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Linking Biogeochemical Cycles to Cattle Pasture Management and Sustainability in the Amazon Basin

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Pasture development has become the largest anthropogenic disturbance of forest land in the Amazon basin (Skole et al. 1994, Serrão and Toledo 1990). The area of forests converted to cattle pasture in Amazonia is currently estimated at approximately 20 million hectares. In the Brazilian Amazon basin, most of the conversion of forest land to pasture began during the early 1960s to the late 1980s, as a consequence of the opening of Amazon highways and government policies aimed at regional development (Hecht 1982, Nepstad et al. 1991, Serrão et al. 1979).

Pasture productivity and longevity in the Amazon basin seem to be closely related to soil fertility and nutrient cycling (e.g., Dias Filho and Serrão 1987, Serrão et al. 1979). Thus, understanding the major biogeochemical cycles that influence soil fertility under pasture is vital for predicting the consequences of continued conversion of tropical forests to cattle pastures. This understanding is also important for devising management technologies that enhance the sustainability of these areas and thus slow further deforestation.

Although during the first three to five years after establishment, the productivity of pastures is often good, after that period a rapid decline in productivity of the planted grasses associated with an increased presence of herbaceous and woody invaders is generally observed (reviewed by Serrão and Toledo 1990). If left uncontrolled, these invader species slowly become dominant and lead to "pasture degradation," a condition characterized by a complete dominance of the weedy community. If left to secondary succession, forest vegetation usually becomes reestablished on these degraded pasture lands in the Amazon, although the species composition is usually different than that of the primary forest (Nepstad 1989). The nutrient status of the degraded pasture soils is among the factors that affect the rate of regrowth of the secondary forests.

One of the first attempts to study soil nutrient dynamics under cultivated pastures in the Amazon basin was conducted in the early 1970s by Falesi (1976). The results of that chronosequence study in different soil types suggested that soil nutrient cycling in pastures differed from that of the traditional slash-and-burn agriculture. The decline in the levels of some nutrients in the soil was found to be more gradual, and the decline in productivity over time in planted pastures could, at least in part, be associated with the behavior of available phosphorus in the soil.

Although the sampling methodology of Falesi's chronosequence study has been criticized (see Hecht 1982), other studies considering the dynamics of soil fertility in pasture areas in the Amazon basin found similar results (Diez et al. 1991, Hecht 1982, Buschbacher 1984, Moraes et al. 1996, Teixeira 1987). The connection between soil fertility (mainly available phosphorus) and pasture productivity and longevity in the Amazon basin was further established in many studies conducted from the late 1970s to the early 1990s (Azevedo et al. 1982a, 1982b; Dias Filho and Serrão 1980, 1981, 1982, 1987, Dias Filho et al. 1989, Dias Filho and Simão Neto 1992, Embrapa 1980, Italiano et al. 1982, Serrão et al. 1979).

The promotion of efficient nutrient recycling has been acknowledged as a means of enhancing the stability of pasture systems in tropical America (e.g., Dias Filho 1986, Serrão and Toledo 1990, Spain and Salinas 1985). In the Amazon basin, efficient nutrient recycling is even more important, as most of the soils where pasture development is concentrated are nutrient-poor Oxisols and Ultisols. However, there has been very little progress in understanding major biogeochemical cycles in active and abandoned pastures in the Amazon basin. An attempt to directly address the role of grazing animals in nutrient recycling of an active pasture in Amazonia was made by Buschbacher (1984, 1987). More detailed studies have followed this, however, without directly addressing the role of cattle and grazing, and considering specifically only nitrogen and carbon dynamics in pasture soils (Dejardins et al. 1994, Feigl et al. 1995a, 1995b, Neill et al. 1995, 1997, Nepstad et al. 1994, Trumbore et. al. 1995). We review these studies here, and we attempt to identify the most important issues for our understanding of pasture biochemistry: How does nutrient availability in pastures change over time? What nutrients are most critical? What is the role of cattle and grazing? What are the effects of altered hydrology? How does weed invasion affect pasture biogeochemistry? Also, we attempt to identify what management practices are important to promote pasture longevity and sustainability, and to reclaim degraded pasture areas.

Land Clearing Methods and Pasture Establishment

The method of land clearing can have important and long-lasting consequences for pasture productivity and longevity (Dias Filho 1986, Dias and Northcliff 1985, Seubert et al. 1977). Land clearing for pasture establishment on forested areas in the Amazon basin normally involves cutting the forest and then burning of the vegetation (Dias Filho 1986, Serrao and Toledo 1990, Toledo and Morales 1979). Clearing is normally done with chain saws, generally after the valuable wood species have been removed. Mechanical clearing for pasture establishment using heavy machinery is seldom practiced in the Amazon basin, mainly because of high operational costs, but also due to the detrimental effects mechanical clearing is known to have on soil properties, reducing pasture productivity (reviewed by Dias Filho 1986).

Perhaps one of the earliest and most influential studies that compared the agricultural impact of manual (slash-and-burn) and mechanical (bulldozer) clearing for pasture establishment in the Amazon basin was that of Seubert et al. (1977) in Peru. In that study, pasture (*Panicum maximum*) biomass production, over a 2-year period, was 68 % higher in the area cleared manually. Other studies have followed, with the general conclusion that pastures manually cleared from forest were more productive and that mechanical clearing adversely affected soil physical and chemical properties (e.g., Alegre 1985, Toledo and Morales 1979, Dias and Northcliff 1985).

The basic difference between the two land clearing methods is that in the "manual" process, fire is the major agent employed to remove vegetation cover and prepare the land for pasture establishment. From an agricultural perspective, burning can be regarded as a fast and economical process to easily prepare the area for cultivation. From an ecological viewpoint, however, it causes immediate disruption of major biogeochemical cycles through the production and release into the atmosphere of particulates and environmentally significant gases, and the release into the soil of relatively large amounts of nutrients previously immobilized in the aboveground biomass. Forest biomass burning also affects the hydrological cycle by changing rates of land evaporation and water runoff.

Losses from the system due to combustion of biomass can be quite high for carbon, nitrogen, and sulfur, and less for major nutrients like phosphorus, potassium, and calcium (Ewel et al. 1981, Frost and Robertson 1985, Kauffman et al. 1995, Sanchez 1976, Wright and Bailey 1982). However, immediate improvement in the soil chemical characteristics – higher pH, reduced aluminum toxicity, and higher nutrient content—is usually observed after forest cutting and burning in the Amazon basin (Buschbacher 1984, Dantas 1989, Dantas and Matos 1980, Diez et al. 1991, Falesi 1976, Seubert et al. 1977, Smyth and Bastos 1984).

