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Chapter C3:
The Maintenance of Soil Fertility in Amazonian Managed Systems

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Abstract

Most of Brazilian Amazonia faces important limitations for conventional agriculture and pastures due to a generally poor chemical fertility as well as the region’s environmental conditions, especially high temperature and moisture. Without proper management, degradation of the soil and resulting unsustainability of agricultural and ranching production occur within a few years, leading to land abandonment. Use of perennial crops, especially those based on native tree species, would be instrumental in order to achieve best management such as that which assure recycling processes similar to those in the primary forest. Recommended alternative land uses are those producing high soil organic matter, recycling of nutrients, substantial agricultural production and economic viability. These include agroforestry systems, enrichment of second growth with valuable native timber or fruit species, accelerated fallow regrowth via enrichment plantings, sequential agroforestry with slash-and-mulch, and diversified forest plantations. Improvement of agricultural soils can be based on lessons learned from the study of processes involved in the formation and maintenance of the rich “dark earths” (terra preta), which owe their high carbon content and fertility in part to high content of charcoal. Adding powdered charcoal combined with selected nutrients can increase soil carbon in modern agriculture. Considering that limitations to expansion of intensified land uses in Amazonia are serious, regional development should emphasis the natural forest, which can maintain itself without external inputs of nutrients. Instead of creating conditions to further expand deforestation, these forests may be used as they stand to provide a variety of valuable environmental services that could offer a sustainable basis for development of Amazonia.

C.3.1 Introduction

C.3.1.1 The role of soil-fertility maintenance in Amazonian development

The Amazon tropical forest is one of the world’s last remaining forests that is sufficiently large and intact to provide globally important environmental services [ASB, 2002], occupying 7.86 million km² in nine countries, and covering approximately 45% of the South American continent. More than 60% of the Amazon forest is located in northern Brazil, an area larger than the whole of western Europe [INPE, 2000].

In recent decades, Brazil’s Amazon forest has been rapidly destroyed and replaced by cattle pasture. Much smaller areas are maintained under agricultural uses such as soybeans or other annual crops, and a very small proportion becomes perennial crops, as in
agroforestry systems. According to the last official survey of land use in Amazonia [IBGE, 2006], 29% of Brazilian lands (or 249 million ha) were occupied by agrosilvopastoril activities, mainly pastures (70% of the total). While in the country as a whole pasture cover decreased by 3.8% relative to the year 1985, in the five million km² Legal Amazonia region there was a 44.2% expansion in pastureland between 1985 and 2006. Pastures represent 82.3% of the land occupied by agrosilvopastoril activities in the Brazilian Legal Amazonia, or 61.6 million hectares [IBGE, 2006]. In national terms, this means that 36% of the country’s cattle ranching is concentrated in Legal Amazonia, as well as 39% of the soybean and 47% of the cotton cultivation in the country. Additionally, 6% of the sugarcane ethanol produced by Brazil came from Legal Amazonia [Smeraldi and May, 2008].

Although “cerrado” occupies 16% of the Legal Amazonia and has high productivity of pastures and crops, large areas of Amazonian forests were also converted into pastures or croplands. This is particularly true for cattle ranching: 73% of the cattle in the region are located in the ‘Amazon biome’, meaning areas of Amazon forest [Smeraldi and May, 2008]. On the other hand, only 2% of soybean production in Legal Amazonia was supposedly produced in former forest lands, while no cotton production took place (all produced in cerrado lands, although some of these were located within the limits of the ‘biological’ Amazonia) (Figure 1).

These land-use changes have important impacts on soil fertility, often leading to degradation of the soil and resulting in unsustainable agricultural and ranching production. (here, our working definition of ‘sustainability’ is the maintenance of the basic functions and mechanisms of the original ecosystem, making the man-made system economically productive for long periods without the constant need for external inputs such as inorganic fertilizers. Of course this implies the conservation of functional biodiversity and of large tracts of natural habitats.

The lack of proper land management, together with natural constraints such as excessive moisture and high temperature, high acidity and low supply of soil nutrients in most of the region, have been causes of limited productivity of food and fiber in Brazilian Amazonia. However, development of new technologies and the understanding of principles behind the practices of traditional populations living in the region may be used to change this situation. Also, a better valuation of the many environmental services of the standing forest may be used as a sound basis for sustainable development in Amazonia.

C.3.1.2 Soil-fertility dynamics under Amazonian land uses

Starting in the early 1960s, the Brazilian Government tried to use the Amazon’s abundant natural resources (forests, agricultural lands and minerals) to fuel regional and national economic growth. However, initial attempts to develop the region through government-oriented establishment of agricultural settlements ran into serious difficulties. Following the establishment of federally subsidized credit in the late 1960s, hundreds of agricultural and industrial projects were approved and implemented in the Amazon, but most of the agricultural projects failed and were abandoned. Likely the main reason for failure was that the process of assisted migration and colonization was rapid and intense, and millions of hectares of forested land were handed over to newcomers with little knowledge of the
potential of these areas to support agriculture, and little consideration was given to soil, water or watershed conditions when sites were chosen [Walker and Homma, 1996].

In the Amazon, two land-use systems (neither of which sustainable if applied in a continuous way) are widely practiced by farmers [ASB, 2002]:
(i) traditional pasture: generally after growing annual crops for 2 or 3 years, farmers plant pasture grass, usually Brachiaria sp., and the pastures are burned to control weeds and insects, sometimes annually, with little or no other management;
(ii) traditional (short) annual crop/fallow rotation: the annual food crops, usually grown for only 2 years, are followed by 3 or more years of natural bush fallow. Usually this is again followed by a rotation of annual crops, after which the plot is usually dedicated to pasture.

The use of slash-and-burn as the main land preparation method for agriculture in the Brazilian Amazon occurs during the dry season found in most of the region (with 1-4 dry months between June and October), which stands in contrast to the slash-and-mulch system found in more humid Amazonian areas but seldom practiced in Brazil [ASB, 2002]. In the slash-and-burn process, the cleared vegetation is allowed to dry and is then burned before the onset of the rains, when the area is planted to annuals, perennials or pasture grasses, using the nutrients released from biomass burning. However, despite this initial fertilization through ashes, which are rich in cations, crop and pasture production are not durable or economically sustained in the region due to several natural and human-induced constraints. One reason is that soil quality varies widely and is patchy, but the predominant soil types - Oxisols and Ultisols - usually have relatively low natural fertility, with high levels of acidity, low phosphorus contents, low levels of cation exchange and high levels of aluminum toxicity [Cochrane and Sanchez, 1982]. Although generalizations on soil fertility levels for a large region like Amazonia must be avoided, only 7% of the land area in Brazilian Amazonia is considered to be free from major limitations on plant growth, while soil phosphorus (P) deficiencies are believed to constrain productivity in 90% of the area, and aluminum (Al) toxicity occurs in 73% of the soils [Cochrane and Sanchez, 1982]. Such limitations appear to be particularly evident in the central and eastern Amazon, far from the Andes, and where most soils are very old and weathered. These soils sometimes have nutrient concentrations that are far too low for plant growth (e.g., Table 1). Such infertile soils are fragile and need special care if they are used for any kind of cultivation after forest removal.

On old, heavily-weathered soils, regular inputs of new nutrients from soil parent material are very small or even negligible [Schubart et al., 1984; Brinkmann, 1989]. Thus, aboveground biomass and the litter layer are two important reservoirs of plant nutrients for Amazonian forest [Brinkmann, 1989; Anderson and Spencer, 1991]. Under such conditions, atmospheric deposition (both dry and wet) also becomes an important source of nutrients, especially for base cations [Brinkmann, 1989], compensating for the small losses from the ecosystem caused by leaching. However, most of the nutrient demand of the plants is met through biologically-mediated remineralization of nutrients from the organic matter. Generally, nutrients are rapidly and efficiently cycled in the lowland evergreen rain forest ecosystem and most nutrients have a very short residence time in the soil-litter system, being quickly remineralized and made available to the plants [Herrera et
The concentration of mineral nutrients in the primary forest litter in the Amazon is generally high for nitrogen; however, for phosphorus and base cations these considerations are lower than in other tropical lowland evergreen rain forests [Proctor, 1984]. The lower concentrations can be considered as an indication of nutrient limitation for the plants in this region [Vitousek, 1984], and these nutrients are strongly translocated and retained in the plants prior to foliar abscission. However, other patterns exist in the region as well: for instance, in Maracá, northern Amazonia, Scott et al. [1992] found nutrient concentrations in litter (especially for phosphorus and calcium) that are much higher than the average values for Amazonia despite high translocation for some nutrients. In general, rapid cycling of nutrients in tropical forests is achieved through high decomposition rates, which are made possible by high temperatures and high annual rainfall, which boosts biological activity in soil and litter.

Despite the intrinsic low chemical fertility in the majority of Amazonian soils, and the relatively low content of soil carbon (C) in most of the region [Moraes et al., 1996; Cerri et al., 2003], soils generally have good physical structure. This, together with the complex, specialized and very active soil biota, can maintain the natural fertility of soils if they are kept well covered and protected against direct sun and raindrop impact, as occurs under the natural forest cover [Ross et al., 1990]. Some of the limiting factors for crop production can be surpassed through technological advances, but others cannot be overcome at a reasonable cost in the region, where there are restrictions for both the intensification of agriculture and ranching uses and the scale to which these land uses can be expanded [Fearnside, 1997a]. These restrictions include agronomic limits on per-hectare yields, physical resource limits such as phosphate deposits available for soil fertilization, and environmental risks. Thus, other strategies should be adopted to benefit Amazonia’s human population, and a number of projects have been searching for sound alternatives to slash-and-burn agriculture [Palm et al., 2001, 2005; Almeida et al., 2006]. In recent years, special emphasis has been given to keeping the forest standing through the creation of compensation mechanisms for the environmental services of intact forest [Fearnside, 1997b, 2008a]; scientists and decision-makers are approaching a consensus that such mechanisms must be pursued and put into action. Therefore, agricultural production would be directed to the already deforested areas in Amazonia, most of which are abandoned or degraded and need alternative practices (e.g., fallow enrichment for no-burn agriculture, agroforestry, etc.) to become productive again.

C.3.1.3 Conversion of forest to well-managed pasture: effect on carbon accumulation in soils

Cattle pastures represent the largest single use of cleared forest land in most of the Brazilian Amazon. Estimates show that 70% of the deforested land has been converted to pastures at one stage or another [Serrão and Toledo, 1990; Dias-Filho et al., 2001]. About 45% of the deforested land in Brazilian Amazonia is occupied by actively grazed cattle pasture, or 24.7 Mha [Fearnside and Barbosa, 1998]. Similar statistics were reported by Homma [1994] and Kitamura [1994]. Farmers were motivated to convert lands cleared from forest into pasture because of the real or perceived increases in land value. Farmers not only maintained cattle as standing “bank accounts” and obtained cash from sales of
animals and milk, but also increased the value of these “savings” by investing time and resources in pasture, fencing, corrals, and ponds [Fujisaka et al., 1996].

Despite the enormous scale of pasture expansion in the Amazon, there is still no clear understanding of the direction of the resulting changes in soil C stocks. Fearnside and Barbosa [1998] reported that conversion of Amazon forest to pasture can produce a net soil C sink (well-managed pasture) or a net C source (overgrazed pasture), depending on management. Neill and Davidson [1999] observed that conversion of forest to pasture in the Amazon occurs on a variety of soils and in regions that differ in the amount and timing of precipitation. The sequence leading to pasture development also differs. Some pastures are created by planting grasses directly into forest slash, while others are created after one or two years of annual cropping or after a cropping and fallow sequence. The choices of grass species and the practices of interplanting with legumes also differ. These factors can influence whether a pasture soil will become a source or sink of C. Once established, pasture management by stocking rate, burning frequency, effectiveness of weed control, fertilizing or disking may also affect soil C balance [Neill and Davidson, 1999].

Therefore, in some locations, C stocks in pastures are lower compared with the original forest [Luizão et al., 1992b; Desjardins et al., 1994]. In other locations, pasture grass productivity declines in older pastures, but soil C concentrations remain relatively constant [Falesi, 1976; Serrão et al., 1979; Buschbacher et al., 1988]. Yet in other locations, inputs of C from roots of pasture grasses cause increases in soil C stocks [Cerri et al., 1991; Bonde et al., 1992; Trumbore et al., 1995; Moraes et al., 1996; Neill et al., 1997; Bernoux et al., 1998; Cerri et al., 2003].

