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Fire in the Brazilian Amazon
2. Biomass, nutrient pools and losses in cattle pastures

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Abstract Conversion to cattle pasture is the most common fate of the ≈426,000 km^2 of tropical forest that has been deforested in the Brazilian Amazon. Yet little is known about the biomass, C, nutrient pools, or their responses to the frequent fires occurring in these pastures. We sampled biomass, nutrient pools and their losses or transformation during fire in three Amazonian cattle pastures with typical, but different, land-use histories. Total aboveground biomass (TAGB) ranged from to 53 to 119 Mg ha^{-1}. Residual wood debris from the forests that formally occupied the sites composed the majority of TAGB (47–87%). Biomass of fine fuels, principally pasture grasses, was ≈16–29 Mg ha^{-1}. Grasses contained as much as 52% of the aboveground K pool and the grass and litter components combined composed as much as 88% of the aboveground P pool. Fires consumed 21–84% of the TAGB. Losses of C to the atmosphere ranged from 11 to 21 Mg ha^{-1} and N losses ranged from 205 to 261 kg ha^{-1}. Losses of S, P, Ca, and K were < 33 kg ha^{-1}. There were no changes in surface soil (0–10 cm) nutrient concentration in pastures compared to adjacent primary forests. Fires occur frequently in cattle pastures (i.e., about every 2 years) and pastures are now likely the most common type of land burned in Amazonia. The first 6 years of a pastures existence would likely include the primary forest slash fire and three pasture fires. Based upon our results, the cumulative losses of N from these fires would be 1935 kg ha^{-1} (equivalent to 94% of the aboveground pool of primary forest). Postfire aboveground C pools in old pastures are as low as 3% of those in adjacent primary forest. The initial primary forest slash fire and the repeated fires occurring in the pastures result in the majority of aboveground C and nutrient pools being released via combustion processes rather than decomposition processes.

Key words Cattle pastures · Deforestation · Nutrient cycling · Biomass burning · Tropical forests

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Introduction

By 1991, an estimated 426,000 km^2 of forests had been cleared in the Brazilian Amazon (data from the Brazilian Space Agency-INPE reported in Fearnside 1993a, b). Annual rates of deforestation ranged from 15,000 to 22,000 km^2 year^{-1} up to 1988 and from 11,100 to 19,000 km year^{-1} since that time (Fearnside 1993b; Skole and Tucker 1993). The dominant land use following forest clearing has been for cattle pasture (Fearnside 1988; Uhl et al. 1988). Fires are set to both create and maintain cattle pastures. Because of the areal extent and frequency of burning, pasture fires are likely significant sources of CO_2 and other greenhouse gasses and aerosols.

Although estimates of aboveground biomass and C pools of Amazon forests are available (e.g., Brown and Lugo 1992; Fearnside et al. 1993; Kauffman et al. 1995), few studies have quantified these parameters in cattle pastures. Uhl and Kauffman (1990) reported that the total aboveground biomass (TAGB) of an old cattle pasture in the eastern Amazon was 52 Mg ha^{-1}. In modeling studies of C dynamics, Detwiler and Hall (1988) and Houghten et al. (1991) utilized a pasture biomass estimate of 10–11 Mg ha^{-1} while Fearnside (1996) used biomass estimates of 11 Mg ha^{-1} for productive and 8 Mg ha^{-1} for degraded pasture. Work on the quantification of the effects of fire on biomass and nutrient pools in cattle pastures is similarly scanty. Fire effects have been reported in slash fires in Brazilian dry forest (Kauffman et al. 1993), Brazilian Cerrado and savannas (Kauffman et al. 1994; Castro 1995) and in the

Brazilian Amazon (Fearnside et al. 1993; Kauffman et al. 1995). In slashed primary forests of Amazonia, Kauffman et al. (1995) reported that TAGB ranged from 290 to 435 Mg ha^{-1}; ≈50% of the TAGB was consumed by fire. There are no similar published studies of pasture fires in the Brazilian Amazon.

Patterns of land use are variable in the Brazilian Amazon (Fearnside 1988; Houghton 1991). Following the initial slash fires of primary forest, lands are often converted directly to pasture, or they may be used in shifting cultivation prior to conversion to pasture. Upon establishment of the pasture, fires are set by local landowners every 2–3 years to favor dominance of pasture grasses over invading or residual vegetation as well as to enhance the palatability and growth of grasses (Uhl et al. 1988; Fearnside 1992). Accidental fires are common in regions of the Amazon characterized as a mosaic of forest fragments and active and abandoned pastures (Uhl and Buschbacher 1985). Pasture fires are now likely to be the most prevalent type of fire in the Brazilian Amazon. Therefore, it is important to quantify the dynamics associated with biomass and nutrient pools in pastures. The objectives of this study were to: (1) quantify the TAGB of pastures of varying age and land use histories; (2) quantify the nutrient and C pools in aboveground biomass and soil surface; (3) quantify losses and redistribution of biomass, C and nutrient pools due to fire; and (4) compare the mass of C and nutrient pools in pastures with those of adjacent primary forests and project the cumulative effects of repeated burns on these pools.

Methods

Study area

Our study areas were located in the Brazilian states of Rondônia and Pará, regions that are undergoing rapid deforestation (Skole and Tucker 1993; Fearnside 1992). The intact primary forests of these areas are classified as Floresta Tropical Perenifolia de Terra Firme (upland tropical evergreen forest) by Eiten (1983) or as Floresta Ombrofila densa submontana (Pará) and Floresta Ombrofila aberta submontana (Rondônia) (i.e., dense and open submontane tropical forests) by the Radam project (Brazil DNPM 1978; Fearnside 1992). Total aboveground biomass of forests in the proximity of the sampled pastures in this study was 292 and 435 Mg ha^{-1} in Pará and 290 and 361 Mg ha^{-1} in Rondônia (Kauffman et al. 1995).