The general improvement in soil chemical properties after burning is mainly due to constituents in the ash that provide substantial inputs of bases and other elements to the soil. Also, heating caused by burning may contribute to the improvement of soil fertility by enhancing the mineralization of elements (N, Ca, Mg, and P) formerly linked and complexed with organic matter (Giovannini et al. 1990).

During pasture establishment, available soil nutrients are generally at their highest levels due to ash deposition (after forest clearing) or chemical fertilization (in reclaimed pastures). In addition, the soil is left unprotected against high temperatures, which accelerates the decomposition rates of fresh organic matter, and against wind and raindrop impact, which contribute to erosion and soil compaction. For these reasons, a fast and efficient soil cover by the forage plant is key in determining the future of pasture productive longevity. Any failure in pasture establishment will expose the area to an increased nutrient loss and weed invasion that ultimately may lead to premature pasture degradation. According to Dias Filho (1986), forage seed quality, seeding rates, time of planting and of beginning grazing, and the agronomic characteristics of the forage species are the most crucial aspects affecting successful pasture establishment.

Carbon Cycling

The most important effect of forest-to-pasture conversion on the carbon cycle is the release of 100–200 tons C ha^{-1} from aboveground forest biomass to the atmosphere. However, soil C stocks are often as large or larger than aboveground biomass-C, and the changes in soil C stocks, although much less than changes in aboveground C, can also be significant (Nepstad et al. 1994). Inventories of soil C stocks following tropical pasture formation have shown increases (Bushbacher 1984, Cerri et al. 1992, Choné et al. 1991, Feigl et al. 1995, Fisher et al. 1994, Neill et al. 1996), decreases (Bushbacher et al. 1988, Desjardin et al. 1994, Detweiler 1986, Eden et al. 1991, García-Oliva et al. 1994, Luizão et al. 1992, Street 1982, Veldkamp 1994), and mixed results (Brown and Lugo 1990, Trumbore et al. 1995). The key to understanding these conflicting responses is variation in the productivity of pasture grasses.

A comparison of C budgets for forests and pastures in the eastern Amazon was made by Trumbore et al. (1995). In a reformed and fertilized pasture of *Brachiaria brizantha*, they estimated gains, relative to forest soil C stocks, of over 20 tons soil C ha^{-1} in the top 1 m of soil and a loss of about 0.5 tons C ha^{-1} in the 1–8 m soil depth interval during the first 5 years following pasture reformation. In contrast, they calculated a loss of about 9 tons soil C ha^{-1} in the top 1 m of an abandoned, degraded pasture soil and no change below 1 m. As shown in Table 6.1, total C inputs were greatest in the forest, due to high rates of litterfall, but 20–30% of the litter C probably decomposes to CO_2 within the litter layer. Root inputs were greatest in the top 1 m of soil in the reformed productive pasture, which resulted in gains of soil C stocks there. However, the deeply rooted woody vegetation had been removed from the reformed pasture, resulting in very low rates of root inputs to deep soils, causing losses of deep soil C in the productive pasture. Low rates of litter and root inputs in the degraded pasture caused loss of soil C in the topsoil. Hence, both total productivity and the distribution of C inputs to the soil among litter and roots of varying depth are the key to determining whether C will be lost or gained in topsoil and subsoil.

These changes in soil C stocks are relatively rapid, due to rapid turnover of soil organic matter (SOM) in these tropical soils. Using radiocarbon derived from atmospheric testing of nuclear weapons in the 1960s as a tracer, Trumbore et al. (1995) estimated that the mean residence time of C in the top 10 cm of soil is about 3 years for 30% of the SOM and 10–30 years for 60% of the SOM. Only about 10% of the C in the top 10 cm of soil cycles on a millennial time scale. This very old C fraction increases to 40-80% in the 10-100 cm depth interval and to 80-90% below 1 m depth. Hence, large changes in soil C can be observed near the surface in a matter of years, although modest changes in modern soil C at depth are also significant when summed over the entire profile of 8 or more meters.

Losses of soil C were also found in pastures of the eastern Amazon by Desjardin et al. (1994), whereas increases of soil C in Amazonian cattle pastures have been reported in the western Brazilian Amazon (Cerri et al. 1992, Choné et al. 1991, Feigl et al. 1995, Neill et al. 1996). The studies in the western Brazilian Amazon were of pasture soils that had not been fertilized. It is tempting to interpret these results as an indication that the soils in the west tend to have greater native soil fertility than those in the east, thus characteristically supporting more productive pastures without fertilization, but this broad generalization may be too simplistic. It is true that the probability of finding eutrophic soils increases in a westerly direction toward the Andes, but both eastern and western regions have a wide variety of soil types. The studies in the western Brazilian Amazon included Oxisols, Ultisols, and Alfisols, but these taxonomic classifications are too broad for reliable regional extrapolation. Both eutrophic and dystrophic soils occur within the orders of Oxisols and Ultisols.

In addition to soil fertility, there are differences in climate and there may be differences

Table 6.1 Comparison of C inputs to soils at Fazenda Vitoria, Paragominas, Brazil.

C Inputs	Forest (tons C ha^{-1})	Degraded Pasture (tons C ha^{-1})	Productive Pasture (tons C ha^{-1})
Litter	4.5	1.4	2.0
Roots (0–1 m depth)	1.9	1.2	3.4
Roots (1–5 m depth)	0.9	1.3	0.2
Total	7.3	3.9	5.6

Source: Trumbore et al. 1995

in management practices between the eastern and western regions of the Brazilian Amazon basin. Most of the eastern Amazon, including the Paragominas study site of Trumbore et al. (1995), has a 3-5 month dry season that limits pasture productivity due to severe water limitation and that may increase the frequency of fire. Frequent fire would reduce nitrogen stocks that might, in turn, decrease grass productivity. Regional differences in pasture management are partly historical. Forest clearing along the Belém-Brasília highway began in the 1960s and 1970s (Nepstad et al. 1991), when the use of less productive grass species was common and when less was known about optimal grazing intensity, which resulted in severe overgrazing in many areas. Examples of overgrazing probably exist in all regions, but they may be more common in the eastern Amazon.

In summary, we know that pasture productivity is the key to predicting whether soil C stocks will increase or decrease following forest conversion to pasture, and we know that several factors, including native soil fertility, fertilization, climate, fire frequency, and grazing intensity, influence pasture productivity. We do not, however, know which of these factors has had the greatest influence in the past and which will be most important in the future for determining productivity in Amazonian pastures. Nor do we know the areal extent of degraded v. productive pastures. Quantitative estimates of areas of productive and nonproductive pasture lands would be necessary to calculate basin-wide effects of forest-to-pasture conversion on the regional C budget. Finally, we wish to emphasize that sustainability of pasture management practices and maintenance of soil C stocks are practically synonymous.