Neill and Davidson [1999] synthesized the available literature on soil C stocks in pasture following deforestation in the Amazon. They reported that 19 of 29 pastures examined accumulated C in surface soils and 10 showed C losses. They also observed a strong relationship between pasture grass species and the change in surface soil C stocks. Pasture planted to Brachiaria humidicola lost C and those planted to Panicum maximum and Brachiaria brizantha gained C.

Moraes et al. [1996] found that total soil C contents to 30 cm depth in 20-year-old well-managed pastures were 17 to 20% higher than in the original forest sites in the western Amazon. A comparison of C budgets for forest and pastures in the eastern Amazon was made by Trumbore et al. [1995]. In a rehabilitated and fertilized pasture of Brachiaria brizantha, they estimated gains, relative to forest soil C stocks, of over 20 Mg C ha\(^{-1}\) in the top 1-m of soil and a loss of about 0.5 Mg C ha\(^{-1}\) in the 1-8 m soil depth during the first 5 years following pasture rehabilitation. More than 50% of the forest-derived C in surface soils of pastures in converted Amazon forest turns over in 10 to 30 years [Choné et al., 1991; Trumbore et al., 1995]. Cerri et al. [1999] reported carbon sequestration of 0.27 Mg C ha\(^{-1}\) yr\(^{-1}\) for the 0-30 cm depth range (C-sequestering rates for a 20-year time range) Neill et al. [1997] reported annual soil C accumulation rate, in the top 50 cm, in the range of 0.2 to 0.3 Mg ha\(^{-1}\) for second growth ages from 3 to 23 years. These results are in the range (0.2-3.9 Mg C ha\(^{-1}\) yr\(^{-1}\)) of those reported by Sampson et al. [2000, p. 199] for pastures in wet tropical areas of the world. For the present study, a mean accumulation rate of 0.27 Mg C
ha\(^{-1}\) yr\(^{-1}\)) in the top 30-cm soil layer is assumed. If 20% of the pastures are well-managed (about 5 Mha), the potential soil C sequestration is between 1 and 19.5 Tg C yr\(^{-1}\) in the well-managed pasture. However, carbon losses in the much larger area of poorly managed pasture dwarf this gain.

In contrast to annual crops, pasture grasses maintain a continuous vegetative cover on the soil, reducing soil temperatures, and often having high productivity and rates of turnover (particularly belowground) that add organic matter to the soil [Brown and Lugo, 1990]. However, any difference between soil contents of C and N in forest and pasture ecosystems does not compensate for the fact that very significant aboveground stocks of C (100-300 Mg C ha\(^{-1}\)) are lost when forests are converted to pasture or agriculture [Fearnside, 2000]. Dias-Filho et al. [2001] reported that forest-to-pasture conversion releases 100-200 Mg C ha\(^{-1}\) from aboveground forest biomass to the atmosphere. There are also additional benefits of intact forests in ameliorating floods, conserving soils, maintaining stable regional climates, preserving biodiversity, and supporting indigenous communities and ecotourism industries [Laurance et al., 2001]. Therefore, any policy changes that reduce the rate of deforestation would have the greatest potential for reducing the net emission of greenhouse gases. Moreover, it is also greatly desired that these policies enhance the rate of carbon sequestration in soil.

C.3.1.4 Conversion from Degraded to Well-Managed Pasture

More than half of the cattle pasture areas in the Brazilian Amazon are degraded [Serrão and Toledo, 1990; Dias-Filho, 2003], which represents approximately 13 Mha. Productivity of Amazonian pastures is often good during the first 3 to 5 years after establishment. A rapid decline in productivity of the planted grasses occurs after this period due to encroachment by herbaceous and woody invaders [Uhl et al., 1988; Serrão and Toledo, 1990]. If uncontrolled, invader species gradually dominate and severely degrade pastures, a condition characterized by a complete dominance of the weedy community [Dias-Filho et al., 2001]. If the entire area now under degraded pasture could be well managed, and assuming the rate of soil C sequestration for well-managed pastures of 0.2 to 3.9 Mg C ha\(^{-1}\) yr\(^{-1}\) in the 0-30 cm soil layer [Sampson et al., 2000], the potential soil C sequestration from converting degraded to well-managed pasture in the Brazilian Amazon varies from 2.6 to 51 Tg C yr\(^{-1}\). Note, however, that the high value of 3.9 Mg C ha\(^{-1}\) is much higher than is likely to occur over a wide area.

There are numerous factors and processes that must be considered in estimating the direction and rate of change in SOC contents from changes in soil management. Post and Kwon [2000] reported that important factors for increasing carbon sequestration include: i) increasing the input rates of organic matter; ii) changing the decomposability of organic input increasing the light organic carbon fraction; iii) placing organic matter deeper in the soil either directly by increasing belowground inputs or indirectly by enhancing surface mixing by soil organisms; and iv) enhancing physical protection through formation of aggregates or organo-mineral complexes.
C.3.2 What are the causes of decline in soil fertility under agricultural and forest management and what is their relative importance?

Amazonian soils seem particularly prone to rapid declines in natural fertility. One reason for this is their medium to low level of soil organic matter (SOM), which is generally associated with rapid C and nutrient mineralization due to favorable moisture conditions during most of the year. Low contents of SOM imply low retention of cations and, consequently, soils susceptible to nutrient leaching.

Any human intervention in the forest implies some degree of change in structure and functioning of the ecosystem, although some interventions are not destructive and may be sustainable over a fairly long time. Forest-management practices, such as selective logging, which supposedly causes little impact on soil, may still represent a considerable impact on soil properties, including soil compaction, erosion and leaching. Also, selectively logged forests become susceptible to fire due to the addition of large quantities of dry material on the soil surface and due to opening of pathways for wind to enter the forest; the wind dries out the soil surface and creates additional combustible material [Nepstad et al. 1999]. Thus, changes in carbon and nutrient cycling are introduced in managed forests after the first intervention.

The impacts of selective logging may be considerably reduced if carefully designed interventions are conducted, such as the reduced-impact logging (RIL) procedures (Table 2). In Pará state, a comparison between plots submitted to conventional selective logging (CL) and plots harvested by reduced impact logging (RIL) procedures, during 4 years, showed that the total area of soil affected (tractor tracks + skid roads + log decks) under CL was between 8.9 and 15.3%, while under RIL it was between 4.6 and 8.6% of the total area [Asner et al., 2004a]. The largest proportion of the plot damages (4-12%) was caused by the skid tracks and, by reducing this specific damage. RIL reduced canopy openings and the susceptibility of the residual forest to fire, producing less coarse woody debris [Asner et al., 2004a; Keller et al., 2004]. The amount of coarse litter (dead wood) found in plots, one year after selective logging, was 2.7 times higher under CL than under RIL and only 8-18% of the mass of the debris was small-diameter wood (< 10 cm), which undergoes faster decomposition. At Floresta Nacional do Tapajós, dead wood increased from 50.7 ± 1.1 Mg ha\(^{-1}\) in intact forest to 76.2 ± 10.2 in RIL, while in Cauaxi it increased from 55.2 ± 4.7 to 74.7 in RIL and to 108 ± 10.5 Mg ha\(^{-1}\) in CL [Keller et al., 2004]. However, despite the many advantages of RIL, most logging in Amazonia is not done in this way, and serious damage continues to be caused to the logged plots.

The impacts of selective logging on soil physical, chemical and biological features are obviously stronger in the log decks and parts of the logged plots such as the tractor tracks, which are submitted directly and repeated times to mechanical impacts, certainly affecting soil physical properties, thus changing the soil water availability to plants. Near Manaus, the BIONTE (Biomass and Nutrients in Tropical Moist Forest, INPA/DFID) Project has evaluated the effect of extracting 34.3 m\(^3\) of timber per ha (6-10 trees, with diameter at breast height -DBH > 55 cm) on the soil properties, nutrient cycling and forest regeneration [BIONTE, 1997]. Soon after the logging, tensiometer measurements showed an increase of 246% in soil water tension in the tractor tracks (to 430 kPA) and a 37%
increase in the center of clearings (to 240 kPA), as compared to the forest control (175 kPA) [Mello-Ivo and Ross, 2006]. The tractor tracks created by a former selective logging procedure 7 years before were still perfectly visible, treeless, and compacted, showing water accumulation on the surface, as a clear indication that these micro-sites could not be rehabilitated in the short or medium term.

Contrary to former assumptions that grasses produce a fair cover of the soil surface after slash-and-burn [Falesi, 1976], some studies show strong losses through erosion/leaching in Amazonia [Barbosa and Fearnside, 2000; Table 3]. For instance, in their 3.5-year study on soil erosion under two land uses - primary forest and adjacent pasture derived from forest in Apiaú, Roraima - Barbosa and Fearnside [2000] found that, for a slope of 20%, soil erosion under pasture (1128 kg ha\(^{-1}\) yr\(^{-1}\)) was 7.5 times higher than under forest (150 kg ha\(^{-1}\) yr\(^{-1}\)). The runoff was almost three times higher in pasture (31.8 cm yr\(^{-1}\)) than in primary forest (11.3 cm yr\(^{-1}\)).

Additional studies have also shown strong losses of soil through laminar erosion after forest conversion, such as the studies conducted on the Trans-Amazon Highway area by Smith [1976] and Fearnside [1980], both using the stake methodology. Both studies stressed that conversion of primary forest into pastures increases soil laminar erosion, which will certainly be reflected in the regional and global socio-economy. The major reason is that removal of forest cover exposes the soil surface to direct impacts of sun and of raindrops (e.g., an increase of 571 mm or 37% in the Apiaú study), in addition to the direct compaction caused by cattle trampling.

The above results confirm studies carried out in forested sites such as the study by Ross et al. [1990], on Maracá Island, Roraima (Table 3). These authors found that, just after the treatments, clear cut of forest increased fourfold the soil and nutrient losses in a topographic gradient, as compared to primary forest with intact litter and canopy cover, and losses were almost 2.5 times higher than in forest plots where canopy was removed but litter layer was kept in place.

In the clayey Oxisols of central Amazonia, the soil micro-pores - between 0.01 and 0.03 \(\mu m\) - result from the compact assemblage of clayey particles of kaolinite, while soil macro-pores - between 0.1 and 100 \(\mu m\) - result mainly from soil fissures and biological activity in various forms: galleries, channels, chambers, etc. [Grimaldi et al., 1993]. Under forest, Oxisols have a bimodal pore spectrum (small and large pores together). A study carried out near Manaus showed that mechanized deforestation before burning and planting caused a 70-80% decrease in pores > 0.1 \(\mu m\), which are the pores containing water available to plants, and the pore spectrum became virtually unimodal, with strong soil compaction between 20-40 cm depth. Under these conditions, hydraulic conductivity decreases, rain water accumulates on the soil surface, and infiltration can be 10 times slower than under forest [Grimaldi et al., 1993]. In the same study, it was found that a young and well-managed pasture at FUCADA (Center for Technical Support of the Manaus Agriculture and Ranching District), 3-5 years old following manual deforestation, showed a strong decrease in pores > 0.1 \(\mu m\), implying limitations to grass root growth, water infiltration and oxygen diffusion.
Besides the impact caused by cattle trampling, part of the soil compaction found in managed pastures may be a consequence of the strong reduction in the diversity of soil biota, thus influencing soil physical structure and fertility. An experimental soil manipulation carried out in central Amazonia showed that a clayey Oxisol under an old Brachiaria humidicola pasture was compacted in the upper layer by the action of the pantropical earthworm Pontoscolex corethrurus [Barros et al., 2001]. This earthworm was very abundant and widely dominant in the area; its feces contained a very high proportion of fine clay that, once deposited on the soil surface, produced a thin compact layer. When a 1-m³ block of the compacted pasture soil were moved and inserted into the neighboring forest in less than 1 year the forest organisms colonized the formerly compacted soil and produced aggregates and porosity similar to the forest soils; in the opposite way, the well-structured forest soil, once moved to the pasture became compact within a year [Barros et al., 2001]. The same pattern was observed in Marabá (PA), showing that soil biodiversity decreased sharply by 70 % under grazed pastures as compared to forest: soil compaction, which begins with rice cropping following slash-and-burn, was accelerated with pasture age and was stronger in the 2-5 cm and 5-10 cm soil layers (T. Desjardins, 2007, pers. comm.). These results emphasize the need for keeping a diversified soil biota (and a diversified mixture of plant species capable of feeding it) in order to maintain the soil structure and fertility.

