal cycle in soil carbon stocks results from the seasonality of carbon inputs, which are only partially offset by increases during the rainy season in microbial biomass (Luizão et al., 1992), and consequently also in soil respiration (Feigl et al., 1995). A seasonal cycle in soil carbon stocks is suggested by changes over time observed under a 7-yr-old pasture in Apiaú, Roraima (Barbosa, 1994). However, data for a young pasture observed over the first 5 months after initial forest burning at Fazenda Nova Vida, Rondônia fail to show a consistent seasonal pattern (P.M.L.A. Graça, unpublished data). The principal seasonal effects are diagrammed in Fig. 3.

Burning is a seasonal event, occurring only in the dry season; burning lowers microbial and fine root biomass and removes the litter that would otherwise decay and provide a source of carbon input to the soil. The seasonal cycle of rainfall produces parallel cycles of microbial biomass, fine roots and litter

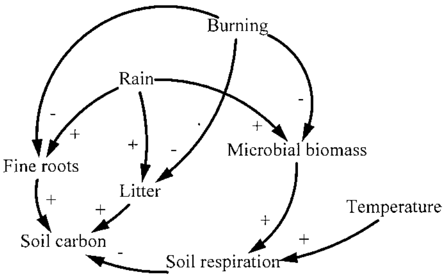

Fig. 3. Principal seasonal effects on soil carbon stocks.

production. Any seasonal changes in temperature will be reflected in the rate of soil respiration.

5. A best current estimate for Amazonia

5.1. Standardization for compaction and sample depth

Global calculations of greenhouse gas fluxes must, of necessity, include estimates for the net carbon release or uptake from soils, even if the value used is zero. The existence of uncertainty is not a valid rationale for failing to arrive at a best current estimate. Where, then, do we stand with respect to a best current estimate for the impact on soil carbon of the massive conversion of forest to pasture now underway in Brazilian Amazonia?

The available data require standardization for the treatment of bulk density effects and for the depth of sampling. Most important, the representativeness of the pasture management systems at the study sites must be taken into account, and an appropriate stratification by pasture management system must be applied to arrive at a best estimate for the region as a whole.

Studies are also varied in the age of pasture for which the effect is estimated (and in the validity of the assumption that a new equilibrium has been established by the year in question). The reliability of the methods used also varies among studies. Corrections are not attempted for these factors, but studies judged to be restricted to short-term effects, to reflect land-use effects other than pasture, or to be unreliable for any reason, were eliminated from consideration.

Soil compaction, or bulk density increase, depends on the depth of the soil examined and the soil type. The current estimate standardizes depths for soil carbon in 0–20 cm, 20–100 cm and 100–800 cm layers. Soil compaction information is not standardized by layer, but rather only results of studies done on the 0–20 cm layer are used. Compaction is (optimistically) assumed to be zero below a depth of 20 cm.

Bulk density changes from five studies are used to calculate a mean increase of 4% for the 0–20 cm layer (Table 1). Other measurements are also presented in Table 1, including measurements for other soil layers and measurements outside of Brazil (studies from Costa Rica's volcanic soils were not used in the calculations). One Brazilian study of the 0–20 cm layer (Hecht, 1981; p. 95) was not used in the calculations because the value reported for forest soil density (0.56 g/cm^3) seems improbably low. The additional studies listed all found higher values for bulk density increase than did those used in the calculations, giving added assurance that the values used do not exaggerate this effect.

A number of studies of soil carbon changes have used soil depth ranges other than those adopted for the current estimate. Approximations for the stocks in the layers considered here can be made based on the relationships among stocks in different layers found by other workers (Table 2). The firmest relation is that between 0–20 cm and 0–100 cm derived by Moraes et al. (1995) based on over 1000 profiles collected throughout the region by the RADAMBRASIL project (Brazil, Projeto RADAMBRASIL, 1973–1983). Other conversions are based on many fewer studies, but variation in the proportional relationship among soil layers is small (Table 2).

Studies of soil carbon stocks at different locations in Brazilian Amazonia and implied long-term changes from conversion to pasture are given in Table 3. These are separated into categories for 'sites with typical management' (five studies) and 'sites with ideal management' (five studies). 'Typical management' is without inputs or other measures designed to maintain soil fertility, while 'ideal management' can have a variety of practices that are considered 'improved'. Two of the studies in the first category and three in the second are for soil layers other than

Table 1
Soil bulk density changes from conversion of forest to pasture

Location	Bulk density (g/cm^3)		Bulk density increase (%)	Pasture age (yr)	Sample depth (cm)	Soil type	Source
	Forest	Pasture					
Measurements used in calculations							
Apiaú, Roraima	1.10	1.21	10	7	20	Ultisol	Barbosa, 1994: 62
Capitao Poco, Para	1.33	1.38	4	10	20	Ultisol	Desjardins et al., 1994: 105
Nova Vida, Rondonia	1.44	1.41	−2	20	20	Ultisol	Moraes et al., 1996: 69
Nova Vida, Rondonia[a]	1.35	1.40	4	81	20	Ultisol	Neill et al., 1996: 115
Nova Vida, Rondonia[a]	1.42	1.47	4	81	20	Ultisol	Moraes et al., 1996: 69
Mean	1.31	1.36	4	30			
Standard deviation	0.15	0.10	6	35			
N[a]	4	4	4	4			
Other measurements							
Paragominas, Para[b]	0.56	1.15	105	14	20 (?)	Ultisol (?)	Hecht, 1981: 95
Ilha de Maraca, Roraima[c]	1.05	1.58	50	12	10	Ultisol	Eden et al., 1991: 287
La Selva, Costa Rica[c,d]	0.655	0.785	20	10–36	10	Ultisol	Reiners et al., 1994: 369–370
La Selva, Costa Rica[c,d]	0.62	0.71	14	35	5	Inceptisol	Keller et al., 1993: 245
Guamico, Costa Rica[d]	0.67	0.79	18	25	5	Inceptisol	Keller et al., 1993: 245

[a] Measurements for same pasture combined in mean.
[b] Not used in calculations because of improbably low density value for forest soil.
[d] Not used in calculations because sample depth is not 20 cm.
[d] Not used in calculations because soils are significantly different from those in Amazonia.