Phosphorus Cycling

Of all the essential plant nutrients, P is most commonly considered a limiting factor to primary productivity in tropical ecosystems on highly weathered soils. Due to its low mobility in the soil and high stability (no biologically important form of P is gaseous), and because inputs (e.g., wet and dry deposition) are roughly equivalent or even higher than outputs (e.g., erosion and runoff), P exists in relatively constant amounts or even increases over time scales of years to centuries in some natural terrestrial ecosystems. In managed ecosystems, however, net losses of P may be high because of harvest of products like grain,

Table 6.2 Forms of P in the top 10 cm of mineral soil at Fazenda Vitoria, Paragominas, Pará.

Site	Total P (kg P ha^{-1})	NaOH Extractable		Mehlich 3 (kg P ha^{-1})
		Organic-P (kg P ha^{-1})	Inorganic-P (kg P ha^{-1})	
Primary forest	194	19	26	1.6
Degraded pasture	250	30	30	1.2
Reformed pasture	298	36	38	1.1
20-yr Secondary forest	209	26	28	0.8

Source: Unpublished data from D. Markewitz and E. A. Davidson.

* Higher values in the pastures are due more to higher bulk density in the compacted pasture soils than due to higher P concentrations.

fiber, and roots. The generally higher vulnerability of these ecosystems to surface runoff and erosion can also contribute to P losses. This condition results from incomplete soil coverage by the crop plant and lower infiltration capacity of disturbed soils.

Highly weathered tropical soils of oxic and kaolinitic mineralogies often contain large stocks of total P, but the vast majority of this is bound to iron and aluminum oxides in forms that are generally thought to be un-available to plants. Table 6.2 demonstrates that the stocks of total P in the top 10 cm of the mineral soil of a kaolinitic Oxisol at Paragominas, Pará, are about 200 kg ha^{-1}, but that less than 1% of the total P is extracted by the Mehlich 3 extraction method. Agronomists often use the Mehlich extraction as an index of P available to plants, but, clearly, this pool is insufficient to provide plant needs of P and must be replenished frequently from other pools (Mengel and Kirkby 1987). It is unclear whether the P that replenishes the plant-available Mehlich pool is derived from enzymatic decomposition of organic P or from gradual desorption of inorganic P, or both. Below 10 cm depth, the concentrations of total P remain similar to those in the top 10 cm, indicating that P is abundant in these soils. However, the concentrations of NaOH-extractable and Mehlich-extractable P decline to barely detectable levels below 50 cm depth. Phosphorus is most likely a limiting nutrient for plant growth in those ecosystems.

For the great majority of Amazon basin soils, available P in the topsoil is known to be naturally low (León and Hammond 1985). After forest clearing and burning, available P in the topsoil usually increases due to ash deposition and slash decomposition (e.g., Falesi 1976, Seubert et al. 1977). However, unlike other nutrients which are often maintained at more or less stable levels under pasture, available P in top soil often falls sharply with time after pasture establishment (Falesi 1976), suggesting that a mining effect might be occurring. This decrease in available soil P is normally associated with a decrease in forage grass biomass production, generally leading to pasture degradation (i.e., an increase in weed biomass) and abandonment. In these degraded pasture areas, forage grasses often respond to P fertilization (Table 6.3) but fail to respond to fertilization with other nutrients (Table 6.4), implying that P availability is the major impediment to pasture productivity.

In view of its importance to pasture primary production, maintaining relatively high levels of available P is a major challenge to pasture managers in the Amazon basin. However, insufficient knowledge of the biogeochemistry of P in active Amazonian pastures is a

Table 6.3 Response of *Andropogon gayanus* to P and N fertilization.

Fertilizer (kg ha^{-1})*	Dry Matter (kg ha^{-1})**
0 N + 0 P	828
0 N + 50 P	1456
75 N + 0 P	405
75 N + 50 P	2875

Source: Dias Filho (1986)

* Area near Paragominas, Brazil, on yellow latosol (Haplustox) originally under degraded pasture, reclaimed by burning, tilling, planting, and fertilizing in May 1980 (end of wet season).

** Dry matter production evaluated at 20 cm from the soil surface in February 1981.

Table 6.4 Dry matter production of a 12-year-old degraded *Panicum maximum* pasture after weeding and fertilizer application at Paragominas on a yellow latosol (Haplustox).

Treatment	Dry Matter (kg ha^{-1})*
Control	7428
Legumes interplanted	6127
Complete** minus N	12508
Complete minus P	4654
Complete minus K	11120
Complete minus S	15667
Complete minus lime	12183
Complete minus micronutrients	11630
Complete	12249

Source: Dias Filho and Serrão (1987).

* Tukey's HSD = 6230.

** Complete = 150 N +100 P + 100 K + 50 S + 1000 Lime + 30 micronutrients (in kg ha^{-1}).

major limitation to achieving this goal. The biogeochemical P cycling of active pastures differs from forest and, to some extent, abandoned pasture and agricultural ecosystems in its complexity and unpredictability. Cattle grazing affects the rate of P movement and distribution within the system and increases its potential for loss.

In active pasture systems for beef production, the harvested product is the animal, with a potential to export relatively low amounts of P. Considering an average carrying capacity of 0.8 animal units per hectare per year (1 animal unit = 450 kg of live weight) for a typical *Brachiaria brizantha* pasture with medium productivity, the amount of P that could potentially be exported from the system in one year, due to meat production, would be around 2.5 kg per ha (based on a body composition of 0.7 % P for beef cattle).

Under the management conditions typical of most beef cattle pasture systems in the Amazon basin, animals would get their dietary P from two major sources: mineral supplementation and pasture plants. Involuntary soil intake could also be a potential, but difficult to quantify, input source of P.

Because of the low P content (usually < 0.2 %) of forage plants, P supplementation is commonly practiced in most ranches throughout the Amazon basin, particularly in Brazilian Amazonia. Assuming a stocking rate of 0.8 animal units per hectare per year, the annual input of P to the system through animal consumption of mineral supplementation would be around 2.0 kg, which is close to the amount expected to be exported annually by animal products (2.5 kg). The amount thus needed to balance the cycle would be only 0.5 kg. In the absence of phosphate fertilizer inputs, this P must come from the soil pool reserves, through forage consumption.

Under the assumption of a forage dry matter P content of 0.12 %, and a stocking rate of 0.8 animal units per hectare, we can estimate that 3.9 kg of P would be ingested by the animal in one year through forage consumption. This gives a total amount of available P for recycling of 3.4 kg ha^{-1} yr^{-1} (3.9–0.5). Therefore, most of the P taken from the soil by plant uptake and consumed in the forage by the grazing animal on a daily basis, would be returned to the system through animal excreta.