C.3.3 Evidence of fertility decline in managed Amazonian forests

Changes in soil fertility in Amazonia are connected to the different levels of human intervention on the forest ecosystem. They may be slight and reversible, for example as in careful selective logging, or highly damaging as in a poorly-managed old pasture that degrades the soil system and makes recovery difficult. Even worse, changes can be highly destructive, as in mining operations, which, although normally restricted to small localized areas, generally represent a very intense destructive impact on native ecosystems. For instance, bauxite mining involves the removal of all vegetation and the entire topsoil layer, causing soil impoverishment, erosion and toxicity, thus affecting the flora, fauna, water quality and people in the region in many ways.

C.3.3.1 Soil fertility under selective logging

The selective logging conducted in central Amazonia by the BIONTE Project [BIONTE, 1997] resulted in an export of 65.3 kg N; 0.86 kg P, and 18.8 kg Ca per hectare of forest. Nutrient export was relatively modest, and at the soil (rather than the ecosystem) level was largely compensated by the addition of a new pool of nutrients from plant residues resulting from the logging. However, part of the clearings had micro-sites with newly exposed soil surface where fine litter decomposition rates were lower in the first months, and available soil nutrients were lost (at least temporarily) from the rooting zone through soil percolation (in the first weeks after logging) due to the addition of large amounts of new organic material and the absence of absorbing roots [Mello Ivo et al., 1996]. Additional losses of N may occur since logging can increase emissions of N₂O and NO from 30% to 350% depending upon soil conditions (Bustamante et al., this book), affected by changes in nutrient and water circulation, along with soil compaction by heavy machinery in skid trails and log storage decks. In the BIONTE Project, the micro-sites of the clearings, where
an accumulation of plant debris (branches, fallen canopies, etc) took place, which were usually located near the remaining forest edges, showed an increase in nutrient availability in the upper soil layers. This occurred because greater amounts of organic substrate, together with higher soil moisture, induced higher decomposition rates in the two first years, resulting in higher concentrations of available nutrients in the soil (especially Ca and Mg) after 1.5 years [BIONTE, 1997]. A parallel study confirmed that wood residues from selective logging caused increases in soil-nutrient availability (via decomposition), especially for the exchangeable bases K, Ca and Mg in the rainy season [Ferreira et al., 2001]. Decomposition of coarse woody debris (CWD, with diameter > 10 cm) released half of its C content in the first 5 years (19.9 Mg C ha\(^{-1}\)); the remaining content is released in another 20 years. However, CWD with diameter between 2 and 10 cm is decomposed in less than 5 years [Summers, 1998]. A high proportion of the nutrients is released in the first 4 years of wood decomposition, especially P and K: fluxes were four times higher for P and K, and three times higher for Mg in the logged plots (though still lower than fluxes via fine litter in intact forest) [Summers, 1998].

A recent study in an open tropical forest in Juruena, Mato Grosso (10°28'S; 58°30'W), at the southern fringe of Amazonia, showed that selective logging induced a strong increase in the mortality of palms (120-340%) and a decrease in annual production of CWD in the plots logged more recently: respectively only 1.1 Mg ha\(^{-1}\) yr\(^{-1}\) and 2.8 Mg ha\(^{-1}\) yr\(^{-1}\) in the plots logged 2 and 6-7 years previously, versus 5.3 Mg ha\(^{-1}\) yr\(^{-1}\) in a plot logged 11-12 years previously and 5.7 Mg ha\(^{-1}\) yr\(^{-1}\) in the undisturbed forest [Pauletto, 2006]. However, the main change observed was the sharp increase of 105% in the stocks and 37% in the volume of the CWD fraction with smaller diameters (2-10 cm). This fraction represented only 15-16% of total stock of CWD, but it stored 29% of total N, 35-40% of P, 18-20% of K, 37-42% of Ca, and 30-35% of Mg. There was an overall increase of 54-109% in nutrient contents of this fraction in the logged plots as compared to the undisturbed forest. The reason for such increases in the relative importance of the smaller fraction relative to the total stocks of nutrients in CWD is that nutrient concentrations in this fraction are much higher (especially for Ca and Mg) than in dead wood with larger diameters.

In the clearings produced during selective logging operations, the micro-sites under decomposing CWD had significantly higher concentrations of soil C and nutrients. This was particularly strikingly for Ca, which reached concentrations up to 590% higher in soil micro-sites under CWD.

However, it must be remembered that nutrient release from CWD is restricted to spatially limited and specific micro-sites within the clearings and there is a risk of not being absorbed by roots and, subsequently being removed from the rooting zone of the forest ecosystem [Mello-Ivo et al., 1996]. Similar removals were found in a parallel study of nutrients released by the decomposition of fine litter in the clearings: early in the wet season, five months after logging, concentrations of the exchangeable bases K\(^{+}\), Ca\(^{2+}\), Mg\(^{2+}\), and Na\(^{+}\) were from two to four fold higher than those in the forest control, but two months later, near the end of the wet season, strong decrease in the concentrations, probably due to leaching, especially for Mg, were observed in the clearings [Ferreira et al., 2006].
The general result of selective logging is, then, a strong redistribution of carbon and nutrients from the standing biomass, associated with the creation of micro-sites, with heavy additions to the soil surface of new organic materials such as fine or coarse litter, which can affect the natural forest regeneration, favoring either pioneer or climax species at different times after logging. Also, a short-term removal of some nutrients by soil percolation is observed in the cleared areas due to high release through decomposition of new plant residues and lack of absorbing roots in these micro-sites.

C.3.3.2 Forest fragmentation and soil fertility

One of the consequences of selective logging (and the majority of other human interventions on dense forests) is forest fragmentation, which also affects the cycles of mineral elements. Long-term studies carried out by the BDFFP - Biological Dynamics of Forest Fragments Project (INPA/Smithsonian Institution), near Manaus, found an increase in the stocks of both fine and coarse litter on the soil surface as a consequence of forest fragmentation, mostly as a result of edge effects [Nascimento and Laurance, 2004]. Annual production of fine litterfall, measured during three years, was 0.68 Mg ha\(^{-1}\) higher in forest areas suffering edge effects than in the forest interior, > 300 m from the edge (9.50 ± 0.23 vs. 8.82 ± 0.14 Mg ha\(^{-1}\)) [Vasconcelos and Luizão, 2004]. The concentrations of Ca in leaf litterfall were higher near the edges, probably because of strong soil Ca mobilization by pioneer species growing by the edges of forest fragments [Lucas et al., 1993]. Thus, forest fragmentation may also affect litter quality by favoring the recruitment of successional tree species at the expense of old-growth species [Laurance et al., 1998], as these two groups can differ strongly in nutrient contents and in decomposition rates [Mesquita et al., 1998].

These changes are closely related to variations in the abundance, species richness, and composition of many groups of soil invertebrates in response to edge effects or changes in vegetation cover [Didham, 1998]. There are indications that land-cover changes in Amazonia affect decomposition mainly through changes in plant species composition, which in turn affects litter quality in fragment edges, particularly in heavily disturbed edges where successional trees become dominant [Vasconcelos and Laurance, 2005].

C.3.3.3 Soil fertility under slash-and-burn practices

Despite the fact that impacts of selective logging, and especially forest fragmentation may be severe in some aspects or for some organisms and for the forests functioning over the long run, by far the greatest impacts on soil fertility and dynamics are caused by the generalized use of the slash-and-burn practice for clearing and preparing land for agriculture or pastures in Amazonia. Biomass burning is used for releasing nutrients stored in biomass to fertilize chemically-poor soils in most of the Amazon. The increase in nutrients incorporated into the soil by the ashes has been confirmed in many studies in the Amazon [ASB, 2002; Palm et al., 2005]. Seubert et al. [1977] calculated that the burning of the primary vegetation on an Ultisol in Peruvian lowland forest incorporated 67 kg ha\(^{-1}\) of N, 6 kg ha\(^{-1}\) of P, 38 kg ha\(^{-1}\) of K, 75 kg ha\(^{-1}\) of Ca and 16 kg ha\(^{-1}\) of Mg to the soil. The newly improved soil fertility situation may endure for several years although crop production generally decreases sharply years before the fertility declines [Seubert et al., 1977; Sanchez et al., 1983; Desjardins et al., 2000]. This implies that other factors are also involved, among which the partial gaseous losses of nutrients and the decrease of biological
activities involved in organic matter decomposition and mineralization. The burning used for releasing mineral nutrients from the biomass also represents a direct and considerable loss of nutrient pools by volatilization just at the beginning of the cultivation process [Kauffman et al., 1998; Kato et al., 2004; Palm et al., 2005]. A field experiment on biomass burning of a 7-year-old second growth in Pará state showed strong nutrient losses through fire: 98% of the C; 96% of the N; 47% of the P; 48% of the K; 35% of the Ca; 40% of the Mg; and 76% of the S are stored in burned material [Mackensen et al., 1996, SHIFT Project]. Further losses can be expected in the succeeding periods because forest conversion to agricultural use through the slash-and-burn system changes both the quantity and the quality of organic matter deposited on the soil surface, altering the soil moisture and temperature regimes, and, consequently, the biological processes that control litter decomposition and the dynamics of the soil organic matter. Biomass burning has the immediate effect of decreasing the biomass stock and nutrient pools of the system, and preventing the continuous and high input of carbon and nutrients to forest soils through litter fall, either fine or coarse. In central Amazonia, the annual input of carbon and nutrients to the soil surface of an upland forest on a plateau with clayey Oxisol was 3.9 Mg of C, 151 kg of N, 3 kg of P, 15 kg of K, 37 kg of Ca, and 14 kg of Mg through fine litterfall alone [Luizão, 1989]. This material is rapidly decomposed, generally within a year after litter fall, releasing carbon and nutrients to plant roots.

Without a litter-layer cover, the soil becomes exposed to heating from the direct sun and to the impact of raindrops, which increase compaction, erosion and consequent nutrient losses. In Rondônia, a 16-month study of the soil solution in an intact forest compared to a slash-and-burn area showed different results at distinct times [Piccolo et al., 1994]. In the first wet season, fluxes of the most abundant ions (Si$^{4+}$, NH$_4^+$, NO$_3^-$, Mg$^{2+}$, SO$_4^{2-}$, K$^+$, Ca$^{2+}$ and Mn$^{2+}$) were higher under burned forest than under intact forest; however, in the following dry and wet seasons the nutrient fluxes were higher under intact forest. This suggests a significant decline in the 25-cm soil solution nutrient concentrations in burned areas, after a short-lived nutrient enrichment.

After the installation of pastures, periodic burnings are common as a cultural practice for cleaning and refertilizing the land, representing further losses of carbon and nutrients from the crop system. A study carried out in a seven-year-old pasture in Apiaú, Roraima state, Brazil, during its third burn, showed that 210 days after burn the C stocks (20.2 Mg ha$^{-1}$) were significantly lower than in the pre-burn (26.0) in the top 20 cm of soil [Barbosa and Fearnside, 2003]. The authors point out that losses of C by burning and mineralization are higher than gains from humification of roots and plant remains but the cumulative imbalance only becomes apparent after a certain time (in this case, after 210 days).

Changes in soil physical conditions such as those described above have strong influences in nutrient recycling and availability to plants, likely because they affect biological transformations of nutrients such as N, P and S in soil. This was confirmed by a reanalysis of former data (e.g., Falesi, 1976) which stated that pastures improve soil fertility [Fearnside, 1980]. In fact, available P (P$_4$O$_6$ by North Carolina extraction: 0.05 N HCl and 0.025 N H$_2$SO$_4$), considered a critically limiting nutrient with initial P$_4$O$_6$ concentrations of 6.9 µg g$^{-1}$ in the forest, increased to 41.8 µg g$^{-1}$ after biomass burning in the Paragominas region, but after 5 years it decreased to a plateau of 4.6 µg g$^{-1}$ (one order of magnitude
lower), remaining there until 10 years of age in the pasture [Fearnside, 1980]. Even considering the unlikely possibility of P now stored in the grass biomass, a considerable net decrease in soil available concentrations was observed instead of enrichment. Probably some of the P in the soil was fixed in oxides while part was lost from rooting zone by slow percolation.