Table 2
Vertical distribution of carbon stock in Amazonian forest soils

Layer A	Layer B	Location	State	C stock (t C/ha)		% in layer A	Soil type	Source
				Layer A	Layer B			
0–10 cm	0–20 cm	Nova Vida	Rondonia	16.2	25.1	64.5	Ultisol	Neill et al., 1996: 115
		Capitao Poco	Para	19.2	31.4	61.0	Ultisol	Desjardins et al., 1994: 105
		Manaus	Amazonas	26.1	45.1	57.9	Oxisol	Cerri and Volkoff, 1987: 37
		Mean		20.5	33.9	60.5		
0–10 cm	0–30 cm	Maraba	Para	27.6	56.3	49.0	Ultisol(?)	Kauffman et al., 1995: 405
		Manaus	Amazonas	26.1	59.6	43.8	Oxisol	Cerri and Volkoff, 1987: 37
		Mean		26.9	58.0	46.3		
0–10 cm	0–100 cm	Paragominas	Para	26	102	25.5	Ultisol	Trumbore et al., 1995: 519
		Manaus	Amazonas	26.1	124.1	21.0	Oxisol	Cerri and Volkoff, 1987: 37
		Mean		26.1	113.1	23.0		
0–20 cm	0–30 cm	Nova Vida	Rondonia	25.1	32.4	77.4	Ultisol	Neill et al., 1996: 115
		Manaus	Amazonas	45.1	59.6	75.7	Oxisol	Cerri and Volkoff, 1987: 37
		Mean		35.1	46.0	76.3		
0–20 cm	0–100 cm	Legal Amazon	All states	42	94	45	All types	Moraes et al., 1995: 245
0–100 cm	0–500 cm	Manaus	Amazonas	124.1	229.6	54.1	Oxisol	Cerri and Volkoff, 1987: 37
0–100 cm	0–800 cm	Paragominas	Para	102	257	39.7	Ultisol	Trumbore et al., 1995: 519

0–20 cm. These are presented both with the original measurements and as values estimated from them for the 0–20 cm soil layer for the purposes of calculation. For consistency, the endpoint of each chronosequence was used to identify the carbon stock under pasture.

Three of the studies that did not make a correction for compaction (Desjardins et al., 1994; Eden et al., 1991; Neill et al., 1996) included bulk density measurements that allow the correction to be performed here (Table 3). In another study (Cerri et al., 1991), the average value from Table 1 (4% bulk density increase) was used to make the correction.

The 4% average compaction value from Table 1 is also used in re-interpreting the two measurements by Falesi (1976). The values of Falesi (1976) indicate the greatest carbon losses. The data from this study had been used in a previous estimate of soil carbon loss from conversion of Amazonian forest to pasture (Fearnside, 1985, 1997b; p. 347), but the compaction correction applied in that estimate may have been exaggerated because the forest soil density used (from Hecht, 1981; p. 95) has an unusually low value. Because the 0–20 cm layer under forest is believed to have greater mass than calculated previously, the results here indicate a higher average carbon release than the average computed earlier (3.92 t C/ha release from the 0–20 cm layer).

The carbon stocks under forest in the studies listed in Table 3 differ substantially from the 42 t C/ha average stock for the Legal Amazon region calculated by Moraes et al. (1995) for the 0–20 cm layer of soil. The five sites with 'typical management' average only 27.5 t C/ha, while the five studies with 'ideal management' are also all lower than 42 t C/ha with the exception of one high value (90 t C/ha) found by Cerri et al. (1991; p. 254), which raised the mean to 57.5 t C/ha. This difference is handled by using the percentage changes in soil carbon in the 0–20 cm layer indicated by the studies in each management category, and applying these percentages to the regional average soil carbon stock as estimated by Moraes et al. (1995).

5.2. Disaggregation by management regime

The suite of existing studies of soil carbon under pasture is highly unrepresentative of pastures in Brazilian Amazonia as a whole. A disproportionately large number of studies has been done in optimally managed pastures either in agricultural experiment stations, or in the few ranches that make significant investments in inputs and pasture management (this minority of ranches is more likely to encourage research groups to study their pastures than is the majority that applies minimal management).

Table 3
Soil carbon stocks at different locations in Brazilian Amazonia and implied long-term changes from conversion of forest to pasture

Location	State	Pasture age (yr)	Carbon stock (t C/ha)			Presumed C stock change (t C/ha)	Change relative to pre-conversion stock (%) (corrected for compaction)	Soil type	Sampling depth (cm)	Correction for compaction	Source
			Pre-conversion	Post-conversion (as published)	Post-conversion (corrected for compaction)						
Sites with typical management											
Values published or calculated for 0–20 cm depth or compaction-corrected equivalent.											
Capitao Poco	Para	10	31.4	29.7	28.6	−2.8	−8.9	Ultisol	0–20	No[a]	Desjardins et al., 1994: 111
Paragominas	Para	10	15.1	12.0	12.0	−3.1	−20.6	Ultisol	0–20	[b]	Falesi, 1976: 42[c]
Suia Missu	Mato Grosso	11	25.1	12.9	12.9	−12.2	−48.7	Oxisol	0–20	[b]	Falesi, 1976: 31[c]
Ilha de Maraca	Roraima	12	25.7	22.4	22.4	−3.3	−12.8	Ultisol	0–10	No[a]	Eden et al., 1991[d]
Paragominas	Para	23	40.3	37.2	37.2	−3.1	−7.7	Ultisol	0–100	Yes	Trumbore et al., 1995: 525 [e,f,g,h]
Mean			27.5	22.8	22.6	−4.9	−17.8				
Original values for studies published for depths other than 0–20 cm:											
Ilha de Maraca	Roraima	12	16.6	14.3		−2.3	−13.9	Ultisol	0–10	No[a]	Eden et al., 1991
Paragominas	Para	23	102	86		−16.0	−15.7	Ultisol	0–100	Yes	Trumbore et al., 1995: 525[e]
Sites with ideal management											
Values published or calculated for 0–20 cm depth or compaction-corrected equivalent											
Nova Vida	Rondonia	81	25.1	41.2	39.7	14.6	58.4	Ultisol	0–20	No[a]	Neill et al., 1996: 115
Manaus	Amazonas	8	90.0	96.0	92.6	2.6	2.9	Oxisol	0–20	No[b]	Cerri et al., 1991: 254
Nova Vida	Rondonia	20	26	26.4	26.4	0.9	3.4	Ultisol	0–30	Yes	Moraes et al., 1996: 71
Nova Vida	Rondonia	20	35	41.1	41.1	6.1	17.4	Ultisol	0–30	Yes	Moraes et al., 1996: 71
Paragominas	Para	23/5[i]	40.3	52.1	52.1	11.8	29.4	Oxisol	0–100	Yes	Trumbore et al., 1995: 525[e,f,h,j]
Mean			57.5	68.6	66.2	8.6	15.0				
Original values for studies published for depths other than 0–20 cm											
Nova Vida	Rondonia	20	33	34		1.0	3.0	Ultisol	0–30	Yes	Moraes et al., 1996: 71[d]
Nova Vida	Rondonia	20	35	42		7.0	20.0	Ultisol	0–30	Yes	Moraes et al., 1996: 71[d]
Paragominas	Para	23/5[i]	102	120		18.0	17.6	Oxisol	0–100	Yes	Trumbore et al., 1995: 525[e]