Although it may seem straightforward that there is an efficient P recycling through the return of ingested P in animal excreta, in practical terms this efficiency is highly questionable, because feces and urine are not distributed uniformly over the pasture but deposited on specific areas. Usually, areas close to drinking troughs, mineral feeders, or shade receive greater amounts of feces and urine. For example, in a ***Brachiaria decumbens*** pasture in the Venezuelan Amazonia, it was found that more than one-half of the dung excreted by Zebu cattle was concentrated in only 30 percent of the pasture area (Buschbacher 1987). The proximity of cattle gathering areas to streams and ponds produces the potential for runoff of P to significantly alter nutrient cycling in aquatic ecosystems, but this topic has not been studied to our knowledge.

In addition to greatly altering P distribution within the pasture area due to the patchy distribution of excretion, cattle can also indirectly enhance the potential of P movement in the landscape. This is because cattle transit within the pasture area can create trails which are channels for runoff, and overgrazing and trampling may result in open areas that facilitate losses through runoff of particulate and dissolved P and erosion of surface soil and plant material. Since most of the excreted P is in the feces and most of the feces are deposited in areas where cattle transit and concentration are more intense, the potential for P loss from the system into surface waters through erosion and runoff can be greatly enhanced during the heavy rains typical of most areas in the Amazon basin. Comparison of the water infiltration capacity of adjacent forest and reclaimed pasture soils in the Paragominas region, eastern Amazonia, has shown that water can infiltrate up to twenty times slower in the soil under pasture (M. B. Dias Filho, unpublished data). Moreover, P transferred in feces to bare trails and campsites will have little or no effect on forage production and P recycling.

Cattle also have a great impact on the internal P (nutrient) cycling of forage species. Because grazing is selective and often the plant parts with the greatest amount of nutrients (i.e., young leaves) are preferentially consumed, there is a profound impact on the resorption efficiency of P, as well as other nutrients. This, in turn, stimulates soil P uptake, and if recycling through animal excretion is not efficient (which is often the case, mainly under low stocking rates) and there is no P fertilizer input, a slow depletion of soil P will usually occur.

The amount of P removed from the soil by forage grass in a typical Amazonian pasture can be substantial (Table 6.5). Ongoing

Table 6.5 Annual dry matter production (mean of two years) and P uptake (kg of P in plant tops) of ***Brachiaria brizantha*** cv. ***Marandu.***

Fertilizer (kg P ha^{-1})*	Dry Matter (kg ha^{-1} $year^{-1}$)**	P Uptake (kg ha^{-1} $year^{-1}$)**
0	13900 (696)	13.7 (1.07)
22	23334 (834)	29.5 (1.75)
44	28159 (1651)	39.7 (2.26)

Source: Adapted from Dias Filho and Simão Neto (1992).

* Area at Paragominas, Brazilian Amazonia on yellow latosol (Haplustox) originally under degraded pasture, reclaimed by burning, tilling, planting and fertilizing with simple superphosphate.

** Data are mean (± s.e.) n= 4.

Table 6.6 P and nitrogen concentration* (% of dry weight) of forage grass (*Hyparrhenia rufa*), herbaceous weedy vegetation and litter** of a 10-year-old degraded pasture in eastern Brazilian Amazonia.

Source	P	N
Grass	0.10	0.63
Weeds	0.20	1.70
Litter	0.06	0.78

Source: Dantas (1989).
* Mean of three sample periods during 2 consecutive years.
** Dead and dry material from many species (mainly weeds).

research (C. J. R. de Carvalho and M. B. Dias Filho, unpublished data) has shown that forage grasses like *Brachiaria humidicola* and *B. brizantha* have the ability to respond to low available P in the soil by secreting acid phosphatase at the rhizosphere. Acid phosphatase can solubilize organic phosphate in the soil, increasing the availability of P for plant uptake.

If not grazed, or if the pasture is not burned, most of the P absorbed in plant tops will be immobilized by plant tissue and, thus, will be unavailable in the soil. Transfer of P and N by nutrient resorption in forage plants and in the weedy vegetation prior to leaf fall induces low nutrient concentration in litter (Table 6.6) and may in part explain the decreasing amount of available soil P often associated with an increase in the weed biomass of a degrading pasture or with pasture age. Without this biological immobilization, however, these nutrients would be more vulnerable to loss through runoff and erosion.

Immobilization of P in the soil in the short or long term by geochemical fixation processes might, in some situations, play an important role in the cycling of P in pasture ecosystems. However, unlike in the acid savanna soils, where high P fixation is prevalent, only a relatively small proportion of Amazon soils (mainly Oxisols and Ultisols of clayey topsoil texture) are known to have the capacity to fix large quantities of P into relatively insoluble forms (Sanchez 1987). For this reason, P fixation has received far less attention than the role of the biologically mediated organic P transformations in this ecosystem. Geo-chemically fixed P is probably not available for plant uptake and subsequent animal consumption, but could be vulnerable to loss from the system via soil erosion.

In summary, we know that available P is often found in low amounts in the great majority of Amazon basin soils. After forest clearing and burning, available P levels increase in topsoil, but during pasture utilization these levels often fall sharply. This decrease in P availability is usually associated with a decline in pasture productivity and an increase in weed biomass. The accumulation of weed biomass sequesters soil P and, thus, temporarily decreases its availability in the soil. In these degraded pastures, forage grasses often respond to P fertilization but fail to respond to fertilization with other nutrients, strongly suggesting that P availability is the major limitation to pasture productivity. Cattle probably play a key role in the P dynamics of active pastures areas by affecting P distribution patterns within the pasture area and facilitating P losses through runoff and erosion.

Nitrogen Cycling

Land use changes, such as conversion of tropical forests to cattle pastures, affect biological N fixation, mineralization, nitrification, and denitrification (Davidson et al. 1993, Keller and Reiners 1994, Matson et al. 1987, Montagnini and Buschbacher 1989, Neill et al. 1995). Matson et al. (1987) estimated that N mobilized annually from deforestation was equivalent to more than half of the industrial N fixed globally and greater than the total amount of N delivered to oceans by rivers.

Tropical ecosystems on highly weathered soils present an enigma to our current understanding of the N cycle. Based on ratios of elements found in foliage and litterfall of both leguminous and nonleguminous trees, N appears to be an abundant nutrient relative to P in forest vegetation growing on many soils dominated by kaolinitic and oxic mineralogies (Vitousek and Sanford 1986). And yet, despite the apparent relative abundance of N in these ecosystems, leguminous trees are abundant in these forests, sometimes being nodulated and sometimes not nodulated (Salati et al 1982). Do these legumes allocate energy to N fixation despite high availability of N in these ecosystems? Is N abundant in these forest ecosystems because symbiotic N fixation exceeds plant demands? (See Cuevas, this volume, for a thorough discussion of nutrient cycling in Amazon forests.)