Forest clearing changes soil physical properties, which in turn affect chemical processes: it exposes soil and residual litter to rainfall events that can accelerate leaching, leading to removal of nutrients at rates higher than those for nutrient mineralization of dead roots and soil organic matter via microbial decomposition. In the long term, these changes can decrease soil nutrient status and soil organic matter concentrations [Woomer et al., 1999]. These severe impacts on soil functioning after forest conversion - generally involving the removal of the forest canopy - should be expected because they are a result of the replacement of a dense tropical forest, with high biological diversity and biomass, by a very simplified crop production system (generally a monoculture) or by pastures composed of a single grass species, usually exotic and implanted in areas with precarious infrastructure. Under these conditions, the basic mechanisms for the functioning of the native ecosystem are disrupted, including its efficient nutrient recycling process that is based on the stock and biological transformations of organic matter. Soil biota is seriously reduced and becomes dominated by a few species that are resistant to the impacts. New litter production is generally very low in the first years, and its quality also may be low, failing to provide adequate diversity of substrates and proper cover of the soil surface, which are essential for recovery of soil biodiversity needed to promote the nutrient recycling in the system. In addition, some nutrients, such as N and S, can be lost in high proportions in the initial and additional burnings, and can become limiting to the new system [Fernandes et al., 1997].

Most of the pastures in Brazilian Amazonia are planted with Brachiaria humidicola, which can generally grow well in the first years, covering the soil surface and can recover within a few years (5-7 years) the soil C content [Cerri et al., 1991]. However, the quality of the new organic matter that has originated from the grass is poor (with higher C:N ratio and lower mineralizable N and P contents) and does not allow the soil biota to act effectively in nutrient recycling, leading to nutritional deficiencies in the soil [Luizão et al., 1992b; Feigl et al., 1995]. Because pasture management is generally poor and inadequate in the region, the degradation factors of soil and/or pasture production evolve very quickly and can cause pasture abandonment in a few years.

The study of a pasture chronosequence from 2 to 13 years of age on clayey Oxisols (> 70% clay), all located within a 10-km radius and all on a flat plateau near Manaus, showed significant changes in soil C and N dynamics [Luizão et al., 1999]: (i) soil microbial biomass -C and soil N mineralization increased up to 5 years in pastures, followed by a gradual decrease, which was very accentuated after 8 years (Figure 2); (ii) pastures showed a higher proportion of soil N as N-NO₃ (while the forest control, as a result of nutrient conservation, had more N as N-NH₄), facilitating losses by leaching, denitrification or complexation in the soil; (iii) the low N mineralization rates corresponded to a decrease of organic N, leading to N deficiency in the soil in the oldest pastures; (iv) the oldest pastures (12-13 years old) also showed an accentuated decrease of soil organic C. This pattern agreed
with the one observed for fine root mass in the upper 0-20 cm soil layer [Luizão et al. 1999]: for roots with diameter 0.1 - 1 mm, biomass decreased from 1110 g m⁻² (five-year-old pasture) to 361 g m⁻² (7 years) and then to 243 g m⁻² (12 years). This is a dramatic change if compared to figures obtained for a nearby young and well-managed two-year-old pasture on similar soils where potentially renewable fine root (0.1 - 1 mm in diameter) mass was 8.9 Mg ha⁻¹ yr⁻¹, versus 5.1 Mg ha⁻¹ yr⁻¹ in the forest, for the upper 20 cm of soil [Luizão et al., 1992a]. In the chronosequence, the C:N ratio in roots increased with pasture age, which also indicates a decrease in the nutritional quality of organic matter derived from roots.

The study of two pasture chronosequences in Santarém, Pará, one on clayey and the other on sandy soil [Asner et al., 2004b], showed that the C stocks in aboveground biomass and soil decrease with pasture age. Plant biomass decreases are probably related to lower C, available-P and exchangeable-Ca concentration in the soil; in addition, ecosystem-P decreases with pasture age.

Another study of pasture chronosequence, also in Santarém [Townsend et al., 2002], showed significant losses of organic matter and soil total-P with pasture age in soils that were already deficient in P; however, losses occurred for the inorganic-P fractions while organic-P forms remained constant or even increased, despite losses of organic matter. The observed losses were attributed to changes in soil micro-organism communities.

Losses of soil N to the atmosphere in young pastures can be substantial, as demonstrated by a comparative study of N₂O fluxes in forest, burned areas and young pasture, all adjacent and on clayey Oxisol (> 70% of clay), in Manaus [Luizão et al., 1989]. The N₂O annual flux increased three fold in pasture as compared to a forest control (which, in the tropics, is considered to be naturally high; Davidson et al., 2004): 1.9 kg ha⁻¹ yr⁻¹ in the forest and burned forest against 5.7 kg ha⁻¹ yr⁻¹ in the young pasture. There was a strong seasonal effect: in the dry season, N₂O fluxes were similar in forest and pasture, but in the wet season fluxes were from three to five fold higher in pasture (> 10 ng cm⁻² h⁻¹ in March and April, the rainiest months). Later measurements in neighboring pastures showed that older pastures decrease the emissions of nitric and nitrous gases as well as N mineralization rates and soil available-P; thus, aging pastures decrease the concentrations of soil nitrate (due to denitrification, nitrate leaching, or soil acidification) and increase the relative concentrations of ammonium [Luizão et al., 1999]. Lower mineralization and nitrification rates are related to decreases in demineralization potential and associate lower rates of N₂O and NOₓ emissions from soils in aging pastures [Bustamante et al., this book].

C.3. 4. Nutrient management regimes in use in agriculture and forestry

Agroforestry systems (AFS) are often mentioned as a type of sustainable agriculture that is appropriate for the edapho-climatic conditions of Amazonia, because the species selection and arrangement can produce a nutrient management regime suitable for keeping or even improving soil fertility [Fernandes et al., 1997]. However, long-term studies on the sustainability of this land use do not exist in Amazonia.
Small-holders in Nova Califórnia, Rondônia, Brazil, are conducting the first large and organized experiment on the sustainability of such systems (now spanning 19 years), implemented after the usual land clearing through slash-and-burn of native dense forest. The systems are based only on three regionally important fruits: “cupuaçu” \((Theobroma grandiflorum,\) Sterculiaceae), peach palm \((Bactris gasipaes,\) Palmae) and Brazil nut \((Bertholletia excelsa,\) Lecythidaceae). All farmers also have pasture plots on their farms. Most of the 200 farmers involved (in the RECA association –Mixed and Dense Economic Reforestation Project) had no leguminous cover plants included in their AFS in the first 10-12 years, when an evaluation was made, comparing AFS to pastures of similar ages and control forest plots [Alfaia et al., 2004]. The AFS soils maintained their improved chemical conditions derived from biomass burning, especially the increased levels of exchangeable Ca and Mg and the reduction of exchangeable Al, while maintaining stable levels of organic C, even when compared to adjacent primary forest soils. In contrast, the improved soil conditions in the pastures were transitory and short lived: after the first years, low soil pH, high level of Al and low levels of exchangeable bases returned. However, K and P fell to extremely low levels in the AFS. This reduction was reasonably attributed to nutrient exports by consecutive harvests of \(T.\) grandiflorum and \(B.\) gasipaes fruits, since K is one of the most important nutrients for the yields of these fruit species. Considering the high cost of mineral fertilizers in the region and the soil characteristics that favor mineral nutrient leaching, a simple solution adopted by farmers was to stop burning \(T.\) grandiflorum fruit rinds, rich in K (and also in N and P), grinding them and adding the product on the AFS soils. Additionally, some of the farmers decided to introduce leguminous cover plants (mostly \(Pueraria phaseoloides\)) in their plots, and apparently the nutrient balance problem is now under control for the RECA farmers [S. Alfaia, pers. comm., 2007].

In central Amazonia, the efficiency of a multi-strata AFS for organic matter recycling was studied at full (FF) and low (LF) fertilization levels, compared to natural fallow [Uguen, 2001; Schroth et al., 2002]. (Full initial fertilization level stands for applications of 38-215 g.plant\(^{-1}.\)year\(^{-1}\) of N, 18-90 g.plant\(^{-1}.\)year\(^{-1}\) of P, 39-331 g.plant\(^{-1}.\)year\(^{-1}\) of K, and 0.2-1.5 kg of dolomitic lime to each tree; low fertilization level corresponded to application of only 30% of the full levels of fertilizer and lime). The organic and nutrient inputs through litterfall and prunings were assessed during 1 year in a 5 year-old AFS composed of four tree species: Brazil nut \((Bertholletia excelsa,\) “cupuaçu” \((Theobroma grandiflorum),\) peach palm \((Bactris gasipaes,\) annatto \((Bixa orellana).\) The soil was heterogeneously covered by a legume cover crop of \(Pueraria phaseoloides\). Large differences in leaf litter nutrient concentrations were found between the four tree species [Uguen, 2001]. Annatto leaf litter had the highest nutrient concentrations for all measured macronutrients while Brazil nut and “cupuaçu” had the lowest N, P and K concentrations.

Total organic inputs were lower in the AFS (4.56 and 3.59 Mg ha\(^{-1}\) in FF and LF, respectively) than in the fallow (5.1 Mg ha\(^{-1}\)). Fine litterfall was relatively low (1.6 and 1.5 Mg ha\(^{-1}\) in FF and LF respectively) in the AFS, and pruning biomass accounted for more than 50% of the total organic inputs (2.95 and 2.12 Mg ha\(^{-1}\) in FF and LF, respectively). Chemical fertilization had no significant effect on litterfall but significantly increased pruning biomass, especially for annatto, from which the main contribution to total nutrient inputs came. Ca and Mg inputs were enhanced for all species and P inputs were
increased only for peach palm and annatto. In a neighboring similar AFS, *Pueraria* (used as a legume cover crop) had a noticeable high litter production (1300 g.m\(^{-2}\)), showing its importance to the nutrient balance and recycling in the AFS [Uguen, 2001]. Chemical fertilization, which increased pruning biomass, also enhanced *Pueraria* growth and, thus, the cover crop litterfall.

There were large differences in nutrient inputs between species. For all nutrients except calcium, “cupuaçu” litterfall represented the lowest nutrient input while annatto had the highest contribution to total nutrient inputs. Major nutrient inputs in the agroforestry system came from prunings: about two thirds of the N and P, and 80% of the K, whereas 60% of Mg and 50% of the Ca came from litter [Uguen, 2001; Schroth et al., 2002]. Thus, it appears that the spatial alternation of species with high and low nutrient cycling could favor a good soil cover and reduce nutrient leaching from prunings. Additionally, the presence of a cover crop such as *Pueraria*, producing abundant and high-quality litterfall, seems to have high importance for soil fertility rehabilitation under AFS.

At the CPAA/Embrapa Experimental Station, near Manaus, different formulations of AFS were tested on abandoned pasturelands, after an initial slash-and-burn of young (4-6-year old) fallow and a light P fertilization (20 kg ha\(^{-1}\)), together with 1.5 Mg ha\(^{-1}\) of lime, applied only at the onset of the experiment [Fernandes et al., 1999]. In the first years, the system, which was planned to reproduce an improved pasture (*Brachiaria brizantha* mixed with the legume *Desmodium ovalifolium*) planted together with commercial timber species (*Schilozobium amazonicum* and *Swietenia macrophylla*), showed the best soil cover and conditions for nutrient recycling and soil biological activities [Luizão et al., 2006]. However, a few years later, the fast tree growth in the two AFS with fruit, palms and timber trees showed better conditions for soil nutrient recycling through higher litter production together with higher litter quality and green manures coming from prunings of *Inga edulis* (planted in the tree lines in the improved pasture and in the multi-strata AFS) and *Gliricidia sepium* (used as green hedge in all AFS). When the palm-based and multi-strata AFS were 6-7 years old, their fine-litter production corresponded to only 25-30% of the amount produced by the adjacent second growth (used as a control). However, most of the AFS species had higher nutrient contents in litterfall, and the AFS received regular additions of green manures, originated from periodic prunings of the leguminous plants *Gliricidia sepium* (from the green hedges) and *Inga edulis* (planted in the tree rows within the AFS); both of these were scattered as mulch on the surface of the AFS soil. Because of the pruning addition, a similar or even better nutrient balance could be observed as compared to the second growth as early as 6-7 years of age (Table 4).

These results suggest that a young agroforestry system, especially if it is not dense enough, requires additional sources of organic matter, e.g., from cover crops or green manures, to replenish soil organic matter and to balance nutrient cycling, thus allowing early and proper nutrient cycling, which is essential for optimal crop development.

Traditional smallholders cultivating rice, beans, maize or, most frequently, manioc (cassava), after slash-and-burn cleaning of the land, usually do not apply any inorganic fertilizer to the soil, and very seldom do they use organic manures in their fields in the
Brazilian Amazon. The most common practice is to burn any organic residues generated on the property, using the resulting ashes to fertilize the soil.