We sampled three pastures with different land-use histories and we refer to the individual pastures by the names of the land owners. We obtained permission to take measurements and observe land use and burning practices from land owners already planning to burn the pastures in the years we conducted this study. We interviewed each land owner to obtain information on the land-use history and future objectives of management of the pastures.

We sampled the first and oldest pasture (Francisco) in 1991. This site is approximately 50 km south of the city of Marabá, Pará (5°21′S, 49°09′W). The primary forest of this site had been felled approximately 20 years prior to this study. Initially, the site was utilized as a cooperative farm under intensive cultivation. Following abandonment for many years, the present owner cut the second-growth forest and converted the site to pasture. The results reported here are from the first pasture fire, which occurred 3 years after the second-growth forest had been cut. However, there were at least two slash fires (the primary forest and the second-growth forest) prior to this fire. Additionally, there was evidence that much of the residual forest slash had been piled and burned.

The second and third pasture sites (Durval and João) were sample near the town of Jamarí, Rondônia in 1992 (9°12′S, 60°3′W). The Durval pasture was the youngest of pastures, having been formed from primary forest 4 years prior to this study. This site had previously experienced a slash fire of the primary forest and a pasture fire 2 years after deforestation. The João pasture was formed from third-growth forest 2 years prior to this study. This was the first pasture fire. However, in the 10 years since it was initially deforested, the João site had undergone two cycles of shifting cultivation prior to pasture formation. Prior to this fire, the site had been burned three times (a primary slash fire, a second-growth slash fire, and a third-growth slash fire). Following these slash fires, residual slash that could be moved by hand was piled and burned prior to planting.

Climatological data from the nearest stations are from Marabá, Pará and Porto Velho, Rondônia. Mean average precipitation of the two stations is 2088 and 2354 mm, respectively. There is a pronounced dry season from June to September with precipitation normally <100 mm during each of these months. Mean average temperatures are 26.1°C and 25.2°C, average minimum temperatures are 22.1°C and 20.9°C, average maximum temperatures are 31.7°C and 31.1°C, and mean relative humidity is 82% and 85%, respectively (Departamento Nacional de Meteorologia-Brasil 1992).

All sites were burned towards the end of the dry season (late August and September). The pastures were burned utilizing circle-fire ignition patterns, where the entire perimeter is ignited, causing the fire to burn most intensely towards the center. Sites were ignited when diurnal temperatures were warmest and relative humidity lowest (1200–1400 hours).

Aboveground biomass

We partitioned the TAGB on the basis of plant morphology, influences on fire behavior, value as a nutrient pool, and considerations of sampling approach. Biomass categories included residual wood debris from primary or secondary forests, litter (dormant grass, dead leaves, slashed palm fronds and all other non-wood dead organic materials), green grass, and live dicots (sprouts and seedlings).

We estimated the prefire mass of residual wood debris, the amounts consumed by fire, and the postfire mass of residual wood debris nondestructively utilizing planar intersect models (Brown and Roussopoulous 1974; Van Wagner 1968). This required collection of the average density and diameter of wood in each size category for each site. At each site, 32 planar intersect transects were systematically established to ensure sample dispersion through the pastures. All transects were marked with small aluminum stakes prior to burning for exact relocation and remeasurement following fire. Diameters of all wood particles intersecting each sample plane were measured. We partitioned the wood debris into standardized size classes based on their diameter. Wood particle diameter is a good predictor of the rate of moisture loss (e.g., a time-lag constant) and hence of relationships to combustion and fire behavior (Deeming et al. 1977). Diameter size classes have also been shown to vary inversely with nutrient concentrations and improve calculations of loss or redistribution by fire (Kauffman et al. 1993, 1994). The diameter classes used to partition wood debris were the same as those of Kauffman et al. (1995): 0–0.64 cm diameter, 0.65–2.54 cm diameter, 2.55–7.5 cm diameter, 7.6–20.5 cm diameter, and >20.5 cm diameter. For wood particles >7.5 cm diameter, we further separated them into sound and rotten classes. Palm wood was separated from other trees. Lengths of the sampling plane varied among the wood debris size classes: 1 m for wood particles ≤0.64 cm diameter, 2 m for wood debris 0.65–2.54 cm diameter, 5 m for wood debris 2.55–7.6 cm diameter, and 15 m for the coarse wood (i.e., logs >7.5 cm diameter). We measured the diameter of each coarse wood debris particle intercepting the plane to the nearest half centimeter. For the equations of biomass for the three wood debris size classes < 7.5 cm diameter, a quadratic mean diameter was calculated from measurement of 100 particles of each size class at each site. Thereafter, for these classes we counted the number of particles that intersected the sampling plane. We corrected for bias due to the angle and slope of wood debris as outlined in Van Wagner (1968) and Brown and Roussopoulous (1974). Thirty randomly collected samples of each size class were measured for specific gravity (particle density) at each site.

The biomass of litter, grass, seedlings, and sprouts was destructively sampled through collection of all materials in 25×25 cm microplots. We placed a microplot at the 2-m mark of each planar intersect transect (i.e., $n = 32$ plots per site). The mass of litter, green grass, and dicots within each microplot was separated, oven dried and weighed. Postfire mass of these components was sampled 2 m away from the prefire microplot.