[a] Compaction correction calculated from data in original publication.
[b] Compaction correction calculated from average bulk density increase of 4% (Table 1).
[c] Revised from Fearnside (1985, 1997b): 347 (see text).
[d] Carbon stock estimated for 0–20 cm based on data from Neill et al. (1996: 115. which indicate ratio of forest C stock in 0–10 cm to stock in 0–20 cm of 0.65, and ratio for 0–30 cm to 0–20 cm of 1.29. Ratio of the compaction–corrected change in carbon stocks to 81 yr in the 0–10 cm layer relative to the change in the 0–20 cm layer is 0.70, while the ratio for change in the 0–30 cm layer relative to the 0–20 cm layer is 1.15.
[e] Modeled result based on rates estimated from isotopic techniques.
[f] Forest C stock for 0–20 cm calculated from 0–10 cm stock, assuming same fraction (0.65) as found by Neill et al. (1996) in Rondonia.
[g] The 0–100 cm layer C flux is apportioned between 0–10 cm and 10–100 cm based on the fraction (0.273) of the 0–100 cm C inputs that occur in the 0–10 cm layer in typically managed pasture at the same site (Trumbore et al., 1995: 521).
[h] The 0–20 cm layer C flux is derived from the 0–10 cm flux based on ratio of 0.70 for flux calculated from data for ideally managed pasture in Rondonia reported by Neill et al. (1996).
[i] Time since clearing = 23 yr; time under ideally managed pasture = 5 yr. Equilibrium had not yet been reached; Trumbore et al. (1995: 526) estimate that this will only occur after 50 years under ideal management, but that only small changes would occur after year 10.
[j] The 0–100 cm layer C flux is apportioned between 0–10 cm and 10–100 cm based on the fraction (0.324) of the 0–100 cm C inputs that occur in the 0–10 cm layer in ideally managed pasture at the same site (Trumbore et al., 1995: 521).

Table 4
Weighted average of regional changes in soil carbon

Management type	Percent of pasture area	Carbon stock change in 0–20 cm layer			Carbon stock change in 20–100 cm layer		Carbon stock change in 100–800 cm layer		Carbon stock change in 0–800 cm layer	
		Observed change at sampled locations (t C/ha)[a]	Percentage change in forest C stock at sampled locations (%)[a]	Calculated change relative to regional average stock (t C/ha)[b]	Observed change at sampled locations (t C/ha)	% of forest C stock[c]	Observed change at sampled locations (t C/ha)[d]	% of forest C stock[e]	Total change (calculated changed for 0–20 cm + observed for 20–800 cm) (t C/ha)	% of forest C stock
Typical (minimal) management	95	−4.9	−17.8	−7.5	−5.6[f]	−10.8	−0.6	−0.4	−13.7	−5.8
Ideal management	5	8.6	15.0	6.3	19.3[g]	37.1	−5.3	−3.7	20.3	8.6
Weighted average	100	−4.2	−16.2	−6.8	−4.3	−8.4	−0.8	−0.6	−12.0	−5.1

[a] Table 3.
[b] Relative to 42 t C/ha regional average stock in 0–20 cm layer (Moraes et al., 1995).
[c] Relative to 52 t C/ha regional average stock in 20–100 cm layer (Moraes et al., 1995).
[d] Based on difference between change over 23 yr at Paragominas, Para in 0–800 cm layer as compared to 0–100 cm layer in the same profile; typical management site is 23 yr old; ideal management site was 18 yr under typical management followed by 5 yr under ideal management (Trumbore et al., 1995: 526).
[e] Relative to 142.8 t C/ha stock in 100–800 cm layer, based on 94 t C/ha 0–100 cm regional average C stock (Moraes et al., 1995) and ratio of 0–100 cm to 100–800 cm stocks from Trumbore et al. (1995: 526) (See Table 2).
[f] Based on difference between 8.7 t C loss in 0–100 cm layer over a 23 year period at Paragominas, Para (Trumbore et al., 1995: 526) and 3.1 t C calculated loss for 0–20 cm layer at the same site (Table 3: note h).
[g] Based on difference between 25.6 t C gain relative to forest cut 23 yr previously in 0–100 cm layer over a 5-yr period at Paragominas, Para (Trumbore et al., 1995: 526) and 2.0 t C calculated average gain for 0–20 cm layer.

No estimate exists of the percentage of pasture land in Brazilian Amazonia that is managed under the ideal regimes in use at the study sites of a number of the available measurements of soil carbon changes under pasture. While the area under such management would have been close to zero (certainly well under 1% of the total pasture area) in the 1970s and 1980s, the financial returns to this kind of ranching improved in the 1990s (Mattos and Uhl, 1994; Arima and Uhl, in press). However, the overwhelming preponderance of pasture with minimal investment in management is still evident. Here, the assumption will be made that 5% of the pastures in the region were under improved management regimes in 1990. Based on this assumption, weighted averages of regional changes in soil carbon are computed (Table 4). Over the full 8-m soil profile, an average of 12.0 t C/ha is released, over half of this coming from the top 20 cm of soil.

5.3. Effect of pasture on the equilibrium landscape

Following deforestation, each point in the landscape will pass through periods under a variety of land uses, such as agriculture, pasture and secondary forest. The deforested area as a whole will eventually approach an 'equilibrium landscape,' which, assuming that land-use behavior patterns remain unchanged, would consist of 4.0% farmland, 43.8% productive (actively used) pasture, 5.2% degraded pasture, 2.0% secondary forest from agriculture and 44.9% secondary forest from pasture (Fearnside, 1996).