Biological N fixation may also be important in active and abandoned pastures, although the evidence for it is mostly inferential. In abandoned pastures, nodulated leguminous species are common early successional plants, suggesting that symbiotic N fixation is advantageous for growth in these degraded soils. The use of planted grass-legume pastures is not common in the Amazon, but spontaneous growth of herbaceous legumes (e.g., *Centrosema* spp., *Calopogonium* spp., *Desmodium* spp.) can be found in most active pasture areas, and their contribution to the N budget is unknown. When degraded pastures are reformed and fertilized with P, grass biomass production can be substantial (Tables 6.4 and 6.5), and it is difficult to believe that all of the N in this new plant biomass ($>$ 100 kg N ha^{-1}) could be derived from nutrient-poor degraded pasture soil. Perhaps symbiotic N fixation by the spontaneous herbaceous legumes or associative N fixation in the grass rhizosphere may be necessary to supply sufficient N to meet plant needs. If N fixation does occur in active pastures, that may explain the absence of a response in grass growth to N fertilization.

Associative N fixation, the process in which free-living N fixing bacteria utilize root exudates as energy sources and can maintain active nitrogenase enzymes in the low O_2 environment of the rhizosphere, has been inferred for tropical grasses such as sugar cane (Döbereiner et al. 1972) and several species of pasture grasses (Boddey and Victoria 1986). Conditions in the grass rhizosphere may be conducive to free-living N fixing bacteria, including abundant root exudates, respiration that consumes O_2, soil pH usually in the range of 6 to 7, and sufficient P from fertilization. Given the high productivity of these tropical grasses, it seems plausible that their rhizosphere N fixation could be significant, but conclusive evidence for associative N fixation in pastures is still lacking. Chronosequence studies in Rondonia showed similar or higher N stocks in the surface soil of pastures compared to forests, although spatial heterogeneity made it difficult to conclude that significant increases in N stocks due to associative N fixation in the pastures had occurred (Piccolo et al. 1994).

Comparisons of total N stocks may belie more important differences in "plant-available N" among forest and pasture soil. Total N is a poor indicator of plant-available, actively cycling N pools. Preliminary results from studies at Paragominas (L. V. Verchot and E. A. Davidson, unpublished data) show that net N mineralization and net nitrification rates are low in the pastures compared to mature forest (Table 6.7). The NO and N_2O flux data follow the same trend as the net N mineralization data: primary forest > secondary forest > improved pasture > degraded pasture (Table 6.7). Because N gas emissions are strongly affected by N cycling rates and N availability in the soil (Firestone and Davidson 1989), these data also imply decreased N availability in the pastures. In Rondonia, a similar pattern has been observed, in which cattle pastures ranging from 3 to 80 years old had lower rates of net N mineralization and net nitrification than did adjacent forest soils (Neill et al. 1995).

The secondary forest at Paragominas has recovered much of the net N mineralization capacity that is characteristic of primary forest, and the improved (reformed) pasture has higher net nitrification than the degraded pasture (Table 6.7). These results provide further inferential evidence that biological N fixation or some other N inputs result in partial recovery of the N cycle in secondary forests and improved pastures.

Nutrients mineralized from soil organic matter during forest clearing usually support shifting agriculture or productive pastures for only a few years in many tropical regions, resulting in nutrient-poor, degraded soils in abandoned farms and pastures (Keller and Reiners 1994, Tiessen et al. 1994, Serrão and Toledo 1990). In the case of N, there are several mechanisms by which N can be rendered unavailable, redistributed, or lost from pasture soils.

Root systems of tropical pasture grasses and decomposing litter are known to immobilize significant amounts of N (Bushby et al. 1992, Robbins et al. 1989), perhaps into organic-N forms that are not readily remineralized. This immobilization may induce N deficiency by reducing both pasture and animal production in some tropical pasture ecosystems (Robbins et al. 1989). It is expected that an increase in stocking rate or pasture burning, by reducing the

Table 6.7 Mineralization indices and N trace gas emissions for soils from primary forest, secondary forest, reformed pasture, and degraded pasture at Fazenda Vitoria.*

Land Use	**Net N Mineralization** (mg N g^{-1} 7d^{-1})**	**Net Nitrification** (mg N g^{-1} 7d^{-1})**	**N_2O Emissions*** (kg N ha^{-1} yr^{-1})**	**NO Emissions*** (mg N g^{-1} 7d^{-1})**
Primary forest	2.57 (0.62) a	2.25 (0.22) a	2.5 (0.7) a	1.2 (0.3) a
Secondary forest	1.55 (0.39) ab	0.30 (0.29) bc	0.9 (0.2) b	0.3 (0.1) b
Improved pasture	0.97 (0.09) bc	0.46 (0.09) b	0.3 (0.3) b	0.8 (0.3) b
Degraded pasture	0.33 (0.11) c	-0.22 (0.05) c	0.0 (0.1) b	0.7 (0.3) b

Source: L. V. Verchot and E. A. Davidson, unpublished data.

* Means and (s.e.) within a column followed by the same letter are not statistically different at = 0.05 by Duncan's Multiple Range test.

** Net N mineralization and net nitrification values were obtained from laboratory incubations (7 days) at room temperature.

*** Annual trace gas emissions are extrapolated from measurements on 9 dates during the wet season and 6 dates during the dry season.

amount of litter, can reverse this trend, but this mobilization of N would also make the N in the system more prone to loss from the ecosystem.

Nitrogen losses due to the direct effect of the grazing ruminant have usually not been accounted for in studies of N dynamics in active Amazonian pastures (but see Buschbacher 1987). Cattle interfere in the internal N cycle of pasture plants, alter nitrogen distribution in the pasture area, and change botanical composition, and these effects are important for the N cycle of active pastures. Much of the N loss that occurs in active pastures is accounted for by the heavy deposition of excreta around campsites, drinking and mineralization troughs, and cattle trails. The greatest potential for N loss are in urine patches which may receive very high amounts of N (> 300 kg N ha^{-1}). Nitrogen returned in urine is rapidly hydrolyzed to ammonia, becoming available for plant uptake. However, because of the very high amount of N in urine patches, plant uptake and utilization is usually not efficient, and large amounts of N may be lost through ammonia volatilization, which is further enhanced by the high temperatures at ground level characteristic of tropical pastures. As pasture utilization increases it is expected that N losses from the system would also increase as a result of more N passing through the animal and being excreted.