We collected ash mass at the Francisco site by gently sweeping all ash in the postfire microplots into paper bags. At the Durval and João sites, ash mass was determined through collection within sixteen 50×50 cm microplots. At these sites we used a portable electric generator and vacuum cleaner to collect the ash within each microplot. The ash from each microplot was oven-dried and weighed.

Nutrient pools

Aboveground nutrient pools were partitioned in the same classes as aboveground biomass. Prior to burning, we collected five samples of each biomass component at each site. Each of these samples consisted of a composite mix of 10–20 collections of material. Ash samples were collected in the same manner following fire. At each site five soil samples at depths of 0–2.5 cm and 2.5–10 cm were also collected. At the Francisco site, we re-sampled postfire soils approximately 1 m away from the prefire soil sampling areas. Samples for the calculation of soil bulk density were collected in the same areas as nutrient samples. All samples were air-dried for at least 1 week in the field and then oven-dried prior to nutrient and C analysis in the laboratory.

All plant and ash samples were analyzed for total C, N, S, P, K, and Ca. Soils at the Durval and João sites were also analyzed for these elements. Only C and N were analyzed for soils at the Francisco site. Prior to analysis, plant and ash samples were ground to pass through a 40 mesh screen (0.5 mm) in a Wiley mill. Total N was determined from Kjehldahl digestion (Bremner and Mulvaney 1982). Total Ca, K, and S were determined by atomic absorption (Tabatabai and Bremner 1970). Total P was determined colorimetrically following wet digestion utilizing a Kjeldahl procedure (Watanabe and Olsen 1965). Total C was analyzed by the induction furnace method (Perkin-Elmer 2400 elemental analyzer for the Francisco site and a Carlo-Erba NA Series 1500 for the Durval and João sites) (Nelson and Sommers 1982). Organic matter of ash from the Durval and João sites was determined through complete combustion of samples at 500°C for 8 h in a muffle furnace (Davies 1974).

To determine the influences of pasture conversion on the mass and partitioning of nutrient pools, we compared the pastures in this study with adjacent primary forests. The primary forest nutrient pools were described in detail by Kauffman et al. (1995).

Fire behavior, weather, and moisture content

Moisture content of biomass, air temperature, relative humidity, and flame length were recorded at each site before and during the flaming phase of combustion. We calculated moisture content (dry-weight basis) through collection of five to ten samples of the following components: soil surface, dead grass, live grass, dicots, and wood > 7.6 cm diameter. Samples were weighed in the field with a portable digital balance. They were then oven dried at 60°C for 5–7 days to calculate dry weight. Air temperature and relative humidity at the time of ignition were measured on site with a sling psychrometer; wind speed was measured with a portable anemometer.

Differences in biomass, nutrient pools, postfire nutrient pools, ash, and nutrient losses between the pastures were tested through analysis of variance in a completely randomized design. If significant, the least significant difference multiple range test was utilized to determine statistical significance among the sites sampled ($P \leq 0.10$)

Results

Aboveground biomass

Although pastures may appear to have a uniform composition of exotic grasses from ground or remotely sensed observations, we found significant differences in the biomass and structure of the sampled pastures. Total aboveground biomass ranged from 53 Mg ha^{-1} at the Francisco site with the longest and most intensive land-use history to 119 Mg ha^{-1} at the relatively recently established pasture site of Durval (Table 1). The grasses were a relatively minor component of the biomass; residual wood debris originating from the primary forests comprised ≈87% of the TAGB at the Durval pasture and 73% at the João pasture. Wood mass remained a dominant component at the older Francisco pasture comprising 47% of the TAGB. There were significant differences in the residual wood debris along the chronological gradient from the youngest to the oldest pasture i.e., (104 Mg ha^{-1} at the Durval pasture, 53 Mg ha^{-1} at the João pasture, and 25 Mg ha^{-1} at the Francisco pasture).

The biomass of all herbaceous materials combined was 29, 15, and 19 Mg ha^{-1}, at the Francisco, Durval, and João pastures, respectively. The litter/dead grass component at the Francisco pasture was significantly higher (24 Mg ha^{-1}) than the others (< 17 Mg ha^{-1}). This was likely a result of the prevalence of slashed palm fronds, as well as a longer period of time since the last burn occurred on the site (3 vs. 2 years).

Fire behavior and biomass consumption

All fires occurred towards the end of the dry season, and after 2–7 rainless days which allowed the pastures a sufficient time to dry to a combustible level (Table 2). Air temperature during the three pasture fires ranged from 30 to 34°C and relative humidity ranged from 38 to 55%. Dead grass, the ecosystem component that facilitates sustained ignition and fire spread in pastures, had a moisture content of 2.7 and 6.0% at the João and Francisco sites, respectively. In contrast, live grass moisture content was > 209% in these areas.