The impact of converting forest to pasture is not restricted to the 43.8% of the deforested landscape that would be expected to remain under productive pasture under equilibrium conditions. A large part of the remainder of this landscape would have passed through periods of use as pasture, and this would be reflected in reduced carbon stocks relative to forest. A rough calculation of the carbon stocks in the equilibrium landscape can be made (Table 5). Secondary forests are assumed to recover the soil carbon stocks characteristic of mature forests in a linear fashion over a period of 15 years, the same assumption made for the top 1 m of soil by Houghton et al. (1983; pp. 239–240), except that full recovery is assumed instead of only 75% of the original carbon stock.

Soil carbon release under agriculture is assumed to be 35% of the pre-clearing carbon pool in the 0–30 cm layer (the midpoint of the 20–50% range identified by Sombroek et al., 1993). This is more conservative than the assumption of Houghton et al. (1983) that 35% is released in the top meter of soil, a depth range that has a carbon stock approximately 40% greater than the top 30 cm. The present calculation assumes conservatively that carbon releases from agricultural soils below 30 cm depth are the same as those under typically managed pasture (Table 5, note b).

Table 5
Carbon stocks in soils under land uses in the equilibrium landscape

Soil layer (cm)	C stock in soil (t C/ha)[a]						
	Forest	Farmland[b]	Productive pasture	Degraded pasture[c]	Secondary forest from agriculture[d,e]	Secondary forest from pasture[d,f]	Equilibrium landscape
0–20	42.0	27.3	35.2	34.5	36.8	39.1	36.6
20–100	52.0	42.5	47.7	46.4	52.0	52.0	49.4
100–800	142.8	142.2	142.0	142.2	142.7	142.6	142.3
Total (0–800)	236.8	212.1	224.9	223.2	231.6	233.7	228.4

[a]Stock in layer compacted from specified depth of forest soil; no compaction is assumed to occur below 20 cm depth.
[b]Assumes 35% reduction in the 0–30 cm layer, this being the midpoint of the 20–50% range reported by Sombroek et al. (1993; 421). Assumes same reduction below 30 cm depth as under pasture with typical management (Table 4); relation between C stocks in 0–20 cm and 0–30 cm layers from Table 2; carbon stock and changes assumed to be evenly distributed within each layer.
[c]Assumed same as pasture under 'typical management'.
[d]Secondary forests are assumed to recover soil carbon stocks characteristic of mature forests in a linear fashion over a period of 15 yr.
[e]Average age of secondary forest derived from agriculture is 3.2 yr (Fearnside, 1996).
[f]Average age of secondary forest derived from pasture is 3.9 yr (Fearnside, 1996).

Table 6
Net committed emissions of soil carbon from deforestation in Brazilian Amazonia in 1990

Land use	Proportion of equilibrium landscape	Area in equilibrium landscape (10^3 ha)	Top 20 cm of soil[a]		Top 1 m of soil[a]		Top 8 m of soil[a]	
			Carbon change relative to forest (t C/ha)	Net committed emission in 1990 (10^6 t C)	Carbon change relative to forest (t C/ha)	Net committed emission in 1990 (10^6 t C)	Carbon change relative to forest (t C/ha)	Net committed emission in 1990 (10^6 t C)
Farmland[b]	0.040	56	−14.7	−0.8	−24.2	−1.3	−24.8	−1.4
Productive pasture	0.438	606	−6.8	−4.1	−11.1	−6.7	−12.0	−7.3
Degraded pasture	0.052	72	−7.5	−0.5	−13.1	−0.9	−13.7	−1.0
Secondary forest from agriculture	0.020	28	−5.2	−0.1	−5.2	−0.1	−5.3	−0.1
Secondary forest from pasture	0.449	620	−2.9	−1.8	−2.9	−1.8	−3.1	−1.9
Equilibrium landscape	1.000	1382	−5.4	−7.4	−7.9	−11.10	−8.5	−11.7

[a]Corrected for compaction in top 20 cm.
[b]See Table 5, note b.

The calculated emissions (Table 6) indicate an area-weighted average release of 8.5 t C/ha from the top 8 m of soil in the deforested landscape. This corresponds to a net committed emission of 11.7×10^6 t C from the 1.38×10^6 ha of deforestation activity in 1990.

Low values for carbon loss are indicated for the 100–800 cm layer. Only 3% of the calculated emissions come from below 1 m depth. However, complacency about the security of the deep soil pool is not recommended: the large size of the carbon pool in this soil layer means that even small changes (near detection limits) could have substantial impact on carbon emissions (see Trumbore et al., 1995; Nepstad et al., 1994). The large pools of carbon in the deep soil layers, very little of which is assumed to be emitted in the present calculation (only 0.4% in the 100–800 cm layer: Table 5), leave ample room for substantially larger emissions. The average turnover time for the entire soil organic matter stock in the 1–8 m depth range, weighted for active and slow pools, has been calculated to be < 25 years (Trumbore et al., 1995; p. 527). The slower dynamics of carbon in these deep layers, as compared to the surface layers, means that the pasture area in Amazonia will continue to release deep soil carbon for many years until a new equilibrium is reached; these additional emissions are not reflected in the results computed here. The values used for carbon changes in the 1–8 m range are probably conservative because the assumption that equilibrium has been attained by the time of the study is least justified for this soil layer.

The values for net committed emissions given here (Table 6) are not exactly what this term implies. The term 'net committed emissions' is normally used to refer to the difference between the carbon stocks under original forest and those under the equilibrium landscape after a long (theoretically infinite) time has elapsed following deforestation (see Fearnside, 1997b). If this definition were strictly applied, the large carbon pool in the deep soil might release substantially more carbon than the emission from this layer considered in the present calculation. Because the deep soil would only approach equilibrium very slowly, the values considered here (which only reflect net emissions from the deep soil in the first 23 years after clearing) may be more relevant to planning time horizons than would true 'net committed emissions'.

The 11.7×10^6 t C emission from soil carbon for the equilibrium landscape from clearing in 1990 represents an emission equivalent to approximately 20% of Brazil's emissions from fossil fuels. The net committed emission of carbon in 1990 from Amazonian deforestation sources other than soil totaled 257.0×10^6 t C, without considering trace-gas effects and excluding clearing of savannas such as *cerrado* (Fearnside, 1997b; p. 351). Considering (conservatively) the soil emission computed here as a net committed emission, this brings the total from Amazonian deforestation that year to 268.7×10^6 t C, of which soils represent 4.4%. Carbon from clearing of *cerrado* biomass would add approximately 8.6×10^6 t C (Fearnside, 1997b: 351). Ignoring any carbon changes in *cerrado* soils converted to pasture and agriculture (primarily soybeans), the 1990 total emission from land-use change in Brazil's 5×10^6 km^2 Legal Amazon region was 277.3×10^6 t C. The potential magnitude of soil carbon releases is significant: if all of the 400×10^6 ha originally forested area in Brazil's 500×10^6 ha Legal Amazon region were converted to the equilibrium landscape, 4.7 Gt of carbon would be released from the soil to a depth of 8 m.