On the other hand, the rapidly expanding area of large-scale industrial cultivation of soybeans relies on heavy doses of fertilizers and pesticides to guarantee good yields in Amazonia [Fearnside, 2001]. Significant applications of fertilizers and pesticides are also required by sugarcane to overcome climatic and edaphic limitations. Sugarcane plantations are currently present only in a small portion of the Amazonian biome but have the potential to spread to many parts of the region in the next few years [Smeraldi and May, 2008].

C.3.4. Carbon sequestration potential in agroforestry systems converted from degraded pasture

Agroforestry is a possible option not only for carbon sequestration but also to increase the value of previously cleared forest land in the humid tropics [Fujisaka and White, 1998]. Wherever agroforestry succeeds in maintaining soil fertility at a satisfactory level and increases farmers’ incomes, additional clearing of primary forest and accompanying carbon emissions are drastically reduced. When established on degraded soils, timber and tree crops in these systems sequester carbon in the biomass and soil and also provide firewood and charcoal as offsets for fossil fuel. On the other hand, when agroforestry systems or tree crop plantations are established on previously cleared fallow or secondary forest land, carbon is released from the fallow vegetation that would have accumulated carbon in biomass and litter. Instead, establishment of pioneer trees, timber trees, annual crops and sometimes cover crops affect the ecosystem C budget through soil management, fertilizer application and suppression of spontaneous vegetation through weeding [Schroth et al., 2002].

Few data are available on the rate of carbon sequestration through agroforestry in the Amazon. A mean rate of 2.7 Mg C ha⁻¹ yr⁻¹ over 25-30 years is supported by the literature elsewhere, in which the values vary from 0.5 to 3.8 Mg C ha⁻¹ yr⁻¹. Woomer et al. [1999], measuring total system carbon in chronosequences in Brazilian Amazonia (Rondônia and Acre states), Cameroon, Indonesia and Peru, reported that agroforestry systems sequestered about 3.3 Mg C ha⁻¹ yr⁻¹ in soils and vegetation. Sampson et al. [2000, p. 199] mentioned a range of 0.5 to 1.8 Mg C ha⁻¹ yr⁻¹ of carbon accumulation for agroforest management in the tropics. McCaffery et al. [2002] found that agroforestry systems based on native fruits and palms, planted on a severely degraded pasture in the Central Amazon accumulated up to 33 Mg C ha⁻¹ in the aboveground biomass after 12 years of management. Biomass in degraded pasture was 9 Mg C ha⁻¹, indicating a net C uptake of 2 Mg C ha⁻¹ yr⁻¹. For the same systems, Rondon et al. [2000] reported soil carbon stocks (to 1 m depth) under agroforestry systems of 120 Mg C ha⁻¹ as compared with the degraded pasture soil which stored 110 Mg C ha⁻¹. This resulted in soil carbon accumulation rates of 0.83 Mg C ha⁻¹ yr⁻¹. Schroth et al. [2002] reported that trees planted as monocultures accumulated carbon at lower rates, i.e. 1.0 Mg ha⁻¹ yr⁻¹ for *Citrus*, 1.3 Mg ha⁻¹ yr⁻¹ for "cupuaçu" (*Theobroma grandiflorum*) and 2.5 Mg ha⁻¹ yr⁻¹ for rubber trees (*Hevea brasiliensis*). However, multistrata agroforestry favors carbon sequestration more than monocultures. A
fast-growing system in central Amazonia accumulated 3.8 Mg ha\(^{-1}\) yr\(^{-1}\) of carbon in the full-fertilization treatment (as described in section C.3.4) and 3.0 Mg ha\(^{-1}\) yr\(^{-1}\) of carbon under low fertilization (only 30% of the full levels of fertilizer and lime). These high rates compared to most monocultures were due to a relatively high tree density and to the association of the smaller tree crops, such as “cupuaçu” and peach palm (*Bactris gasipaes*) for heart-of-palm production, with larger and faster-growing trees, *i.e.* rubber and Brazil nut trees (*Bertholletia excelsa*).

Schroth and colleagues also observed that in all of the investigated plantation systems there was more than twice as much carbon in the soil organic matter than in the biomass and litter combined. Changes in the soil organic-matter stocks could, therefore, be of crucial importance to evaluate the effect of land-use transformations on the carbon budget. However, Schroth and colleagues observed no effects of vegetation types and plant species on the organic-matter stocks of the soil profile to 2-m depth, although carbon content of the topsoil was affected, being stored more superficially. These authors proposed two possible explanations for this trend. First, the conversion of primary forest to different tree crop plantations may have affected the distribution of carbon in the soil, but not its total quantity. Such changes may occur through an altered distribution of root mass in the soil profile, or through differences in the abundance and activity of burrowing soil fauna among vegetation types and plant species. Second, the total carbon stock in the soil to 2-m depth may be less sensitive than the carbon content of the topsoil as a measure for soil organic matter loss over a relatively short time period [Schroth et al., 2002]. The authors also observed that the tree crops with low litter quality (*e.g.*, “cupuaçu” and Brazil nut) restored and maintained organic matter levels in the topsoil comparable to those in the primary forest, even when they were grown in association with tree and cover crops that produced easily decomposable litter. When integrated into multistraata agroforestry systems, such tree crops reduce soil organic matter loss. Thus, these systems must be established on sites with low standing biomass (*e.g.*, degraded pastures or other degraded soils) while preserving vigorously growing secondary forests. However, Amazonian pastures are often prone to topsoil compaction and erosion [Fearnsidce, 1985], with adverse effects on growth and yields of tree crops.

Considering the complexities involved, it is apparent that the estimate of 6.5 to 49.4 Tg C yr\(^{-1}\) potential gain from conversion of degraded pasture to agroforestry within a period of 25-30 years is rather crude. This estimate is obtained by multiplying 13 Mha of degraded pasture area by the soil + biomass carbon accumulation range of 0.5 to 3.8 Mg C ha\(^{-1}\) yr\(^{-1}\). This estimate is based on the assumption that the entire area under degraded pasture is converted to agroforestry. Furthermore, soil organic carbon sequestration in agroforestry is affected by species. Though the total area in Amazonia dedicated to agroforestry is relatively small compared to other management systems, and there are severe limitations that restrict the potential of converting degraded pastures into agroforestry systems [Fearnsidce, 1995], agroforests remains as a viable alternative land use for the region, and the area under these systems has been steadily increasing since the mid 1980s.

C.3.5. Promising alternatives to slash-and-burn agriculture
Lack of proper knowledge of soils and of crop or pasture management generally leads to short-lived crops (typically 2-3 years) or pasture (mostly from 5 to 10 years) systems, after which they are abandoned and colonized by spontaneous second growth (“capoeira”). After a fallow period, a new crop or pasture can be established, again following slash-and-burn for clearing the land. In the abandoned areas, the development of second-growth biomass and plant diversity will depend on the previous use of the area (clearing methods, intensity of use and management) and on the size of the area cultivated [Uhl et al., 1988; Moran et al., 2000; Mesquita et al., 2001]. However, even an area subjected to only moderate use may only recover partly the original plant diversity and biomass, even after long periods of time [Moran et al., 2000]. A recent study from the Museu Emilio Goeldi (Belém, Pará) in association with the LBA Project, in the Bragantine Zone (the oldest colonization zone in Amazonia), showed that 70-year-old second growth had recovered only 20-35% of the number of tree species present in the forest control, and 50-60% of the original C stocks [I.C. G. Vieira, pers. comm., 2008]. Another study done in the same region showed that the nitrogen (N) cycle takes 70 years to recover to a condition similar to that of primary forest [Davidson et al., 2007]. Thus, one can suppose that cycles of organic matter and nutrients in these second growths are also distinct from those in the original forest and that the pace and dynamics of tree growth are affected by shortages of key nutrients [Markewitz et al., 2004]. A fertilizer experiment in 6-year-old second growth in Paragominas showed that additions of N and of N + P produced an increase in the biomass of woody species while the addition of P increased only the biomass of herbs and grasses. Thus, regenerating forest biomass is limited by N on old pasturelands that have been subjected to successive burnings [Davidson et al., 2004]. In a fertilizing experiment in the Bragantina region, it was shown that secondary regrowth was mainly P-limited, and to a lesser extent N-limited [Gebring et al., 1999]. In a chronosequence of second growth up to 14 years old in central Amazonia, it was found that soils up to 45 cm depth accumulated N but lost available-P, suggesting possible limitations by P (and Ca) for second growth [Feldpausch et al., 2004].

In order to prevent new cycles of deforestation, short-lived cropping and/or pasture systems and consequent land abandonment, alternative land use and utilization of already deforested areas have been proposed in recent years [Vieira et al., 1993; Fearnside, 1997a]. These large areas of abandoned land could be re-utilized through different alternative techniques: 1. enrichment of second growth (especially with introduction of valuable timber and/or fruit species); 2. creation of new cropping systems using plant biomass without burning; 3. use of agroforestry systems (AFS).

Use of alternatives to slash-and-burn agriculture in the Brazilian Amazonia is relatively infrequent and many times these practices are not known outside a specific site where they are applied. However, they do exist and could be adopted more widely if properly advertised and tested. For instance, the settlers (‘caboclos’) in the Arroz Cru settlement (Xingu region), coming from Xingu river islands, received plots of 100 ha for cultivation: they used 4-year fallows for mixed plantations, obtained good and continued production of manioc, corn, beans, and bananas [Silva-Forsberg and Fearnside, 1995]. This could be a potentially suitable system for cultivating second growth areas, with no need for new forest slash-and-burn.
In Central America, the “frijol tapado” is a slash/mulch bean (*Phaseolus vulgaris* L.) production system that has been practiced since pre-Columbian times in the region now known as Costa Rica [Meléndez et al., 1999]. Under slash/mulch systems, crops are sown in direct association with *in situ* slashed vegetative mulches. They are still common throughout humid tropical Central America and northern South America, where high rainfall prevents burning and induces rapid decomposition and nutrient mineralization. By providing continuous ground cover, they are particularly well adapted to the steep slopes that account for most of the land area of Central America. Traditionally, the “frijol tapado” system allowed a fallow period of 3 years or longer but presently its fallow period can be as short as 9 months, or even shorter, raising serious concerns about its sustainability [Meléndez et al., 1999].

In northeast Pará, Brazil, the Tipitamba Project (Embrapa-Amazônia Oriental/SHIFT/LBA Projects) used early fallow enrichment and a chop-and-mulch procedure, eliminating the use of fire for land preparation, then cultivating annual crops successfully for several years [Sá et al., 2007]. The experiment consisted of the initial enrichment of smallholder fallows, on the occasion of land abandonment, with fast-growing legumes (*Acacia* sp. or other tree species) in order to increase and maintain biomass production and to diversify the chemical quality of the organic material produced. After a short period of fallow (reduced to approximately three years instead of the minimum seven years that is usual in the region), land preparation for new cultivation without use of fire was tested (meaning, fire was not used in the process). Plant biomass of the improved fallow was cut and ground using a large grinder coupled to a tractor, and the resulting organic material was deposited on the soil surface as mulch, followed by planting beans, maize, manioc, passion fruit, etc. Despite some problems (and lack of specific incentives to farmers), the system was found to be economically more efficient than the traditional slash-and-burn procedure after five years. This coincided with the period when the soil fauna (especially the macro-invertebrates) is recomposed to levels similar to those found in nearby forests [T. Sá, pers. comm., 2008]. Even though strong increases in CH₄ emissions were found in the first two years after mulching, the CO₂-equivalent emissions calculated for the entire crop system were at least five times lower than in the traditional slash-and-burn process [Davidson et al., 2008]. Although a moderate NPK fertilization was required in the chop-and-mulch system for the first year, available-P and mineral-N (NH₄ and NO₃) increased significantly by the end of the second year, contrasting with a sharp decline in the slash-and-burn plots. These results indicate that, if proper incentives and technical supervision are given in Amazonia, it is possible to prevent, or at least reduce, the use of fire by farmers during land preparation for cropping. This chop-and-mulch experiment has now been replicated in the states of Acre, Amapá, Amazonas, Maranhão, Rondônia and Roraima, using annual, semi-perennial (black pepper, passion fruit) species, and pastures.