Flame lengths of these pasture fires were highly variable, ranging from 0.5 to 4 m among sites (Table 2). Flaming combustion was short-lived; total pasture

Table 1 Total aboveground biomass (Mg ha^{-1}) prior to, and following biomass burning in cattle pastures of Pará and Rondônia, Brazil. Pastures are arranged in order of increasing time since initially deforested. The Durval site had been deforested 4 years prior to sampling, the João site ≈10 years, and the Francisco site > 20 years. Numbers are mean ± SE. Different *superscript capital letters* denote a significant difference in biomass among the three pasture sites prior to fire. Different *superscript lower case letters* indicate a significant difference in biomass among the three pastures after fire

	Durval, Rondônia		João, Rondônia		Francisco, Pará	
	Prefire	Postfire	Prefire	Postfire	Prefire	Postfire
Litter/dead grass	11.62 ± 1.21^{B}	0.09 ± 0.04^{a}	16.95 ± 1.61^{B}	0.27 ± 0.19^{a}	23.65 ± 2.37^{A}	0.03 ± 0.02^{a}
Grass	3.76 ± 0.67^{A}	0.46 ± 0.17^{b}	2.55 ± 0.52^{A}	0.07 ± 0.04^{a}	4.24 ± 3.19^{A}	0.00 ± 0.00^{a}
Dicots	0.30 ± 0.30^{B}	0.00 ± 0.00^{a}	0.10 ± 0.10^{B}	0.01 ± 0.01^{a}	1.50 ± 0.52^{A}	0.24 ± 0.15^{a}
Wood debris (diam cm)						
0–0.64	0.13 ± 0.10^{A}	0.01 ± 0.01^{a}	0.01 ± 0.01^{B}	0.01 ± 0.01^{a}	0.60 ± 0.18^{A}	0.05 ± 0.05^{a}
0.65–2.54	0.37 ± 0.26^{B}	0.52 ± 0.42^{a}	0.26 ± 0.11^{B}	0.79 ± 0.05^{a}	1.51 ± 0.57^{A}	0.41 ± 0.23^{a}
2.55–7.6	6.12 ± 1.51^{A}	4.40 ± 1.42^{b}	0.27 ± 0.15^{B}	0.09 ± 0.09^{c}	4.20 ± 0.88^{A}	0.84 ± 0.28^{a}
7.6–20.5 sound	37.36 ± 7.74^{B}	29.73 ± 6.34^{b}	5.99 ± 1.33^{A}	5.71 ± 1.55^{a}	6.73 ± 1.83^{A}	1.96 ± 0.72^{a}
7.6–20.5 rotten	1.64 ± 1.04^{A}	2.03 ± 1.06^{b}	1.00 ± 0.48^{A}	0.58 ± 0.41^{c}	0.36 ± 0.22^{A}	0.00 ± 0.00^{a}
> 20.5 sound	57.94 ± 28.64^{B}	57.49 ± 31.56^{b}	45.53 ± 12.65^{B}	31.43 ± 8.99^{b}	8.24 ± 3.17^{A}	5.26 ± 2.08^{a}
> 20.5 rotten	0.00 ± 0.00^{A}	0.00 ± 0.00^{a}	0.00 ± 0.00^{A}	0.00 ± 0.00^{a}	1.35 ± 1.35^{A}	0.00 ± 0.00^{a}
Palms	0.00 ± 0.00^{A}	0.00 ± 0.00^{a}	0.00 ± 0.00^{A}	0.00 ± 0.00^{a}	1.41 ± 1.01^{A}	0.00 ± 0.00^{a}
Total wood	103.56 ± 35.10^{B}	94.18 ± 35.22^{b}	53.06 ± 12.34^{C}	37.89 ± 8.96^{c}	25.39 ± 4.07^{A}	8.52 ± 1.98^{a}
Total	119.24 ± 35.02^{B}	94.74 ± 35.18^{b}	72.67 ± 13.06^{B}	38.24 ± 8.96^{c}	53.31 ± 4.84^{A}	8.78 ± 1.98^{a}
Ash		2.05 ± 0.17^{b}		3.00 ± 0.28^{c}		4.78 ± 0.56^{a}

Table 2 General weather conditions, moisture content of selected fuels (mean ± SE), and range in flame length during fires in cattle pastures converted from tropical moist forest, Pará and Rondônia, Brazil

	Durval, Rondônia	João, Rondônia	Francisco, Pará
Date of burn	25 September 1992	31 August 1992	9 September 1991
Temperature (°C)	30	32	34
Relative humidity (%)	46	55	38
Wind speed (kph)	0–6	0–8	1–8
Moisture content (%)			
Soil surface	–[a]	17.8 ± 2.5	2.0 ± 0.5
Dicots	–	181.5 ± 17.0	–
Wood > 7.6 cm diam	–	8.2 ± 1.0	7.5 ± 2.0
Litter/dead grass	–	2.7 ± 0.7	6.0 ± 0.5
Live grass	–	225.0 ± 16.7	209.1 ± 11.5
Flame length (m)	1–4	3–4	0.5–4

[a] – No data

coverage of the fire always required < 2 h. However, smoldering combustion of residual wood debris often lasted for > 3 days.

The proportion of total aboveground biomass consumed by fire (the combustion factor) ranged from 21% at the Durval pasture to 84% at the Francisco pasture. This highly variable level of combustion resulted in significant and dramatic differences in postfire TAGB (Table 1). The postfire TAGB largely consisted of residual uncombusted wood debris ranging from 8.8 Mg ha^{-1} at the Francisco pasture to 94.7 Mg ha^{-1} at the Durval pasture. Virtually all (> 96%) of the fine herbaceous biomass was consumed by fire. In contrast, the consumption of wood debris by fire was only 9% at the Durval site and 29% at the João site. However, dry conditions and a greater composition of the fine wood fractions contributed to a total wood consumption rate of 66% at the Francisco site. Sound residual coarse wood debris (> 20.5 cm diam.) represented that remaining fraction of the primary forest wood component most resistant to both decomposition and consumption. While deforested ≈20 years prior to this study, mass of this residual coarse wood component was 8 Mg ha^{-1} at the Francisco pasture, significantly lower than that of the younger Durval or João pastures (58 and 46 Mg ha^{-1}, respectively). Because of the scattered arrangement and high density of the residual coarse wood component, consumption was only ≈1% at the Durval pasture, 31% at the João pasture and 36% at the Francisco pasture (Table 1). Ash was the only other significant postfire aboveground nutrient or organic matter pool. Ash mass significantly differed among the three sites; ash mass increased with increasing biomass consumption by fire (Table 1).