5.4. Robustness of the estimate

The percentage of Amazonian pastures, where management practices are similar to those where the studies in the 'ideal management' category were carried out, is critical to the overall effect of the region's pasture soils on carbon emissions (Fig. 4). If none of the pastures in the region were in the 'ideal' category, average carbon release from an 8–m soil profile would be 13.7 t C/ha, or 13.0×10^6 t C from the equilibrium landscape from clearing in 1990. Were all of the pastures kept under ideal management, the soil C stock under these pastures would increase by 20.3 t C/ha, or 13.0×10^6 t C for the equilibrium landscape resulting from deforestation in 1990. Considering 5% as a reasonable value for the percentage of the pastures under ideal management in 1990, average release of soil carbon would be 12.0 t C/ha of pasture, or 11.7×10^6 t C for the equilibrium landscape in the area cleared in

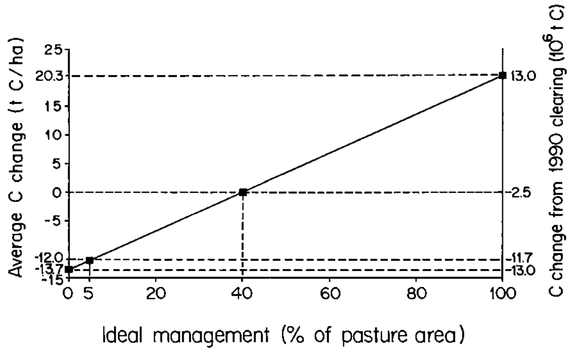

Fig. 4. Sensitivity of soil carbon stock changes to the percentage of the actively managed pasture area that is under ideal management. This percentage is believed to be about 5%; an improbably high 40% would have to be under ideal management to turn pasture soils from a net source to a net sink of carbon. The scale at the right indicates the effect on the soil carbon stock in the 1.38×10^6 ha cleared in 1990 when converted to the equilibrium landscape (which includes agriculture and secondary forest, as well as pasture).

1990. If 0–10% is taken as a reasonable range for the percentage of pastures under ideal management, at the lower bound pasture would be a source of 10.3 t C/ha or 10.4×10^6 t C for the equilibrium landscape from clearing in 1990. An improbably high 40% would have to be under ideal management in order to turn pasture soils from a net source to a net sink of carbon.

The varied estimates produced by different studies for the magnitude of soil carbon changes resulting from conversion of forest to pasture gives reason for caution in using the average values computed here. However, there is reason to believe that the overall conclusion is robust that conversion of forest to pasture in Brazilian Amazonia in 1990 results in a substantial emission of carbon. All of the studies from sites with ‘typical management’ resulted in soil C losses relative to the pre-conversion stock, while all studies on sites with ‘ideal management’ resulted in carbon gains (Table 3). Nevertheless, variability in the values is high in both cases.

An indication of both the robustness of the overall conclusion and of the low level of confidence that can be placed in the exact values of the numerical results is provided by an alternative calculation in which the two highest values for carbon loss under typical management were excluded (the two studies by Falesi, 1976). When this was done, the mean change in the top 20 cm relative to the pre-conversion stock decreased from −4.9 t C/ha (Table 3) to −3.1 t C/ha, while the percentage change diminished from −17.8% (Table 3) to −9.5%. This reduced the weighted average for C stock change for the full 0–800 cm profile from −12.0 t C/ha (Table 4) to −8.6 t C/ha. The overall conclusion also remained the same when the percentage of pasture that was assumed to be under ideal management was simultaneously doubled to 10%.

The results for changes below 1 m depth are clearly the least trustworthy, as they are based on a single study (Trumbore et al., 1995). The site of this study (Paragominas, Pará) has a stronger dry season than most of the forested portion of the Brazilian Legal Amazon; the strong dry season leads the forest to allocate a greater proportion of its carbon to deep roots, making the loss of C inputs to the deep soil greater when forest is replaced by pasture. The climate at Paragominas, although atypical of Amazon forest as a whole, is not atypical of the places where deforestation activity was concentrated in 1990 (Fearnside, 1993).

6. Conclusions

The literature indicates a mixture of findings for soil carbon stocks ranging from increases to decreases as a result of conversion of forest to pasture. Some of the varied results can be explained by correction (or lack of correction) for factors such as soil compaction and clay content, and the effect of the short-term seasonal cycles. Factors like sampling depth, number of samples, soil type, dominant vegetation and the quantity and type of carbon previously present are of fundamental importance to calculating mean values for use in simulations of carbon emissions and uptakes. The need is evident for longitudinal studies monitoring soil carbon stocks and related parameters in long-term plots established in areas converted from forest to pasture. Whether pasture soils are a net sink or a net source of carbon depends on their management, but an approximation of the fraction of pastures under ‘typical’ and ‘ideal’ management practices indicates that pasture soils in Brazilian Amazonia are a net carbon source, with an average release of 12.0 t C/ha under pasture. The equilibrium landscape would release 11.7×10^6 t C, of which 2.5×10^6 t C (21%) is the result of agriculture and 9.2×10^6 t C (79%) is from the effects of pasture conversion. Although this represents only 4.4% of the impact of 1990 deforestation (biomass + soil emissions), it represents a carbon release equivalent to approximately 20% of Brazil's annual emission from fossil fuels.

Acknowledgements

We thank Thierry Desjardins, Wim G. Sombroek, Paulo Maurício Lima de Alencastro Graça and Flávio Luizão for comments on the manuscript.

References

Allen, J.C., 1985. Soil response to forest clearing in the United States and the tropics: Geological and biological factors. Biotropica 17 (1), 15–27.

Andreux, F.G., Cerri, C.C., Eduardo, B.P., Choné, T., 1990. Humus contents and transformations in native and cultivated soils. Sci. Total Environ. 90, 249–265.