Also in Brazilian Amazonia, as part of the “Crop production without burning” Project (Viver, Produzir e Preservar Foundation), at the end of the 1990s small farmers used diversified cropping systems mixing annual and perennial species, adopting principles of the agro-ecological transition now included in the Pro-Ambiente Program in Brazil [Sá et al., 2007]. Well-established farmers with and sufficient economic resources may also be able to adopt the ‘Bragantine System’ of continuous cropping of varied and mixed species.
under rotation and consortium systems, using mulching and keeping the soil cultivated and covered throughout the year [Cravo et al., 2005]. This management system requires liming and fertilization (P and micronutrients), at least temporarily.

Direct and fast recovery of soil properties in abandoned or degraded lands, used for agriculture and/or pastures in Amazonia has been tried in some experiments in Brazil. For instance, in central Amazonia, the use of two different cover crops was tested for rehabilitating soil structure and function. The leguminous species Pueraria phaseoloides and Desmodium ovalifolium were planted as cover crops together with an Elaeis guineensis (oil palm) plantation, in order to reverse the compaction effects of both mechanized and manual deforestation for land clearing [Grimaldi et al., 1993]. After two years, the soils cultivated with oil palm and a cover of Desmodium recovered a bimodal pattern of soil-pore distribution similar to that under the intact forest, with pores > 0.1 µm in the plots deforested manually, but not in the plots deforested with machinery. When the soil cover was Pueraria, in both situations soil macroporosity was recovered after two years due to strong root development plus the biological activity favored by Pueraria.

In Amazonia, some traditional agroforestry systems have been shown to be sustainable as parts of indigenous [Hecht et al., 1989] and small-holder [Jong, 1996] agriculture and are now considered as a promising and sustainable land use in Amazonia that is especially suitable for degraded areas [Fernandes et al., 1997]. Multi-strata AFS are expected to be more sustainable land uses than are annual crops because they include various long-lived woody species and have a larger canopy and a more complete ground cover, thus limiting nutrient losses through runoff and leaching, being as efficient as or even better than natural fallow for soil rehabilitation [Young, 1997]. They can maintain soil organic matter and biological activity at satisfactory levels for soil fertility and to restore degraded soil [Ewel, 1986; Young, 1997]. Depending on the type and diversity of the new ecosystem, the soil microbial biomass, which shows surprising capacity for recovery, may return to previous forest levels within a few years [Woomer et al., 1999; Barros et al., 2003]. In Manaus, soil microbial biomass was measured in four different five-year-old agroforestry systems [Tapia-Coral et al., 1999]. All the agroforestry systems had from amounts of microbial biomass 1.3 to 3.1 higher than the ten-year-old second growth system used as a control, and the highest values were found in the most diverse and dense multi-strata system.

In spite of some limits to expanding agroforestry systems to large productive systems, mainly due to market constraints [Fearnside, 1995], they seem to be good alternatives for smallholders, as illustrated by some successful experiments carried out in the region. In Brazil, the RECA system in Rondonia is an example of success, after the farmers overcame the market constraints on selling their crops. The Manaus experiment carried out by Embrapa/CPAA, using four different formulations of agroforestry systems, has also yielded good results and helped to understand processes involved as well as to establish useful principles to be applied in other agroforestry systems [Fernandes et al., 1997; Luizão et al., 2006].

The evolution of the agroforestry system in Manaus showed that: (i) in the two first years, the systems with improved pastures, with a fast and efficient establishment of soil cover (by a mixture of Brachiaria humidicola grass and the leguminous cover crop Desmodium
ovalifolium) allowed a soil organic cover and a better recovery of soil fauna diversity [Barros et al., 2001]; (ii) after four years, there was a decrease in concentrations of all soil cations due to nutrient export through crops, and tree growth in the agroforestry system; (iii) after five years, the most diversified agroforestry system had higher diversity of taxonomic groups and higher biomass of soil macro-invertebrates, which were also related to the quantity and quality (C/N, C/P, N/P, P) of litter produced and deposited on the soil surface [Tapia-Coral et al., 1999]; (iv) after, 6-7 years, the agroforestry system composed of timber and fruit trees, plus a leguminous green hedge, produced fine litterfall equivalent to only 25-30% of the total litterfall in the 10 year old second growth (taken as a control), but with higher nutrient contents. Together with the green manures from the Gliricidia sepium hedge and the Inga edulis planted in the AFS, this input produced a balance of nutrients in the agroforestry system (see Table 4 above); (v) after ten years, the agroforestry system presented similar or higher C and nutrient stocks (highly variable, depending upon the composition of tree species in the agroforestry system) than the second growth, indicating the high potential of AFS as C sink and for recovering the ecosystem nutrient recycling mechanisms; (vi) after ten years, the agroforestry system also had better soil structure (especially the macro-porosity, produced by soil macro-invertebrates and roots and essential for water movement and availability in the soil) than the second growth, indicating the recovery of another ecosystem service in the agroforestry system: the water recycling in the system [Cortes-Tarra, 2003]. Among the tree species in the agroforestry system, Inga edulis had the highest biomass of soil macro-invertebrates per hectare and the best soil macro-porosity in the soil directly influenced by the trees, probably because of the high-quality litter, which is associated with the production of highly energetic root and branch exudates; (vii) after 12 years, the agroforestry system had soil P and cation concentrations that were higher than second growth soils, which can at least partly be attributed to better litter quality and pruning inputs to the soil (Table 5).

Forest plantations with native tree species are still surprisingly rare in Brazilian Amazonia, since they represent a natural vocation for areas that have been deforested in the region. Such plantations could be used for: (i) enrichment of mature or secondary forest with valuable timber species [Yared, 1996]; (ii) timber or charcoal production; (iii) rehabilitating degraded lands [Higuchi et al., 1998 Jacaranda Project]. Only a few examples are known for all three cases, and generally they lack evaluations of soil nutrients. Forest plantation in Amazonia should have a mixed species approach in order to prevent problems with diseases, which easily spread through a monoculture, or nutrient shortages in the soil (all trees demanding the same nutrient at the same time and position in soil), or problems with the timing for the nutrient-release rates. One example is a Brazil nut plantation in Itacoatiara, Amazonas, which produces slowly decomposing litter (Table 6). On the other hand, its litter covers the soil surface, thereby improving moisture and other soil conditions.
A few experiments have been done on the rehabilitation of mining sites in Amazonia, such as the recent plantation of climax and pioneer native species at the oil extraction site in Uruçu, Amazonas (CT-Petro experiments). Another example is from Trombetas, Pará, where diversified forest plantations were installed to rehabilitate soils mined for bauxite. To rehabilitate these badly damaged sites, reforestation programs using native tree species have been implemented by the “Systems of Production and Studies of Degraded Land, Reconstitution and Rehabilitation of Forest Ecosystems” project (UNESCO/EEC-Economic European Community/GTZ-Deutsche Gesellschaft für Technische Zusammenarbeit/MRN-Mineração Rio do Norte). At the Saracá Mine in the eastern part of Pará, reforestation treatments (topsoil replacement versus no topsoil replacement; natural regeneration versus reforestation) were compared in treatment plots including: a 1-year-old reforested plot where no topsoil replacement occurred; two 11-year-old plots where the topsoil had been replaced (one reforested and the other with vegetation allowed to regenerate naturally); and a primary forest that was used as a control. The 1-year-old plantations without topsoil replacement always had very low concentrations of nutrients in the soil, particularly for P and K (Table 7). Soil exchangeable Ca and Mg were higher in the two 11-year-old treatments [Costa et al., 2002].

Thus, reforestation with native tree species (and to a lesser extent, the naturally regenerated forest) allows the recovery of the soil litter cover and microclimate within a few years. This, in turn, leads to the recovery of soil microbial activity and the decomposition process. Topsoil nutrient availability is improved in one to two decades due to the mobilization of nutrients from the soil, particularly Ca and Mg, promoted by planted or pioneer species. The organic carbon content of the soil is the slowest to recover and may therefore act as the ultimate indicator of the rehabilitation process. In mining sites, topsoil replacement prior to planting native tree species can speed up the soil rehabilitation process [Costa et al., 2002].

Studies on natural regeneration in the central Amazonia have shown that pioneer species such as Cecropia sp. and some planted agroforestry species produce leaf litter that is rich in base cations, particularly Ca [Lucas et al., 1993; Gallardo-Ordinola, 1999; Tapia-Coral et al., 2005]. These species appear to have a high capacity to extract Ca and Mg from soil compounds, incorporating these nutrients as biomass, and releasing them to the soil surface in a more available form as plant litter.

Secondary forest associated with agriculture in Amazonia follows a clear pattern of development. During pasture use, burning and weeding delay succession, but the forest begins to regenerate once the field is abandoned. Secondary vegetation is established through four main processes: regeneration of remnant individuals, germination from the soil seed bank, sprouting from cut or crushed roots and stems, and dispersal and migration of seeds from other areas [Tucker et al., 1998]. Variations in the speed of forest regrowth are evident across regions and along a soil-fertility gradient in Brazilian Amazonia. The rate of forest succession is determined by several factors. Original floristic composition, neighboring vegetation, and soil fertility and texture may affect regrowth. In addition, farmers’ land-use decisions (such as clearing size, clearing procedures, crops planted, frequency and duration of use) influence tree establishment and path of secondary
succession [Moran et al., 2000; Mesquita et al., 2001]. At the regional scale, soil fertility and land-use history are the critical factors influencing forest regrowth [Tucker et al., 1998].

In Amazonia, secondary forests have high rates of regeneration following slash-and-burn agriculture but slower regeneration after abandonment of degraded pasture [Fearnside and Guimarães, 1996; Mesquita et al., 2001]. Brown and Lugo [1990] reported that abandoned agricultural lands reverting to forests accumulated carbon at rates proportional to the initial forest biomass. Rates ranged from about 1.5 Mg C ha\(^{-1}\) yr\(^{-1}\) in forests with initial biomass of < 100 Mg C ha\(^{-1}\) to about 5.5 Mg C ha\(^{-1}\) yr\(^{-1}\) for forests with biomass of > 190 Mg C ha\(^{-1}\). Woomer et al. [1999] observed a rate of 6.2 ± 1.3 Mg C ha\(^{-1}\) yr\(^{-1}\) of carbon sequestration in secondary forest regrowth in agricultural fallows in Brazilian Amazonia (Rondônia and Acre). Sampson et al. [2000, p. 200] suggested a range of carbon accumulation from 3.1 to 4.6 Mg C ha\(^{-1}\) yr\(^{-1}\) for tropical regions over 40 years. Schroth et al. [2002] reported that secondary forest on an infertile upland soil in Manaus, central Amazonia, accumulated carbon in above- and below-ground biomass and litter at a rate of about 4 Mg ha\(^{-1}\) yr\(^{-1}\). The rate of accumulation in aboveground biomass reported by Nepstad et al. [2001] ranged from 2.5 to 5 Mg C ha\(^{-1}\) yr\(^{-1}\) for a 20-year-old secondary forest in Pará, eastern Amazonia. In Paragominas, Pará, a 19-year-old secondary forest accumulated 20% of the forest biomass at a rate of 9 kg C ha\(^{-1}\) yr\(^{-1}\) [Markewitz et al., 2004]. In Manaus, Feldpausch et al. [2004] reported C accumulation of 128 Mg C ha\(^{-1}\) for a 12-year-old secondary forest dominated by Vismia spp. regenerated on an abandoned and severely degraded pasture. However, soil degradation under typically managed pasture can severely slow the subsequent regeneration of secondary forest [Uhl et al., 1988; Fearnside and Guimarães, 1996].

Potential carbon sequestration when degraded pastures in the Brazilian Amazon are abandoned for secondary forest regrowth, were calculated by multiplying 13 Mha (degraded pasture area) by the accumulation range of 1.5 to 5.5 Mg C ha\(^{-1}\) yr\(^{-1}\) or 19.5 to 71.5 Tg C yr\(^{-1}\). This potential for carbon sequestration accounts for carbon in soil + aboveground biomass. As of 1986, secondary forests covered 30% of the area of the Brazilian Amazon that had been cleared by the 1980s [Houghton et al., 2000]. It should be noted that less secondary forest is present in the region now than was the case in 1986, the year of Landsat satellite imagery studied by D. Skole that is believed to provide the basis of Houghton et al.’s [2000] assumption that 30% of the region was under secondary forest. As of 2002, Landsat satellite imagery indicated 16.1 million hectares of secondary forest were present in Brazilian Amazonia, or 19% of the deforested area [Neeff et al., 2006]. If this 16.1 million-hectare area were to remain abandoned, it could sequester carbon in soil and plant biomass. Thus, multiplying the same accumulation range values (1.5 to 5.5 Mg C ha\(^{-1}\) yr\(^{-1}\)) there would be an additional sequestration of 24 to 86 Tg C yr\(^{-1}\). Therefore, potential soil + biomass carbon sequestration in the Brazilian Amazon due to secondary forest regrowth is between 43.5 and 157.5 Tg C yr\(^{-1}\).