Aboveground nutrient pools

Although we sampled elemental concentrations of all sites separately, only mean concentrations of the three pastures are given in Table 3. Concentrations of all elements (except C) were typically highest in the grass and dicot components of the pastures (Table 3). In contrast to the nutrients, the concentration of C was slightly lower in grass and litter fractions (≈46%) compared to dicot and wood fractions (≈50%). Concentrations of

Table 3 Mean nutrient concentrations of aboveground biomass in Amazonian cattle pastures. Numbers are mean ± SE of all samples combined from pasture sites in Pará and Rondônia, Brazil

Component	Carbon (%)	Nitrogen ($mg\ g^{-1}$)	Sulphur ($mg\ g^{-1}$)	Phosphorus ($mg\ g^{-1}$)	Potassium ($mg\ g^{-1}$)	Calcium ($mg\ g^{-1}$)
Litter/dead grass	45.40 ± 0.75	8.48 ± 2.02	1.10 ± 0.10	0.63 ± 0.17	3.75 ± 1.12	3.22 ± 0.08
Grass-prefire	45.91 ± 0.67	13.06 ± 1.02	1.43 ± 0.15	1.11 ± 0.22	15.81 ± 4.49	1.70 ± 0.25
Grass-postfire	46.60 ± 0.04	7.01 ± 0.37	1.25 ± 0.07	0.78 ± 0.02	15.59 ± 0.98	0.90 ± 0.06
Dicots-prefire	49.65 ± 0.72	14.85 ± 2.35	2.04 ± 0.23	1.13 ± 0.35	10.60 ± 0.50	4.79 ± 0.51
Dicots-postfire	49.16 ± 0.21	7.54 ± 0.94	1.30 ± 0.10	1.25 ± 0.10	8.35 ± 0.89	6.13 ± 0.47
Wood debris (cm diam)						
0–0.64	49.17 ± 0.30	7.71 ± 0.20	0.83 ± 0.17	0.63 ± 0.34	4.51 ± 1.60	4.53 ± 0.48
0.65–2.54	49.71 ± 0.22	5.25 ± 0.70	0.53 ± 0.03	0.33 ± 0.09	3.28 ± 0.51	2.62 ± 0.57
2.55–7.6	50.00 ± 0.16	3.88 ± 0.96	0.41 ± 0.21	0.19 ± 0.07	1.65 ± 0.78	1.27 ± 0.70
> 7.6 sound	50.69 ± 0.30	3.82 ± 0.34	0.47 ± 0.05	0.19 ± 0.10	0.82 ± 0.44	1.29 ± 0.75
> 7.6 rotten	49.07 ± 0.70	5.93 ± 0.44	0.58 ± 0.07	0.12 ± 0.01	0.77 ± 0.11	1.47 ± 0.39
Palms	47.96 ± 0.02	6.87 ± 0.94	1.02 ± 0.09	1.04 ± 0.08	15.42 ± 0.47	1.15 ± 0.10

other nutrients within wood debris declined with increasing diameter. Palm trunks had dramatically higher concentrations of N, S, P, and K than sound coarse wood debris of dicotyledonous trees. For example, concentration of K in palm trunks was almost 19-fold greater than in coarse wood debris (15.42 and 0.82 $mg\ g^{-1}$, respectively). Concentration of K in palm trunks was similar to that of the pasture grasses (Table 3). Because of the high concentration of K, grasses composed a large proportion of the aboveground K pool. For example, grass only composed 3.5% of the TAGB at the João site (Table 1), but composed 51.5% of the aboveground K pool (Fig. 1). At the João site, the grass and litter components combined composed 27% of the TAGB but composed 87.6% of the total aboveground P pool

The concentrations of nutrients in ash were much different than that in uncombusted debris (Table 4). The concentration of C, which is readily volatilized, was 17–20% in ash. N concentration in ash ranged from 4.2 to 8.9 $mg\ g^{-1}$ as compared to a mean concentration in biomass of 5.1–5.7 $mg\ g^{-1}$ (Table 5). In contrast, the concentration of nutrients with a high temperature of volatilization was dramatically higher in ash as compared to unburned debris. For example, the mean concentrations of Ca in the aboveground biomass prior to burning were 0.9–3.0 $mg\ g^{-1}$; concentrations of Ca in ash ranged from 16.8 to 49.0 $mg\ g^{-1}$.

Differences in ash nutrient concentrations among sites tended to reflect the influences of site differences in tissue nutrient concentration, as well as the influences of variable levels of biomass consumption (Table 4). Ash concentrations of readily volatilizable nutrients (N and C) were lower, while nutrients with a high temperature of volatilization (P, K, and Ca) were higher in the Francisco pasture (i.e., the pasture with the highest combustion factor).

Total aboveground pools of C resembled the structure of TAGB ranging from 26 to 59 $Mg\ ha^{-1}$ (≈50% of the TAGB; Table 5, Fig. 1). However, nutrient pools cannot be easily inferred from a knowledge of the TAGB alone. For example, while TAGB was significantly greater at the João pasture compared to the Francisco pasture, the P and Ca pools were significantly greater at the Francisco pasture and there were no significant differences in N and S pools between these sites.