Arima, E.Y., Uhl, C., in press. Ranching in the Brazilian Amazon in a national context: Economics, policy, and practice. Society and Natural Resources.

Barbosa, R.I., 1991. Erosão do solo na Colônia do Apiaú, Roraima, Brasil: dados preliminares. Boletim Museu Integrado Roraima 1 (2), 22–40.

Barbosa, R.I., 1994. Efeito Estufa na Amazônia: Estimativa da Biomassa e a Quantificaçáo do Estoque e Liberação de Carbono na Queima de Pastagens Convertidas de Florestas Tropicais em Roraima, Brasil. Masters thesis in ecology. Instituto Nacional de Pesquisas de Amazônia (INPA) and Universidade Federal do Amazonas (UFAM), Manaus, Amazonas, Brazil, 85 pp.

Barbosa, R.I., Fearnside, P.M., 1996. Pasture burning in Amazonia: dynamics of residual biomass and the storage and release of aboveground carbon. J. Geophys. Res. (Atmos.) 101 (D20), 25847–25857.

Batjes, N.H., Sombroek, W.G., 1997. Possibilities for carbon sequestration in tropical and subtropical soils. Global Change Biol. 3, 161–173.

Brazil, Instituto Nacional de Pesquisas Espaciais (INPE), 1996. Levantamento das Áreas Desflorestadas na Amazônia Legal no Período 1991–1994: Resultados. INPE, São José dos Campos, São Paulo, Brazil, 9 pp.

Brazil, Projeto RADAMBRASIL, 1973–1983. Levantamento de Recursos Naturais, Vols. 1–23. Ministério das Minas e Energia, Departamento Nacional de Produção Mineral (DNPM) Rio de Janeiro, Brazil.

Brown, S., Lugo, A.E., 1982. The storage and production of organic matter in tropical forests and their role in the global carbon cycle. Biotropica 14 (2), 161–187.

Buschbacher, R.J., 1984. Changes in Productivity and Nutrient Cycling Following Conversion of Amazon Rainforest to Pasture. PhD Dissertation, Univ. of Georgia, Athens, GA, USA, 193 pp.

Cerri, C.C., Moraes, J.F.L., Volkoff, B., 1992. Dinâmica do carbono orgânico em solos vinculados a pastagens na Amazônia. Rev. INIA Invest. Agr. 1 (1), 95–102.

Cerri, C.C., Volkoff, B., 1987. Carbon content in a yellow latosol of Central Amazon rain forest. Acta Oecologica 8 (1), 29–42.

Cerri, C.C., Volkoff, B., Andreux, F., 1991. Nature and behaviour of organic matter in soils under natural forest, and after deforestation, burning and cultivation, near Manaus. For. Ecol. Manage. 38, 247–257.

Cerri, C.C., Volkoff, B., Eduardo, B.P., 1985. Efeito do desmatamento sobre a biomassa microbiana em latossolo amarelo da Amazônia. Revista Brasileira Ciên. Solo 9 (1), 1–4.

Cerri, C.C., Bernoux, M., Feigl, B.J., 1996. Deforestation and use of soil as pasture: Climatic impacts. In: Lieberei, R., Reisdorff, C., Machado, A.D. (Eds.), Interdisciplinary Research on the Conservation and Sustainable Use of the Amazonian Rain Forest and its Information Requirements. Forschungszentrum Geesthacht (GKSS), Bremen, Germany, pp. 177–186.

Chauvel, A., Grimaldi, M., Tessier, D., 1991. Changes in pore-space distribution following deforestation and revegetation: An example from the Central Amazon Basin, Brazil. For. Ecol. Manage. 38, 247–257.

Choné, T., Andreux, F., Correa, J.C., Volkoff, B., Cerri, C.C., 1991. Changes in organic matter in an oxisol from the Central Amazonian forest during eight years as pasture, determined by ^{13}C isotopic composition. In: Berthelin, J. (Ed.), Diversity of Environmental Biogeochemistry. Elsevier, Amsterdam, pp. 397–405.

Cunningham, R.H., 1963. The effect of clearing a tropical forest soil. J. Soil Sci. 14, 334–344.

Davidson, E.A., Nepstad, D.C., Trumbore, S.E., 1993. Soil carbon dynamics in pastures and forests of the eastern Amazon. Bull. Ecol. Soc. Am. 74 (2), 208, Abstract.

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

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

Detwiler, R.P., Hall, C.A.S., 1988. Tropical forests and the global carbon cycle. Science 239, 42–47.

Eden, M.J., Furley, P.A., McGregor, D.F.M., Milliken, W., Ratter, J.A., 1991. Effect of forest clearance and burning on soil properties in northern Roraima, Brazil. For. Ecol. Manage. 38, 283–290.

Falesi, I.C., 1976. Ecossistema de Pastagem Cultivada na Amazônia Brasileira. (Boletim Técnico No. 1). Empresa Brasileira de Pesquisa Agropecuária/Centro de Pesquisas Agro-Pecuárias do Trópico Úmido (EMBRAPA/CPATU), Belém, Pará, Brazil, 193 pp.

Fearnside, P.M., 1978. Estimation of Carrying Capacity for Human Populations in a Part of the Transamazon Highway Colonization Area of Brazil. PhD Dissertation in Biological Sciences, Univ. of Michigan, Ann Arbor. Univ. Microfilms Int., Ann Arbor, MI, USA, 624 pp.

Fearnside, P.M., 1980a. The effects of cattle pasture on soil fertility in the Brazilian Amazon: Consequences for beef production sustainability. Trop. Ecol. 21 (1), 125–137.

Fearnside, P.M., 1980b. The prediction of soil erosion losses under various land uses in the Transamazon Highway colonization area of Brazil. In: Furtado, J.I. (Ed.), Tropical Ecology and Development: Proceedings of the Vth International Symposium of Tropical Ecology, 16-21 April 1979, Kuala Lumpur, Malaysia. Int. Soc. Tropical Ecology (ISTE), Kuala Lumpur, Malaysia, pp. 1287–1295.

Fearnside, P.M., 1985. Brazil's Amazon forest and the global carbon problem. Interciencia 10 (4), 179–186.

Fearnside, P.M., 1986a. Brazil's Amazon forest and the global carbon problem: reply to Lugo and Brown. Interciencia 11 (2), 58–64.

Fearnside, P.M., 1986b. Human Carrying Capacity of the Brazilian Rainforest. Columbia University Press, New York, USA, 293 pp.