For a typical active *B. brizantha* beef cattle pasture in Brazilian Amazonia with an average carrying capacity of 0.8 animal units per hectare per, the amount of N that would be converted in live weight gain and potentially exported from the system in one year as animal product would be about 7 kg N ha^{-1} (based on a body composition of 2% N for beef cattle). Assuming an annual herbage consumption of about 3300 kg (9 kgd^{-1} x 365d) of *B. brizantha* dry matter per hectare, we can estimate that 33 kg of N are ingested by the animal in one year (based on an average N content of 1% in *B. brizantha* dry matter). Considering that only about 10% of the N ingested by the animal is retained, 30 kg of the ingested N is excreted in urine and dung. If a 30% recovery of excreta N by pasture plants is assumed (Ball et al. 1979, Ledgard and Sanders 1982, Ledgard et al. 1982), then about 20 kg of N returned to each hectare of pasture via excreta could potentially be lost during the period of one year through volatilization of ammonia, leaching, erosion, and denitrification. Since N fertilization and protein or urea supplementation are not common management practices in beef cattle pasture systems in the Amazon basin, N inputs are mostly restricted to biological N fixation and additions in precipitation. Loss of N from the system must be balanced by atmospheric deposition, N fixation, or depletion of soil N through mineralization of soil organic matter.

Increases in emissions of NO and N_2O have been observed in young tropical cattle pastures relative to forests (Keller et al. 1993, Luizão et al. 1992), which is due to an initial pulse of net N mineralization following forest clearing that provides mineral N for nitrifying and denitrifying bacteria (Firestone and Davidson, 1989). However, Keller and Reiners (1994) found that N gas emissions declined below levels of the primary forest in pastures that were older than 10 years in the Atlantic coastal plain of Costa Rica. There are indications that the period of enhanced N gas emissions in young pastures of the Amazon may be more brief (i.e., only 1–3 years) than was observed in Costa Rica (personal communication of preliminary results from Rondônia by Paul Steudler and from Pará by Louis Verchot). The reason for this difference is unknown, but the volcanic soils of Costa Rica may be more fertile with respect to N availability. If it is true that N_2O emissions from soils of most Amazonian pastures that are > 3 years old are lower than emissions from the primary forests (Table 6.7), then one effect of deforestation at the basin scale may be a net reduction in N_2O emissions.

Seasonality of precipitation is also an important factor controlling N trace gas emissions from Amazonian soils. At Paragominas in the highly seasonal eastern Amazon, N_2O is the most important N gas during the wet season when diffusivity is restricted, thus favoring the more reduced N gas form (Table 6.8). During the dry season, the more oxidized NO form is the dominant N gas emitted. Differences in total N gas emissions among land uses is controlled by N availability as indicated by net N mineralization and net nitrification (Table 6.7), while the seasonal variation in soil moisture content provides the controls over the relative proportions of NO and N_2O emissions (Table 6.8).

In addition to affecting trace gas emissions, lower N availability in old pastures relative to primary forests may also have implications for stream water chemistry. Preliminary results from studies in Rondônia and Pará indicate very low concentrations of NO_3 in first-order streams draining pastures (C. Neill and D. Markewitz, personal communication).

In summary, N cycling processes clearly change when forests are converted to cattle pastures. In Amazonia, soil N availability generally declines (except perhaps during a brief period of high N availability immediately after clearing), which results in lower emissions of N trace gases and probably lower hydrologic export of NO_3. Direct export of N via the cattle is estimated at about 7 kg N ha^{-1}. Cattle are also important in redistributing N within the pasture by concentrating N in excreta around campsites, drinking troughs, and cattle trails, where the potential for further N loss may be as high as 20 kg N ha^{-1}. It is not clear whether the N needs of pasture grasses are provided by associative N fixation or if the most successful introduced grass species are simply well adapted to obtaining N from nutrient-poor soils. Similarly, it is unclear whether reduced N availability in old pasture soils limits rates of regrowth of secondary forests upon pasture abandonment, or if N fixation rates in young secondary forests keep up with demand for N while some other factors limit forest regrowth. For much of the Amazon basin, N fertilization does not appear necessary, but nevertheless, the N nutrition of pasture grasses needs to be better understood for developing sustainable management practices. Moreover, altered N cycling processes following land use change clearly affect the exchange of N between the atmosphere and the biosphere and between terrestrial and aquatic ecosystems.

Table 6.8 Preliminary results for seasonal variation in N gas emissions at Fazenda Vitoria, Paragominas, Brazil, 1995.

	March(Wet Season)		**September(DrySeason)**	
	N_2O (ng N cm^{-2} hr^{-1})	**NO (ng N cm^{-2} hr^{-1})**	**N_2O (ng N cm^{-2} hr^{-1})**	**NO (ng N cm^{-2} hr^{-1})**
Primary forest	2.6	1.0	0.6	1.5
Secondary forest	1.6	0.4	0.3	0.3
Active pasture	1.0	0.9	0.1	0.2
Degraded pasture	0.7	0.5	0.1	0.4

Source: L. V. Verchot and E. A. Davidson, unpublished data.

Acidity and Base Cations

The effective cation exchange capacity (ECEC) of the variable-charge kaolinitic and oxic soils of the Amazon is typically low, and what little ECEC exists is pH dependent. At pH 4 to 5 (in water), which is characteristic of most of the primary forest soils, the ECEC is often in the range of 2–3 cmol charge per 100 g topsoil. Depending on the soil type, 20–80% of the ECEC may be base saturated (with the lower end of this range being more common), and the remainder is exchangeable acidity (mostly aluminum). When the forest is cleared and the site is burned, the pH of the new pasture soil often increases to 6 or 7. The ECEC often doubles and the base saturation increases. In effect, a significant fraction of the base cations (Ca, Mg, K) that were present in the above ground biomass of the forest is retained in the soil as a result of the increased ECEC caused by the change in pH of these variable charge soils. While observations of increased base cation content in the soil following fire are common (Andreux and Cerri 1989, Ewel et al. 1981, Kaufman et al. 1995, Moraes et al. 1996, Sanchez et al. 1983) cation budgets that trace masses of nutrients from the primary forest biomass to the soil and then to pasture or secondary forest vegetation are lacking. Kaufman et al. (1995) calculate that < 10% of the base cations in the aboveground forest biomass are lost during forest clearing and burning in Pará and Rondônia, but they caution that most of the cations were found in the ash immediately after the fire, which could be prone to subsequent loss by erosion. On the other hand, if the ash becomes incorporated into the soil, the increased pH-dependent ECEC of the soil may be sufficient to retain those nutrients.