Fire effects on nutrient pools

As a result of fire, nutrients were either lost via volatilization or aerosol transport, transformed from organic pools into ash, or remained unaffected in residual uncombusted debris (Table 5, Fig. 1). Fire-mediated changes in nutrient pools appeared to be strongly influenced by three fire or ecosystem factors: (1) levels of biomass consumption; (2) nutrient distribution within the biomass classes with varying susceptibility to combustion losses; and (3) temperatures of volatilization of selected elements. The combination of a low temperature of volatilization as well as a disproportionate concentration in fine fuels which were consumed in very high percentages resulted in high N losses from all sites. Proportionally, more N was lost than the quantity of biomass consumed by fire. Site losses of N ranged from 205 to 261 $kg\ ha^{-1}$. In contrast, even though higher concentrations of K and Ca were found in grasses and litter (of which >96% were consumed by fire), low quantities were actually lost from the site. Losses of Ca and K were <15% and 22% of the prefire aboveground pool, respectively. Losses of P ranged from 1 to 37% of the prefire aboveground pool. Losses of S were intermediate to the relatively low losses of these cations and the high losses of N. S losses were 24–52% of the prefire aboveground pool.

Although prefire C pools were significantly larger at the Durval pasture, than the mass loss of C by burning was significantly larger at the Francisco pasture. Losses of C as a result of fire ranged from 11 $Mg\ ha^{-1}$ at the Durval site to 21 $Mg\ ha^{-1}$ at the Francisco site. This represents 19% and 81% of the prefire aboveground C pools and exemplifies the highly variable effect of biomass burning on C pools. Similar responses were measured for N, S, and P. For example, fires resulted in losses of 86 and 36%

Fig. 1 Nutrient pools of C, N, P, K, S, and Ca in three Amazonian cattle pastures before and after fires in Rondônia and Pará, Brazil. The *vertical lines* represent 1 SE of the total elemental pools

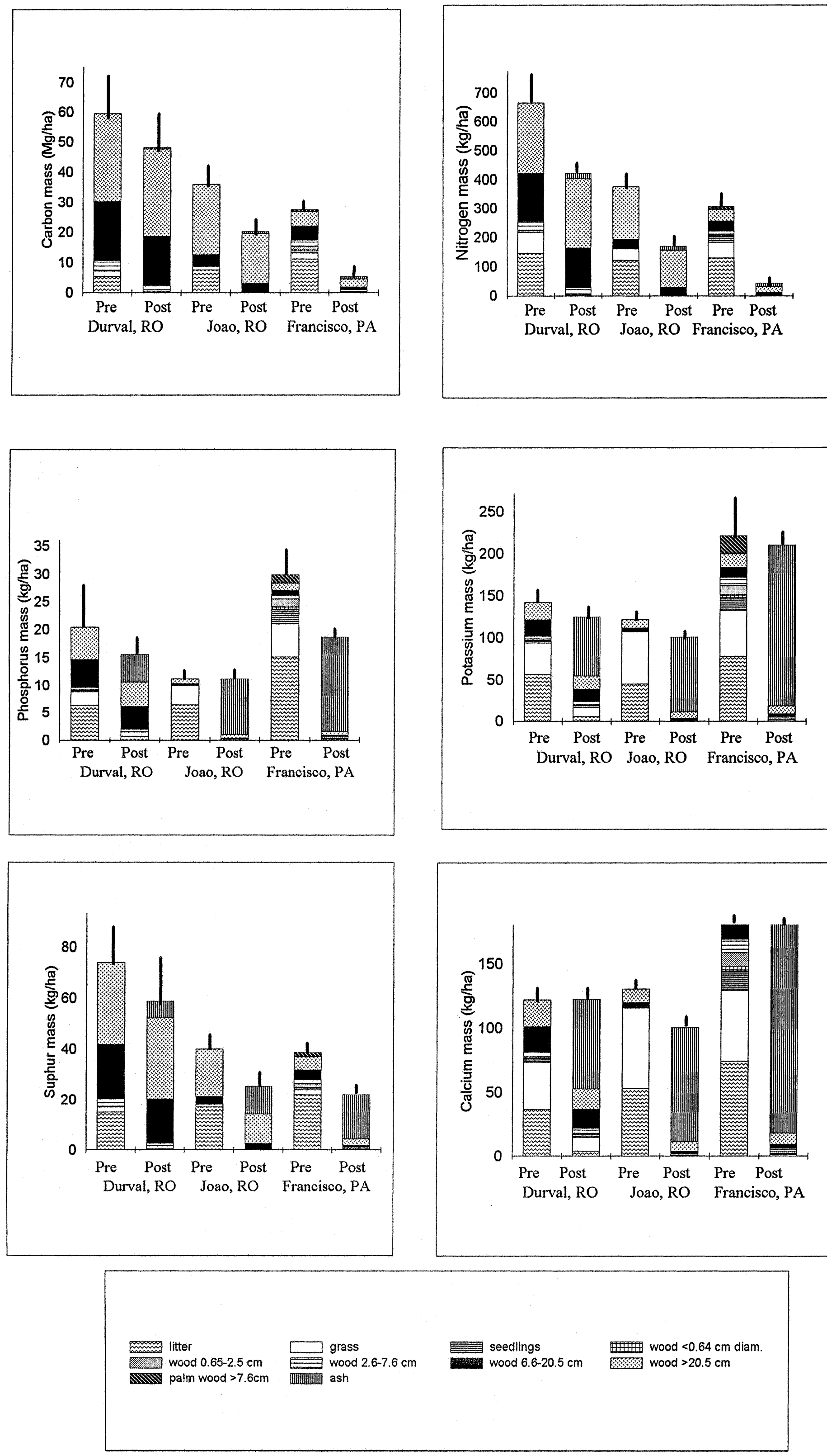

Table 4 Nutrient concentration of ash following fires in cattle pastures of Pará and Rondônia, Brazil. Numbers are mean ± SE