Fearnside, P.M., 1989. Ocupação Humana de Rondônia: Impactos, Limites e Planejamento. Relatórios de Pesquisa No. 5. Conselho Nacional de Desenvolvimento Científico e Tecnológico (CNPq), Brasília, Brazil, 76 pp.

Fearnside, P.M., 1993. Deforestation in Brazilian Amazonia: the effect of population and land tenure. Ambio 22 (8), 537–545.

Fearnside, P.M., 1996. Amazon deforestation and global warming: carbon stocks in vegetation replacing Brazil's Amazon forest. For. Ecol. Manage. 80, 21–34.

Fearnside, P.M., 1997a. Amazonie: la déforestation repart de plus belle. La destruction de la forêt dépend étroitemente des choix politiques. La Recherche 294, 44–46.

Fearnside, P.M., 1997b. Greenhouse gases from deforestation in Brazilian Amazonia: net committed emissions. Clim. Change 35 (3), 321–360.

Fearnside, P.M., Leal Filho, N., Fernandes, F.M., 1993. Rainforest burning and the global carbon budget: Biomass, combustion efficiency and charcoal formation in the Brazilian Amazon. J. Geophys. Res. (Atmos.) 98D9, 16733–16743.

Feigl, B.J., Steudler, P.A., Cerri, C.C., 1995. Effects of pasture introduction on soil CO_2 emissions during the dry season in the state of Rondônia, Brazil. Biogeochemistry 31, 1–14.

Feller, C., Fritsch, E., Poss, R., Valentin, C., 1991. Effet de la texture sur le stockage et la dynamique des matières organiques dans quelques sols ferugineux et ferralitiques (Afrique de l'Ouest, en particuler). Cahiers ORSTOM, Série Pédologie 26, 25–36.

Fisher, M.J., Rao, I.M., Ayarza, M.A., Lascano, C.E., Sanz, J.I., Thomas, R.J., Vera, R.R., 1994. Carbon storage by introduced deep-rooted grasses in South American savannas. Nature 371, 236–238.

Greenland, D.J., Nye, P.H., 1959. Increases in the carbon and nitrogen contents of tropical soils under natural fallows. J. Soil Sci. 10, 284–299.

Guimarães, W.M., 1993. Liberação de Carbono e Mudanças nos Estoques de Nutrientes Contidos na Biomassa Aérea e no Solo, Resultante de Queimadas de Florestas Secundárias em Áreas de Pastagens Abandonadas, em Altamira, Pará. Masters Thesis. Instituto Nacional de Pesquisas da Amazônia/Fundação Universidade do Amazonas (INPA/FUA), Manaus, Amazonas, Brazil, 69 pp.

Hecht, S.B., 1981. Deforestation in the Amazon Basin: magnitude, dynamics and soil resource effects. Studies Third World Soc. 13, 61–108.

Hecht, S.B., 1982. Cattle Ranching Development in the Eastern Amazon: Evaluation of a Development Policy. Ph.D. Dissertation, Univ. of California, Los Angeles, CA, USA, 454 pp.

Houghton, R.A., Hobbie, J.E., Melillo, J.M., Moore, B., Peterson, B.J., Shaver, G.R., Woodwell, G.M., 1983. Changes in the carbon content of the terrestrial biota and soils between 1860 and 1980: a net release of CO_2 to the atmosphere. Ecol. Monogr. 53 (3), 235–262.

Houghton, R.A., Boone, R.D., Fruci, J.R., Hobbie, J.E., Melillo, J.M., Palm, C.A., Peterson, B.J., Shaver, G.R., Woodwell, G.M., 1987. The flux of carbon from terrestrial ecosystems to the atmosphere in 1980 due to changes in land use: geographic distribution of the global flux. Tellus 1–2, 122–139.

Intergovernmental Panel on Climate Change (IPCC)/Organization for Economic Cooperation and Development (OECD) Joint Programme, 1994. Greenhouse Gas Inventory. OECD, Paris, France, 3 vols.

Kauffman, J.B., Cummings, D.L., Ward, D.E., Babbitt, R., 1995. Fire in the Brazilian Amazon: 1. Biomass, nutrient pools, and losses in slashed primary forests. Oecologia 104, 397–408.

Keller, M., Kaplan, W.A., Wofsy, S.C., 1986. Emissions of N_2O, CH_4 and CO_2 from tropical forest soils. J. Geophys. Res. (Atmos.) 91D11, 11791–11802.

Keller, M., Veldkamp, E., Weltz, A.M., Reiners, W.A., 1993. Effect of pasture age on soil trace-gas emissions from a deforested area of Costa Rica. Nature 365, 244–246.

Lal, R., 1977. Analysis of factors affecting rainfall erosivity and soil erodibility. In: Greenland, D.J., Lal, R. (Eds.), Soil Conservation and Management in the Humid Tropics. Wiley, New York, pp. 49–56.

Laudelot, H., 1961. Dynamics of Tropical Soils in Relation to their Fallowing Techniques. Paper 11266/E. Food and Agricultural Organization of the United Nations (FAO), Rome, Italy, 111 pp.

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

Luizão, F., Luizão, R., Chauvel, A., 1992. Premiers résultats sur la dynamique des biomasses racinaires et microbiennes dans un latosol d'Amazonie centrale (Brésil) sous forêt et sous pâturage. Cahiers Orstom-Pédologie 27 (1), 69–79.

Mann, L.K., 1986. Changes in soil carbon storage after cultivation. Soil Sci. 142 (5), 279–288.

Martius, C., Fearnside, P.M., Bandeira, A.G., Wassmann, R., 1996. Deforestation and methane release from termites in Amazonia. Chemosphere 33 (3), 517–536.

Mattos, M.M., Uhl, C., 1994. Economic and ecological perspectives on ranching in the Eastern Amazon. World Dev. 22 (2), 145–158.

Mitchell, J.K., Bubenzer, G.D., 1980. Soil loss estimation. In: Kirkby, M.J., Morgan, R.P.C. (Eds.), Soil Erosion. Wiley, New York, pp. 17–62.

Moraes, J.L. de, Cerri, C.C., Melillo, J.M., Kicklighter, D., Neil, C., Skole, D.L., Steudler, P.A., 1995. Soil carbon stocks of the Brazilian Amazon Basin. Soil Sci. Soc. Am. J. 59, 244–247.

Moraes, J.F.L. de, Volkoff, B., Cerri, C.C., Bernoux, M., 1996. Soil properties under Amazon forest and changes due to pasture installation in Rondônia, Brazil . Geoderma 70, 63–81.