Studying chronosequences of pastures in Pará and Mato Grosso, Falesi (1976) found that the pH, ECEC, and base saturation of the topsoil remained elevated in all pastures. The oldest pasture he studied was 11 years. These results indicate that, provided serious erosion is avoided by good pasture management, base cations can be retained. A similar result was observed by Moraes et al. (1996) in two pasture chronosequences in Rondônia. The pH and the sum of base cations remained elevated in pasture soils relative to the forest in all cases except an 81-year-old pasture, where the pH was still slightly elevated and the sum of base cations was about the same as the primary forest. They also show a trend of gradual decreasing pH and sum of base cations in pastures older than 5 years. Whether the gradual loss of base cations is due to leaching, erosion, repeated fires, or harvest exports is not known. Hence, base cations are not retained by the soil indefinitely, but it appears that losses often do not become significant for a decade or more. Fertilization trials conducted throughout the Amazon basin have shown no significant responses of unproductive (i.e., degraded) pasture areas, ranging from 8 to 13 years old, to liming or potassium amendments (Dias Filho and Serrão 1987, Serrão et al. 1979), reflecting the adequate pH and base saturation status of these areas.

As in the case for P and N, cations are obtained by foraging over a large area and are then deposited as excreta on a small area, thereby becoming concentrated, principally near waterers, mineral feeders, and campsites. Potassium is mostly excreted in the urine, while the major pathway of excretion for calcium and magnesium is in the feces (reviewed by Barrow 1987). About 90% (circa 70 kg ha^{-1} yr^{-1} for a *B. brizantha* pasture with a medium productivity) of the ingested potassium is returned to the soil via excreta in an immediately available form (K). Because it is excreted in a soluble form, highly concentrated in urine patches, small amounts of K can be lost from soil through leaching if it is not efficiently absorbed by roots or retained in the soil. Pasture overgrazing and the premature grazing of recently burned areas can greatly contribute to increase these losses by directly interfering in the uptake efficiency by pasture species and by increasing the potential of erosion and leaching. Periodic pasture burning could aid in the cycling of base cations, particularly the less mobile Ca that is immobilized in slash (woody residue from forest), litter, and standing weedy vegetation. On the other hand, repeated fires or fires too late in the dry season, when pasture recovery would be slow, could make base cations (and other nutrients) more prone to loss by erosion.

For beef cattle pasture systems in the Amazon basin, only calcium, among the base cations, is normally added to the system via pasture management practices. This is because animal supplementation with bone meal or bicalcium phosphate is commonly practiced in most beef cattle systems throughout Amazonia. For a typical active pasture with a medium carrying capacity, the amount of calcium entering the system in one year through mineral supplementation would be around 4.0 kg per hectare which is close to the amount expected to be exported in animal products.

We are not aware of published results on the changes in soil acidity and base cation retention in secondary forests of the Amazon. Preliminary results of a study of a 20-year-old forest at Paragominas indicate that soil pH, ECEC, and base saturation remain elevated in the top 20 cm of mineral soil (D. Markewitz and E. A. Davidson, unpublished data). However, this forest is one of the sites classified by Uhl et al. (1988) has having had "light use" while in the pasture phase. It is possible that more intensive use of pastures that might result in greater harvest export, greater erosion, or a longer period for leaching may result in depleted base cation stocks upon pasture abandonment. Woody forest vegetation requires large quantities of cations, particularly Ca, but we do not know if the rate of secondary forest growth is limited by the availability of these nutrients. Intuitively, it seems that growth of pasture and forest vegetation on highly weathered, acidic soils with low cation exchange capacity should be limited by the availability of base cations, but Amazonian pasture soils appear surprisingly capable of retaining base cations, and the same may be true for soils of secondary forests.

Hydrological Cycle

The modification of natural ecosystems, as when all the plant cover is substituted during conversion of forests to cattle pastures, affects the transport of water within these systems. These changes result from modifications of turbulence, radiation balances, temperature, and humidity. Pasture canopies are more uniform than forests, have a more smooth surface, and, therefore, pastures have a different microclimate (Grace 1994). The difference in the conditions between pastures and forests are greatest in the dry season. Measurements made above the canopy in active pastures of ***B. brizantha*** in Ji-Paraná, Rondônia, during the dry season show that the specific humidity deficit (D) above the surface in an area of pasture is higher (19.0 g kg^{-1}), than that of a forest area (12.5 g kg^{-1}) (McWilliam et al. 1996). Close to the soil surface, D is even higher. In a degraded pasture in the region of Paragominas, mean maximum values of D 10 cm from the soil surface were in the order of 24.7 g kg^{-1} in the dry season, while below the canopy of an adjacent primary forest these values were not above 6.0 g kg^{-1}. The higher values of D in areas of degraded pastures were a result of the high temperatures and low air humidity typical of these areas during the dry season (Nepstad et al. 1996). During the rainy season, the maximum temperatures and D are similar between forest areas and abandoned pastures (Nepstad 1989). Studies conducted during many years in Amazonia show that the substitution of forest by pastures has caused a reduction of 11% in net radiation, due mainly to differences of albedo and long-wave net radiation (Wright et al. 1992, Culf et al. 1996).

Mobilization of water from the soil is closely related to root depth and root density in each layer of soil. Fine roots of active *B. brizantha* pastures, established in deeply weathered clayey soils in eastern Amazonia, reach depths of 8 m or more (Nepstad et al. 1994). In abandoned pastures (50% *B. humidicola* and *P. maximum* cover and 50% invading shrubs and small trees), fine roots (< 1 mm in diameter) were found at depths of 12 m (Nepstad et al. 1994). Fine-root biomass in the superficial soil layers of an active pasture in Paragominas, eastern Amazonia, was 3 times higher than that found in an adjacent primary forest area. Fine root biomass in the active pasture decreased by a factor of 100 between the surface and 6 m depth. In an abandoned pasture area, the distribution pattern of fine-root biomass was similar to that observed in the deeper soil layers of the forest ecosystem. This pattern is associated with the fine roots of the existing dicotyledonous invading species.

The monitoring of the soil plant available water (PAW) between 0 and 8 m depth in the soil has shown that during the dry season, in active *B. brizantha* pastures areas, a considerable use of water reserves occurs primarily in the top 2 m of soil. This is similar to observations made in primary and secondary forest areas. However, below 2 m, the depletion of the soil water reserves was greater in the forest ecosystem (Jipp et al. 1998). In general, active pasture ecosystems have a greater proportion of fine roots in soil layers between 0–2 meters, and the water in these layers is depleted more quickly, while a major part of the water reserve in the soil is stored in deeper layers. As the pasture is invaded by dicotyledonous species, which often have deeper root systems, nutrient cycling may be intensified as water reserves (and nutrients) in deep soil layers are accessed. For example, the surface soil below *Cordia multispicata* plants in degraded pastures is richer in Ca, Mg, and K relative to surface soil under other species as a result of inputs from the nutrient rich *Cordia* litter (Vieira et al. 1994). It has also been shown that in *Cecropia palmata*, a pioneer species that maintains high stomatal conductance, P levels in leaf tissue are always high, even in dry periods (Denich 1989). These results suggest that the ability of these species to access soil water and maintain a transpirational flux may also be influencing their ability to absorb soil nutrients.