Component	Durval, Rondônia	João, Rondônia	Francisco, Pará
Nitrogen (mg g^{-1})	8.85 ± 1.08	4.7 ± 0.69	4.22 ± 0.63
Carbon (%)	25.35 ± 1.60	20.73 ± 1.59	17.48 ± 2.20
Sulphur (mg g^{-1})	3.17 ± 0.11	3.69 ± 0.41	6.30 ± 1.29
Phosphorus (mg g^{-1})	2.45 ± 0.14	3.79 ± 0.24	6.11 ± 0.86
Potassium (mg g^{-1})	34.01 ± 1.22	33.42 ± 6.82	68.40 ± 10.48
Calcium (mg g^{-1})	16.80 ± 2.22	20.65 ± 3.84	48.96 ± 6.03
Organic matter (%)	35.78 ± 2.30	28.55 ± 3.12	nd[a]

[a] *nd* no data

Table 5 Dynamics of aboveground nutrient pools before and after burning cattle pastures in Pará and Rondônia, Brazil. Numbers are mean ± SE. Different *superscript letters* indicate a significant difference between the pastures.

	Durval, Rondônia	João, Rondônia	Francisco, Pará
		NITROGEN (kg ha^{-1})	
Total pool-prefire	661 ± 145^{b}	374 ± 57^{a}	304 ± 41^{a}
% TAGB	0.55	0.51	0.57
Residual fuels-postfire	404 ± 15^{b}	155 ± 36^{c}	32 ± 7^{a}
Release from biomass	257 ± 50^{a}	220 ± 29^{a}	272 ± 42^{a}
Ash	18 ± 1^{c}	14 ± 1^{b}	11 ± 1^{a}
Residual + ash	420 ± 14^{c}	169 ± 35^{b}	43 ± 7^{a}
Site loss	240 ± 50^{a}	205 ± 29^{a}	261 ± 42^{a}
		CARBON (Mg ha^{-1})	
Total pool-prefire	59 ± 18^{b}	36 ± 7^{b}	26 ± 2^{a}
% TAGB	49.48	49.54	48.77
Residual fuels-postfire	48 ± 18^{c}	19 ± 5^{b}	4 ± 1^{a}
Release from biomass	12 ± 5^{b}	16 ± 3^{ab}	21 ± 2^{a}
Ash	0.5 ± 0^{b}	0.6 ± 0.0^{b}	0.8 ± 0.1^{a}
Residual + ash	48 ± 18^{c}	20 ± 5^{b}	5 ± 1^{a}
Site loss	11 ± 5^{b}	16 ± 3^{ab}	21 ± 2^{a}
		SULPHUR (kg ha^{-1})	
Total pool-prefire	78 ± 14^{b}	42 ± 6^{a}	48 ± 5^{a}
% TAGB	0.06	0.06	0.09
Residual fuels-postfire	53 ± 19^{b}	14 ± 3^{a}	5 ± 1^{a}
Release from biomass	25 ± 6^{b}	27 ± 3^{b}	43 ± 5^{a}
Ash	6 ± 0^{c}	11 ± 1^{b}	18 ± 2^{a}
Residual + ash	59 ± 20^{b}	25 ± 3^{a}	22 ± 2^{a}
Site loss	19 ± 6^{a}	16 ± 3^{a}	25 ± 5^{a}
		PHOSPHORUS (kg ha^{-1})	
Total pool-prefire	23 ± 5^{a}	11 ± 1^{b}	30 ± 5^{a}
% TAGB	0.02	0.02	0.06
Residual fuels-postfire	11 ± 2^{b}	1 ± 0.2^{a}	2 ± 0^{a}
Release from biomass	10 ± 2^{b}	10 ± 1^{b}	28 ± 5^{a}
Ash	5 ± 0^{c}	10 ± 0.4^{b}	17 ± 2^{a}
Residual + ash	16 ± 2^{b}	11 ± 0.5^{c}	19 ± 2^{a}
Site loss	5 ± 2^{a}	0.1 ± 1.1^{b}	11 ± 5^{a}
		POTASSIUM (kg ha^{-1})	
Total pool-prefire	152 ± 18^{a}	122 ± 15^{a}	221 ± 11^{a}
% TAGB	0.13	0.17	0.41
Residual fuels-postfire	55 ± 11^{b}	11 ± 2^{a}	18 ± 4^{a}
Release from biomass	97 ± 12^{b}	111 ± 13^{b}	203 ± 42^{a}
Ash	70 ± 4^{b}	89 ± 4^{b}	192 ± 23^{a}
Residual + ash	125 ± 12^{b}	101 ± 4^{b}	210 ± 23^{a}
Site loss	33 ± 14^{b}	21 ± 14^{b}	11 ± 5^{a}
		CALCIUM (kg ha^{-1})	
Total pool-prefire	108 ± 84^{b}	84 ± 9^{b}	162 ± 15^{a}
% TAGB	0.09	0.12	0.30
Residual fuels-postfire	49 ± 10^{b}	19 ± 4^{a}	25 ± 6^{a}
Release from biomass	47 ± 7^{b}	65 ± 6^{b}	137 ± 14^{a}
Ash	34 ± 2^{b}	55 ± 2^{b}	137 ± 16^{a}
Residual + ash	93 ± 23^{b}	74 ± 5^{b}	162 ± 17^{a}
Site loss	16 ± 8^{b}	10 ± 6^{b}	0 ± 0^{a}

of the aboveground N pools in the Francisco and Durval pastures, respectively (Table 5, Fig. 1).