By fixing carbon in biomass and gradually restoring soil physical and chemical properties, forests that develop on abandoned land also counteract many of the deleterious impacts of forest conversion to agriculture and cattle pasture. These forests play an important role in the regional carbon budget, as they re-assimilate part of the carbon that was released upon cutting and burning of the original forest vegetation. Secondary forests allow the
expansion of native plant and animal populations from mature forest remnants back into agricultural landscapes [Nepstad et al., 2001]; they restore hydrological functions performed by mature forests and reduce the flammability of agricultural landscapes; they also transfer nutrients from the soil to living biomass, thereby reducing the potential losses of nutrients from the land through leaching and erosion.

In Paragominas, Pará, degraded and managed pastures, as well as 19-year-old secondary forest showed similar or higher contents of nutrients, especially basic cations (Ca, Mg) in relation to the forest [Markewitz et al., 2004]. In part, this can be attributed to the long-term effect of biomass burning and the initial release of nutrients such as Ca from the ashes. The basic cations K, Ca and Mg were mostly retained in soil after 19 years of second growth, although soil was slowly returning to an acidic condition [Markewitz et al., 2004]. Reaccumulation of macronutrients in vegetation in the 19-year-old second growth was equivalent to 20% of the N, 21% of the P, 42% of the K, 50% of the Ca, and 27% of the Mg of the original forest. In the degraded pastures, the reaccumulation was much smaller, as expected: 2% of the N, 4% of the P, 15% of the K, 11% of the Ca, and 6% of the Mg of the original forest. Both managed and degraded pastures only possessed about 2% of original N of the forest biomass, implying that N was not transferred to the soil by land-use conversion; rather, the N was probably lost from the ecosystem through fire or by leaching to the deep soil solution or to streams [Markewitz et al., 2004]. Thus, plant demands are probably supplied by N mineralization from soil organic N, but in second growth the rates of actively cycling N, including soil organic N, are low. Lower N in soil solutions in secondary forests indicates low cycling of available forms of N and possible N limitation [Davidson et al. 2004]. Phosphorus (Mehlich-III extraction) in both pastures and the 19-year-old second growth was very low (< 1 µg g⁻¹) and only 8.8 kg ha⁻¹ of P was found in the second growth biomass, representing an accumulation of only 0.25 kg ha⁻¹ yr⁻¹ from bio-available P. This also indicates that losses of P from the ecosystem through internal cycling of P occur predominantly through litterfall and grass turnover, which recycle nutrients annually in a way that conserves ecosystem stocks [Markewitz et al., 2004].

C.3.6. Terra preta soils and contemporary nutrient management in Amazonia

Patches of soil with varying spatial extent show high fertility in an otherwise comparatively infertile soilscape. These so-called “terras pretas do indio” or Amazonian Dark Earths (ADE) are man-made soils that occur throughout the Amazon Basin, but with greater concentrations in the middle Amazon and along the larger tributaries [Sombroek et al., 2003]. The total contribution to Amazonian soils and particularly agricultural soils in the Amazon Basin is unclear, and could be as high as several percent but more likely less than 0.1% or 0.3% [Sombroek et al., 2003]. In any event, the proportion of agricultural soils that are terra preta is small and is rather local in importance. However, such patches of terra preta are highly valued by local farmers due to their superior productivity [Lehmann et al., 2003]. The relevance of terra preta currently lies less in its quantitative importance for Amazonian agriculture than in the lessons it may provide for recreation of terra preta. A deeper understanding of the mechanisms by which terra preta has maintained its high fertility over millennia is required to develop meaningful recommendations for modern agriculture. Also, having a high concentration of stable C, it could become an important
sink and stock of atmospheric C if its area is augmented to 5-10% of Amazonia [Sombroek et al., 2003].

Our understanding to date suggests that one of the most important aspects of the fertility of terra preta is its high black carbon (biochar or charcoal) content [Glaser et al., 2001]. The black C increases cation exchange capacity through greater surface area and charge density [Liang et al., 2006]. However, this greater ability of terra preta to retain cations does not, by itself, explain the fact that total nutrient contents are often greater by orders of magnitude than in other soil types, principally for P and Ca [Lehmann et al., 2003]. The most likely source of large amounts of P and Ca is fish residues from the preparation of meals [Lima, 2001; Lehmann et al., 2004]. Other refuse materials may also play a role, and their importance will most likely change significantly as a function of site characteristics and the history of human habitation.

While soil management with fish residues may be feasible in certain situations where fish wastes are available in sufficient quantities, the application of biochar has a much broader applicability. Indeed, research on biochar application to soil has intensified over the past years and provided clear evidence for the potential to improve crop productivity in highly weathered tropical soil [Lehmann and Rondon, 2006]. Biochar can be applied to field crops on an annual basis with as little as a few tons per hectare as well as on larger amounts of several tens of tons to recapitalize soil functions. Such applications have been done using the broadcast method or by application directly to the root zone. In addition, tree crops in plantations or in agroforestry systems can receive biochar placed in the planting hole at the time of site establishment or placed around the stem after planting.

Whether or not biochar soil management will be able to have a major impact on agricultural production in the Amazon depends not only on the refinement of a biochar product but also on the biochar production itself. Lehmann and Rondon [2006] demonstrated that even relatively young secondary forests in the Amazon provide sufficient quantities of biomass to produce biochar in amounts that significantly improve crop productivity. Therefore, charring the biomass instead of burning it during a shifting cultivation cycle would provide the quantities of biochar that are necessary to transform a soil with low cation exchange capacity into a soil with a significantly improved ability to retain cations. Over long periods of time as shown by terra preta soils, approximately a doubling of cation exchange capacity may be achieved [Liang et al., 2006].

However, other factors constraining adoption of biochar application at a larger scale have not been sufficiently taken into account up to now. The work load associated with charring instead of burning the slashed biomass may constitute a significant obstacle. Even though most farmers are familiar with the techniques of making charcoal or have access to the technology, they may find the time or financial expense too large to engage in biochar production. The alternative of marketing as a fuel any charcoal produced provides immediate returns. The full financial benefit of biochar has not been sufficiently demonstrated to farmers to build confidence in the long-term returns, as would be needed to justify the required short-term investments.

C.3.7. Limits to the intensification of agriculture and ranching
Severe limits restrain both the intensification of agriculture and ranching uses and the scale to which these land uses can be expanded [Fearnside, 1997a]. In addition to agronomic limits on per-hectare yields, physical resource limits such as phosphate deposits restrain land uses that depend on these inputs. Amazonian soils are poor in phosphorus, and the vast extent of Amazonia means that converting these areas to land uses that require fertilization with phosphates would quickly exceed existing deposits, both within Brazil and globally [Fearnside, 1998]. Market limits restrain the potential expansion of some of the less-destructive production systems, such as agroforestry [Fearnside, 1995], but provide little restraint on the most destructive land uses, such as cattle pasture [Fearnside, 2005]. In addition, converting tropical forest to these uses carries environmental risks that make policies leading to forest loss unwise as a strategy for developing the region.

In summary, degradation of soil and forest is not inevitable in Amazonia. Cattle pastures, which dominate deforested landscapes in Brazilian Amazonia, can have increased soil organic matter if the best management techniques are used, including certain soil amendments. Soil organic matter sustains the levels of a series of nutrients, leading to greater plant productivity. Agricultural soils can be improved based on lessons learned from the region’s history prior to European contact: indigenous populations left many patches of rich “dark earths” (terra preta), which owe their high carbon content and fertility in part to high content of charcoal. Soil carbon and nutrient retention can be increased in modern agriculture by adding powdered charcoal to the soil, especially if combined with supply of nutrients through fertilization. Agroforestry systems also represent a means of maintaining and increasing soil fertility because of they have greater capacity to cycle nutrients than do other land uses and because the soil surface is protected by a litter layer. The various means of maintaining soil fertility are subject to severe limitations that would prevent their being expanded to the vast areas that have already been deforested, let alone to the much larger areas that are still covered by standing forest but that could meet the same fate if deforestation trends continue unchecked.

Some limiting factors can be faced through technological advances or by applying what is already known about Amazonian productive systems. Any successful production system in Amazonia needs to respect the nature of the region by always keeping the soil covered by forest and by maintaining its high biodiversity in order to remain productive for a long time. It is essential to recognize that some limiting factors cannot be overcome. Thus, other strategies should be adopted to benefit the Amazonian population. The limits to expansion of intensified land uses mean that further deforestation must be prevented, and that development should emphasize the natural forest, which can maintain itself without outside inputs of nutrients.

C.3.8. Environmental services as a basis for development

Maintenance of soil fertility in Amazonian managed systems is not a goal in itself, but rather a means of achieving ends such as supporting the region’s human population in a sustainable way and maintaining the region’s environmental services. The choices of systems to be implanted and managed will be key factors in determining the extent to which these larger goals are met. Cattle pasture, which is by far the most common land use
in deforested areas in Brazilian Amazonia, is not only the least sustainable as managed in the region it is also one of the worst in terms of maintaining the human population. Very few employees are needed to maintain fences and herd cattle on the large and medium ranches that account for most Amazonian deforestation [Fearnside, 1983]. Soybeans are grown in large properties with mechanized agriculture that provides little employment [Fearnside, 2001].

The environmental services provided by Amazonian forests offer a far more valuable and sustainable basis for development than does the expansion of deforestation [Fearnside, 1997b, 2008a]. Tapping the environmental services of intact forest, including maintaining ecosystem carbon stocks in order to avoid global warming [Fearnside, 2008b], maintaining the water cycle [Fearnside, 2004] that supplies rainfall both to Amazonia and the remainder of Brazil, and maintaining biodiversity [Fearnside, 1999], must be recognized as sound strategies for the region.

Conclusions

1.) Amazonian managed systems have very different effects on soil fertility depending on the type of managed system, the way it is managed, and the initial stage of soil degradation.

2.) Pasture management affects the organic matter content of the soil and thereby the soil’s capacity to retain nutrients. Well-managed pastures lead to higher fertility than pastures that are degraded through the extensive management practices that predominate in Brazilian Amazonia today. Note that conversion of Amazon forest to pasture always implies a large loss of ecosystem carbon stocks due to the much higher biomass of the forest.

3.) Agroforestry systems can have an important role in maintaining and improving soil fertility and offer potentially sustainable livelihoods to small farmers in Amazonia.

4.) The possibility of large-scale expansion of intensive pasture, agroforestry or other uses with fertilizer inputs is subject to severe limits from available nutrient sources, especially phosphate deposits. Agroforestry expansion is limited by markets for the products. These limits add to the evidence indicating the wisdom of halting further clearing of Amazonian forest, and of administering the already cleared area in ways that sustain production (a challenge that includes maintaining and improving soil fertility). The soils in deforested areas need to be used in a way that maintains the region’s human population, a role that can be served by smallholder agroforestry but not by large cattle ranches or agribusiness operations such as growing soybeans.

5.) The limits to expansion of intensified land uses mean that further deforestation must be prevented and that development should be based on maintaining the natural forest. Amazonian forests provide environmental services that are more valuable and more sustainable as a foundation for the region’s development than is the expansion of deforestation.
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Figure 1: Brazil’s Legal Amazon region: Savannas (mostly cerrado) is shown in pink, deforested areas in the Amazonian biome in grey, and remaining Amazonian forest in green. The forested and formerly forested areas, together with the small patches of savannas within this area, constitute “biological” Amazonia. ................................. 02

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Table 5: Soil surface (0-10 cm depth) characteristics in a 12-year-old AFS based on palms and fruit trees, and in improved (mixed) pasture and second growth. Means followed by different letters indicate significant differences between treatments (ANOVA, p < 0.05). n = 3. Source: Silva [2005]. ................................. 19

Table 6. Litter production and decomposition rates in managed ecosystems in the Brazilian Amazonia. ................................. 19

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Figure 1: Brazil’s Legal Amazon region: Savannas (mostly cerrado) is shown in pink, deforested areas in the Amazonian biome in grey, and remaining Amazonian forest in green. The forested and formerly forested areas, together with the small patches of savannas within this area, constitute “biological” Amazonia.
Table 1: Volumetric content of exchangeable bases (kg ha\(^{-3}\) m\(^{-1}\)) in the top 1 m of an alic, dystrophic Oxisol, near Manaus (Amazonas) and in a eutrophic Oxisol from Rio Grande do Sul [E.C. Fernandes, pers. comm., 2006].