Neill, C., Fry, B., Melillo, J.M., Steudler, P.A., Moraes, J.F.L., Cerri, C.C., 1996. Forest- and pasture-derived carbon contributions to carbon stocks and microbial respiration of tropical pasture soils. Oecologia 107, 113–119.

Nepstad, D.C., Carvalho, C.R., Davidson, E.A., Jipp, P.H., Lefebvre, P.A., Negreiros, G.H., Silva, E.D., Stone, T.A., Trumbore, S.E., Vieira, S., 1994. The role of deep roots in the hydrological cycles of Amazonian forests and pastures. Nature 372, 666–669.

Nepstad, D.C., Jipp, P., Moutinho, P., Negreiros, G., Vieira, S., 1995a. Forest recovery following pasture abandonment in Amazonia: Canopy seasonality, fire resistance and ants. In: Rapport, D.J., Gaudet, C.L., Calow, P. (Eds.), Evaluating and Monitoring the Health of Large-Scale Ecosystems. NATO ASI Series Vol. 128. Springer, Berlin, pp. 333–349.

Nepstad, D.C., Klink, C., Trumbore, S.E., 1995b. Pasture soils as carbon sink. Nature 376, 472–473.

Nepstad, D.C., Uhl, C., Serrão, E.A.S., 1991. Recuperation of a degraded Amazonian landscape: forest recovery and agricultural restoration. Ambio 20, 248–255.

Nye, P.H., Greenland, D.J., 1960. The Soil under Shifting Cultivation. Technical Communication No. 51. Commonwealth Agricultural Bureaux of Soils, Harpenden, UK, 156 pp.

Post, M.W., Anderson, D.W., Dahmke, A., Houghton, R.A., Huc, A.Y., Lassiter, R., Majjar, R.G., Meue, H-V., Pedersen, T.F., Trumbore, S.E., Vaikmae, R., 1995. Group report: What is the role of nonliving organic matter cycling on the global scale? In: Zepp, C.H., Sontag, R.G. (Eds.), Role of Nonliving Organic Matter in the Earth's Carbon Cycle. Wiley, New York, pp. 155–174.

Post, M.W., Emanuel, W.R., Zinke, P.J., Stangerberger, A.G., 1982. Soil carbon pools and world life zones. Nature 298, 156–159.

Reiners, W.A., Bouwman, A.F., Parsons, W.F.J., Keller, M., 1994. Tropical rain forest conversion to pasture: Changes in vegetation and soil properties. Ecological Applications 4, 363–367.

Sánchez, P.A., 1976. Properties and Management of Soils in the Tropics. Wiley, New York, 618 pp.

Sánchez, P.A., Villachica, J.H., Bandy, D.E., 1983. Soil fertility dynamics after clearing a tropical rainforest in Peru. Soil Sci. Soc. Am. J. 47 (6), 1171–1178.

Scott, G.A.J., 1975. Soil profile changes resulting from the conversion of forest to grassland in the Montaña of Peru. Great Plains–Rocky Mountain Geogr. J. 4, 124–130.

Scott, G.A.J., 1978. Grassland Development in the Gran Pajonal of Eastern Peru: a Study of Soil-Vegetation Nutrient Systems. Hawaii Monographs in Geography No. 1. Univ. of Hawaii at Manoa, Honolulu, Hawaii, USA, 187 pp.

Sombroek, W.G., Nachtergaele, F., Hebel, A., 1993. Amounts, dynamics and sequestering of carbon in tropical and subtropical soils. Ambio 22 (7), 417–426.

Serrão, E.A.S., Falesi, I.C., 1977. Pastagens do Trópico Úmido Brasileiro. Empresa Brasileira de Pesquisa Agropecuária/Centro de Pesquisas Agro-Pecuárias do Trópico Úmido (EMBRAPA/CPATU), Belém, Pará, Brazil, 63 pp.

Suares de Castro, F., 1957. Las quemas como práctica agricola y sus efectos. Federación Nacional Cafetaleros de Colombia Boletin Técnico, 2.

Teixeira, L.B., 1987. Dinâmica do Ecossistema de Pastagem Cultivada em Área de Floresta na Amazônia Central. PhD dissertation in ecology, Instituto Nacional de Pesquisas da Amazônia (INPA) and Fundação Universidade do Amazonas (FUA), Manaus, Amazonas, Brazil, 100 pp.

Townsend, A.R., Vitousek, P.M., Holland, E.A., 1992. Tropical soils could dominate the short-term carbon cycle feedbacks to increase global temperatures. Clim. Change 22, 293–303.

Trumbore, S.E., Davidson, E.A., Camargo, P.B., Nepstad, D.C., Martinelli, L.A., 1995. Below-ground cycling of carbon in forests and pastures of eastern Amazonia. Global Biogeochem. Cycles 9 (4), 515–528.

Uhl, C., Buschbacher, R., Serrão, E.A.S., 1988. Abandoned pastures in Eastern Amazonia: I. Patterns of plant succession. J. Ecol. 76, 663–681.

Uhl, C., Jordan, C.F., 1984. Succession and nutrient dynamics following forest cutting and burning in Amazonia. Ecology 65 (5), 1476–1490.

Uhl, C., Kauffman, J.B., 1990. Deforestation, fire susceptibility, and potential tree responses to fire in the Eastern Amazon. Ecology 71, 437–449.

Veldkamp, E., 1993. Soil Organic Carbon Dynamics in Pastures Established after Deforestation in the Humid Tropics of Costa Rica. PhD dissertation, Agricultural University of Wageningen, Wageningen, The Netherlands, 117 pp.

Veldkamp, E., 1994. Organic carbon turnover in 3 tropical soils under pasture after deforestation. Soil Sci. Soc. Am. J. 58, 175–180.

Veldkamp, E., Weitz, A.M., 1994. Uncertainty analysis of δ^{13} C method in soil organic matter studies. Studies Biol. Biochem. 26 (6), 153–160.

Zinke, P.J., Sabhasri, S., Kunstadter, P., 1978. Soil fertility aspects of the Lua' forest fallow system of shifting cultivation. In: Kunstadter, P., Chapman, E.C., Sabhasri, S. (Eds.), Farmers in the Forest: Economic Development and Marginal Agriculture in Northern Thailand. East–West Center, Honolulu, HI, USA, pp. 134–159.