The progression of plant succession through the formation of "vegetation islands" and later secondary forest greatly changes the hydrological cycle at a local scale. This can be demonstrated by the similarity between the water depletion profiles obtained in secondary and primary forests (Jipp et al. 1998).

If the reduction in plant-available water below 4 m depth in pastures is small and the losses of rainwater through grass canopy interception are also low, it is possible that there is a greater infiltration to the water table from pasture soils, as well as nutrient losses through erosion and runoff during the rainy season (Dunin 1987, Nepstad et al. 1994). Most of the water lost to the atmosphere in the pasture system is subjected to stomatal regulation, which is influenced by environmental variables such as available water in the soil, and D. In Manaus, central Amazonia, at the beginning of the dry season, very high values for stomatal conductance (gs) at mid day were observed in the forage grass *B. decumbens*. These reached extremes of 1.0 mol m^{-2} s^{-1}, without a correlation between gs and D. However, these values fell drastically in response to reductions of soil available water (Roberts et al. 1996). On the other hand, the values for maximum gs obtained in pastures of *P. maximum* during the rainy season in Marabá, eastern Amazonia, rarely exceed 0.40 mol m^{-2} s^{-1}, in the period between 10:00 am and 2:00 pm. The most frequently observed values were around 0.25 mol m^{-2} s^{-1}. Stomatal behavior of P. maximum showed a negative correlation with an increase of D in the air in the dry season. The lowest values of gs (0.10 mol m^{-2} s^{-1}) were observed in periods when D was greater than 15 g kg^{-1}. In *B. brizantha* pastures, no clear relationship between gs and D was observed. However, the effect of the reduction of soil available water was drastic, causing the gs close to midday to be reduced from values of 0.62 mol m^{-2} s^{-1} to only 0.17 mol m^{-2} s^{-1} at the beginning of the dry season (McWilliam et al. 1996).

Under normal conditions of water availability, values for LAI in pastures of *B. brizantha* have been measured above 4.0. However with the establishment of a water deficit in the soil, these values decrease to below two or even lower in pastures of *P. maximum* (Roberts et al. 1996). A similar situation is found in abandoned pastures in eastern Amazonia, where a reduction of approximately 68% of green tissue has been observed in the dry season, while in an adjacent area of primary forest this reduction was only 16% (Nepstad et al. 1994). Primary forests, which have deep root systems and little seasonal variation in LAI, maintain stable subcanopy microclimatic conditions and transpirational flux, even during the dry season. Because of an evergreen forest canopy, the return of the rainy season has less impact on the microclimate near the soil in the forest than in the pastures, and the deep soil water stores are also more efficiently recharged in the forest.

In the active pastures, the response to environmental water deficits can be influenced by the grass species. In general, however, gs values are normally high when water supply is adequate, but decreases abruptly in situations when both high evapotranspiration demand and low rainfall occur. Despite stomatal control of water vapor loss, the drying of most aboveground pasture biomass is usually observed in areas with an intense dry season. This leads to a dramatic reduction in the flux to the atmosphere of water stored in soil layers below 4 m. This desiccation and reduction in LAI exposes pasture areas to more severe conditions of temperature, wind, and humidity, which consequently cause greater disposition to fire. After the beginning of the rainy season, the upper layers of the pasture soil become saturated quickly, creating a risk of surface runoff, erosion of sediments and nutrients to rivers, and increased water loss to groundwater seepage. In abandoned pastures invaded by deep-rooted dicotyledonous shrubs, some of the plant-available water below the rooting depth of the grasses can be transpired. As secondary forests grow, the hydrologic characteristics of the forest become more like those of the primary forest.

Conclusions

The biogeochemistry of active pastures is perhaps one of the least studied processes within managed systems of the Amazon basin. Although the dynamics of the major nutrients in this ecosystem are fairly well known, mostly through agronomic trials conducted during the last two decades, the biogeochemical processes governing these dynamics are still poorly understood. One of the greatest gaps in understanding the biogeochemistry of active pasture systems concerns the role of the grazing cattle in affecting the nutrient cycling processes.

Recent studies conducted in active Amazon basin pastures have emphasized the dynamics of N and C without paying sufficient attention to important pasture management issues like stocking rate or the frequency (and methods) of weed control. Phosphorus, probably the most important element for pasture primary production in the Amazon basin, has received far less attention than N and C. Most of what we know about phosphorus cycling in active Amazonian pastures is derived from agronomic trials conducted in the Brazilian Amazon and extrapolation of research results obtained in other regions.

The increased vulnerability of bioactive elements to loss from the pasture ecosystem is often believed to be one of the greatest differences between this and the antecedent primary forest ecosystem. While this is true during the pasture establishment phase, when the soil is left unprotected from rain, wind, and excessive solar heat, it has been observed that well-managed pastures are usually capable of maintaining soil nutrients at rather constant levels through time, the exception being available phosphorus. In many circumstances, low-productivity pastures can be reclaimed with relatively small additions of phosphate fertilizer, strongly suggesting that all other major plant nutrients (and soil pH) are present in adequate levels in the soil.

While phosphorus nutrition can be met relatively easily with fertilization, another challenge for developing sustainable management practices in pastures of the Amazon basin is to balance the need for and means of weed control in order to avoid land degradation. Weed control for pasture reclamation often includes the use of herbicides and/or the removal of tree root biomass by bulldozers, which prevents resprouting of treelets. This practice, along with frequent use of fire, may very likely delay the rate of regrowth of secondary forests, if the pasture is eventually abandoned. Hence, while some management practices, like maintaining good ground cover and minimizing erosion, are salutary for both agroecosystems and successional forest ecosystems, other practices, like bulldozing and herbiciding, may slow ecosystem recovery and commit the site to long-term dependency on intensive inputs to maintain pasture productivity. Intensification of pasture management, if done soundly, may lead to sustainable and permanent pasture usage for some areas. If done poorly, it may lead to impoverished and degraded lands.

As abandoned pasture areas become increasingly invaded by shrubs and small trees, which in time lead to secondary forest formation, the biogeochemical cycles of plant nutrients and the hydrological cycle are expected, eventually, to resemble the cycles originally found in the primary forest. The rate of recuperation of biogeochemical cycles in secondary forests and the factors that influence those rates, however, deserve further attention.

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