<table>
<thead>
<tr>
<th></th>
<th>K</th>
<th>Ca</th>
<th>Mg</th>
</tr>
</thead>
<tbody>
<tr>
<td>Manaus, Amazonas (Haplustox)</td>
<td>103</td>
<td>242</td>
<td>120</td>
</tr>
<tr>
<td>Rio Grande do Sul (Hapludoll)</td>
<td>3,330</td>
<td>42,300</td>
<td>9,920</td>
</tr>
</tbody>
</table>
Table 2: Percent (%) of the forest soil affected by different actions during forest management in some selective logging studies in Brazilian Amazonia (RIL = reduced impact logging; CL = conventional logging).

<table>
<thead>
<tr>
<th></th>
<th>Manaus¹ CL</th>
<th>Moju² CL</th>
<th>Fundação Floresta Tropical³</th>
<th>Paragominas⁴ RIL</th>
<th>Paragominas⁴ CL</th>
<th>Acre⁵ RIL</th>
<th>Acre⁵ CL</th>
</tr>
</thead>
<tbody>
<tr>
<td>Log decks</td>
<td>0.4%</td>
<td>0.6%</td>
<td>1.0%</td>
<td>0.6%</td>
<td>1.5%</td>
<td>0.9%</td>
<td></td>
</tr>
<tr>
<td>Skid tracks</td>
<td>1.2%</td>
<td>0.6%</td>
<td>1.3%</td>
<td>2.0%</td>
<td>3.4%</td>
<td>1.1%</td>
<td></td>
</tr>
<tr>
<td>Log tracks</td>
<td>7.7%</td>
<td>3.9%</td>
<td>7.7%</td>
<td>5.1%</td>
<td>10.1%</td>
<td>1.8%</td>
<td></td>
</tr>
<tr>
<td>Total area affected</td>
<td>&gt;12.0%</td>
<td>9.3%</td>
<td>5.1%</td>
<td>10.0%</td>
<td>7.7%</td>
<td>15.0%</td>
<td>3.8%</td>
</tr>
</tbody>
</table>

¹ BIONTE [1997] (timber harvesting: 34 m³)
² Silva et al. [2001] (harvesting: 35 m³)
³ Holmes et al. [2002] (harvesting: CL = 25 m³ RIL = 26 m³)
⁴ Johns et al. [1996] (harvesting: CL = 37 m³ RIL = 30 m³)
⁵ Oliveira and Braz [1995] (harvesting: 20 m³)
Table 3: Comparative studies on soil erosion, runoff and the runoff/total rainfall ratio (%) in several Brazilian Amazon locations. Source: adapted from Barbosa and Fearnside [2000].

<table>
<thead>
<tr>
<th>Vegetation cover</th>
<th>Soil loss kg.ha(^{-1}) yr(^{-1})</th>
<th>Runoff (10^6) L.ha(^{-1}) yr(^{-1})</th>
<th>Runoff: Rainfall (%)</th>
<th>Location</th>
<th>Author</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pasture (Bb)</td>
<td>1,703</td>
<td>2.32</td>
<td>-</td>
<td>Manaus, Amazonas</td>
<td>Fearnside et al. [1986]</td>
</tr>
<tr>
<td>Primary forest</td>
<td>158</td>
<td>0.27</td>
<td>-</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Clean pasture (Pm)</td>
<td>3,556</td>
<td>9.87</td>
<td>49.8</td>
<td>Ouro Preto do Oeste, Rondônia</td>
<td>Fearnside [1989]</td>
</tr>
<tr>
<td>Pasture with weeds (Pm)</td>
<td>664</td>
<td>5.09</td>
<td>25.7</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Primary forest</td>
<td>330</td>
<td>0.37</td>
<td>2.2</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Clear cut</td>
<td>1,140</td>
<td>-</td>
<td>-</td>
<td>Maracá, Roraima</td>
<td>Ross et al. [1990]</td>
</tr>
<tr>
<td>Canopy removal</td>
<td>475</td>
<td>-</td>
<td>-</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Primary forest</td>
<td>270</td>
<td>-</td>
<td>-</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pasture (Bb)</td>
<td>1,128</td>
<td>3.18</td>
<td>15.1</td>
<td>Apiaú, Roraima</td>
<td>Barbosa and Fearnside [2000]</td>
</tr>
<tr>
<td>Primary forest</td>
<td>150</td>
<td>1.13</td>
<td>7.4</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

\(Bb\): *Brachiaria humidicola*  \(Pm\): *Panicum maximum*
Figure 2: Soil microbial biomass-C (µgC g⁻¹) and concentrations of N-NH₄ (µgN g⁻¹) in the 0-10 cm soil layer in pastures of *Brachiaria humidicola* aged from 2 to 13 years, in Manaus, Amazonas. Source: *Luizão et al.* [1999], redrawn.
Table 4: Annual nutrient input to the soil (kg ha\(^{-1}\)) in two 6-year-old agroforestry systems (AS1- palm-based, and AS2- multi-strata), and in a 10-year-old second growth (CAP), via fine litter and green manure (from prunings). Values in parenthesis represent the relative contribution (%) of each source (fine litter and green manure) to the total nutrient input in the AFS. Source: Gallardo-Ordinola [1999].

<table>
<thead>
<tr>
<th>Treatment</th>
<th>Components</th>
<th>N</th>
<th>P</th>
<th>K</th>
<th>Ca</th>
<th>Mg</th>
</tr>
</thead>
<tbody>
<tr>
<td>AS1</td>
<td>Litter</td>
<td>36.8</td>
<td>2.35</td>
<td>5.76</td>
<td>32.7</td>
<td>8.64</td>
</tr>
<tr>
<td></td>
<td>(54%)</td>
<td>(72%)</td>
<td>(48%)</td>
<td>(92%)</td>
<td>(80%)</td>
<td></td>
</tr>
<tr>
<td>AS1</td>
<td>Prunings</td>
<td>16.8</td>
<td>0.94</td>
<td>6.23</td>
<td>2.87</td>
<td>2.11</td>
</tr>
<tr>
<td></td>
<td>(46%)</td>
<td>(28%)</td>
<td>(52%)</td>
<td>(8%)</td>
<td>(20%)</td>
<td></td>
</tr>
<tr>
<td>AS1</td>
<td>Total</td>
<td>53.6</td>
<td>3.29</td>
<td>12.0</td>
<td>35.6</td>
<td>10.8</td>
</tr>
<tr>
<td>AS2</td>
<td>Litter</td>
<td>36.3</td>
<td>1.90</td>
<td>5.01</td>
<td>28.7</td>
<td>8.58</td>
</tr>
<tr>
<td></td>
<td>(60%)</td>
<td>(59%)</td>
<td>(37%)</td>
<td>(84%)</td>
<td>(45%)</td>
<td></td>
</tr>
<tr>
<td>AS2</td>
<td>Prunings</td>
<td>24.5</td>
<td>1.33</td>
<td>8.57</td>
<td>5.31</td>
<td>10.8</td>
</tr>
<tr>
<td></td>
<td>(40%)</td>
<td>(41%)</td>
<td>(63%)</td>
<td>(16%)</td>
<td>(55%)</td>
<td></td>
</tr>
<tr>
<td>AS2</td>
<td>Total</td>
<td>60.8</td>
<td>3.23</td>
<td>13.6</td>
<td>34.0</td>
<td>19.4</td>
</tr>
<tr>
<td>CAP</td>
<td>Litter = Total</td>
<td>64.1</td>
<td>3.82</td>
<td>12.6</td>
<td>45.2</td>
<td>13.6</td>
</tr>
</tbody>
</table>
Table 5: Soil surface (0-10 cm depth) characteristics in a 12-year-old AFS based on palms and fruit trees, and in improved (mixed) pasture and second growth. Means followed by different letters indicate significant differences between treatments (ANOVA, p < 0.05). n = 3. Source: Silva [2005].

<table>
<thead>
<tr>
<th></th>
<th>Agroforestry Systems</th>
<th>Secondary Forest</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Palms</td>
<td>Fruit trees</td>
</tr>
<tr>
<td>pH (4.5 a ± 0.14)</td>
<td>4.4 ab ± 0.08</td>
<td>4.5 a ± 0.11</td>
</tr>
<tr>
<td>C (%) (3.1 ± 0.9)</td>
<td>2.8 ± 0.5</td>
<td>3.1 ± 0.7</td>
</tr>
<tr>
<td>N (%) (0.2 ± 0.03)</td>
<td>0.2 ± 0.03</td>
<td>0.21 ± 0.03</td>
</tr>
<tr>
<td>P (mg.kg(^{-1}))</td>
<td>58.2 a ± 26.5</td>
<td>68.8 a ± 41.4</td>
</tr>
<tr>
<td>K (mg.kg(^{-1}))</td>
<td>147 ± 126</td>
<td>152 ± 140</td>
</tr>
<tr>
<td>Mn (mg.kg(^{-1}))</td>
<td>6.5 ± 2.1</td>
<td>7.1 ± 2.4</td>
</tr>
</tbody>
</table>
Table 6. Litter production and decomposition rates in managed ecosystems in the Brazilian Amazonia.

<table>
<thead>
<tr>
<th>Location</th>
<th>Ecosystem</th>
<th>Litter production (Mg ha$^{-1}$)</th>
<th>Litter decomposition (half life)</th>
<th>Authors</th>
</tr>
</thead>
<tbody>
<tr>
<td>Manaus</td>
<td>10-year-old <em>Bertholletia excelsa</em> plantations</td>
<td>1.3</td>
<td>518 days</td>
<td>Kato [1995]</td>
</tr>
<tr>
<td></td>
<td>5-year-old <em>Bertholletia excelsa</em> plantations</td>
<td>0.67</td>
<td>518 days</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Forest (control)</td>
<td>7.0 – 10.0</td>
<td>83 days</td>
<td>Gallardo-Ordinola [1999]</td>
</tr>
<tr>
<td></td>
<td>Agroforestry 1</td>
<td>2.0</td>
<td>25-116 days</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Agroforestry 2</td>
<td>2.3</td>
<td>25-116 days</td>
<td></td>
</tr>
<tr>
<td>Capitão Poço</td>
<td>3-year-old fallow field</td>
<td>5.0</td>
<td>-</td>
<td>Dantas and Phillipson [1989]</td>
</tr>
</tbody>
</table>
Table 7: Soil nutrients from the upper layer (0-10 cm) in 1-year-old reforested plots with (RWTS1) and without (RWTH1) topsoil replacement; in 11-year-old natural regeneration (NR11); in 11-year-old reforested plots (REF11); and in primary forest (PF) at the Saracá Mine, Porto Trombetas, Pará in the wet season. Values are means ± SE (n = 3). Source: Costa et al. [2002].

<table>
<thead>
<tr>
<th>Treatment</th>
<th>total-N g kg⁻¹</th>
<th>P mg dm⁻³</th>
<th>K⁺ cmol·dm⁻³</th>
<th>Ca ++ cmol·dm⁻³</th>
<th>Mg ++ cmol·dm⁻³</th>
</tr>
</thead>
<tbody>
<tr>
<td>RWTS1</td>
<td>0.80 b ± 0.20</td>
<td>1.38 a</td>
<td>0.09 abc</td>
<td>0.46 c</td>
<td>0.16 cd</td>
</tr>
<tr>
<td>RWTS1</td>
<td>0.88 b ± 0.11</td>
<td>0.88 b</td>
<td>0.01</td>
<td>0.03 c</td>
<td>0.01 d</td>
</tr>
<tr>
<td>NR11</td>
<td>1.02 b ± 0.31</td>
<td>1.50 a</td>
<td>0.11 ab</td>
<td>1.53 ab</td>
<td>0.38 ab</td>
</tr>
<tr>
<td>REF11</td>
<td>2.33 a ± 0.20</td>
<td>1.68 a</td>
<td>0.16 a</td>
<td>2.34 a</td>
<td>0.49 a</td>
</tr>
<tr>
<td>PF</td>
<td>2.11 a ± 0.12</td>
<td>1.46 a</td>
<td>0.06 bc</td>
<td>0.07 c</td>
<td>0.11 cd</td>
</tr>
</tbody>
</table>

Means followed by similar letters are not significantly different (p < 0.05) according to the Tukey test.