While <4% of the prefire C and N pools remained in the ash component, we found larger proportions of the postfire pools of those nutrients with high temperatures of volatilization in the ash component following fire (Table 5, Fig. 1). The variable levels in biomass consumption between pastures resulted in differences in the partitioning of aboveground nutrient pools following fire (Fig. 1). Ash nutrient pools of S, P, K and Ca were significantly higher at the Francisco site compared to the other burned pastures. With a high (84%) combustion factor at the Francisco site, >82% of the postfire pools of S, P, K, and Ca were found in the ash component. In contrast, at the Durval site, with a relatively low combustion factor (21%), <37% of the residual postfire nutrient pool of S, P, and Ca was in ash.

At each site, ash always composed a large proportion of the postfire K pool because a large quantity of K was sequestered in the grass/litter component which burned most completely. The variable fate of nutrients among the three sites suggests that fire severity influences nutrient losses both during and after combustion. As ash is much more mobile than residual uncombusted wood debris, it is likely that greater site losses via erosion

Table 6 Soil nutrient concentrations in cattle pastures of Pará and Rondônia, Brazil. Numbers are mean ± SE

Depth (cm)	Durval, Rondônia	João, Rondônia	Francisco, Pará	
	Prefire	Prefire	Prefire	Postfire
		NITROGEN (mg g^{-1})		
0–2.5	3.70 ± 0.82	2.86 ± 0.49	2.57 ± 0.12	2.93 ± 0.17
2.5–10	2.58 ± 0.60	1.67 ± 0.10	1.75 ± 0.13	1.76 ± 0.02
		CARBON (%)		
0–2.5	5.78 ± 1.58	3.76 ± 0.60	3.55 ± 0.12	4.22 ± 0.30
2.5–10	3.50 ± 0.87	2.11 ± 0.17	2.45 ± 0.10	2.25 ± 0.11
		SULPHUR (mg g^{-1})		
0–2.5	0.31 ± 0.05	0.23 ± 0.01		
2.5–10	0.25 ± 0.02	0.23 ± 0.01		
		PHOSPHORUS (mg g^{-1})		
0–2.5	0.23 ± 0.05	0.19 ± 0.02		
2.5–10	0.13 ± 0.01	0.11 ± 0.00		
		POTASSIUM (mg g^{-1})		
0–2.5	0.18 ± 0.04	0.37 ± 0.03		
2.5–10	0.09 ± 0.02	0.08 ± 0.02		
		CALCIUM (mg g^{-1})		
0–2.5	0.42 ± 0.18	0.72 ± 0.05		
2.5–10	0.25 ± 0.09	0.21 ± 0.06		

Table 7 Soil nutrient mass in cattle pastures of Pará and Rondônia, Brazil. Numbers are mean ± SE

Depth (cm)	Durval, Rondônia	João, Rondônia	Francisco, Pará	
	Prefire	Prefire	Prefire	Postfire
		NITROGEN (kg ha^{-1})		
0–2.5	1182 ± 65	914 ± 50	860 ± 20	980 ± 23
2.5–10	2477 ± 137	1599 ± 88	1734 ± 74	1748 ± 75
		CARBON (Mg ha^{-1})		
0–2.5	18.5 ± 1.0	12.0 ± 0.7	11.9 ± 0.4	14.1 ± 1.0
2.5–10	33.6 ± 1.9	20.2 ± 1.1	24.3 ± 1.0	22.3 ± 1.1
		SULPHUR (kg ha^{-1})		
0–2.5	99.7 ± 5.5	75.0 ± 4.1		
2.5–10	244.0 ± 13.9	223.5 ± 12.3		
		PHOSPHORUS (kg ha^{-1})		
0–2.5	63.9 ± 3.5	59.9 ± 3.3		
2.5–10	124.6 ± 6.9	105.4 ± 5.8		
		POTASSIUM (kg ha^{-1})		
0–2.5	44.7 ± 2.5	117.4 ± 6.5		
2.5–10	86.3 ± 4.8	76.7 ± 4.2		
		CALCIUM (kg ha^{-1})		
0–2.5	105.4 ± 5.8	230.0 ± 12.7		
2.5–10	239.6 ± 13.2	201.3 ± 11.1		

processes would occur at sites with high combustion factors such as the Francisco site than at sites with low combustion factors such as the Durval site.

Soil nutrient pools

Nutrient concentrations of soils are indicative of the oligotrophic nature of the oxisols underlying these pastures (Table 6). Nutrient concentrations at the 0–2.5 cm layer were consistently higher than those of the 2.5–10 cm layer. Concentration of C, N, S, and P were highest in the recently established Durval pasture than the older Francisco and João pastures. In contrast, K and Ca were higher in the 0–2.5 cm layer of the older João pasture than the younger Durval pasture. Higher K and Ca concentrations in older pastures may be reflective of the cumulative influence of nutrient inputs from ash during a longer history of slash and pasture fires as well as litter inputs from pasture grasses, known to be higher in K than wood debris or litter from primary forest (Kauffman et al. 1995). Soil nutrient concentrations were much lower than in ash (Tables 4 and 7). At the Francisco site, where prefire and postfire soils were measured, we found slight increases of N and C concentration as a result of fire.

The bulk density of soils ranged from 1.28 to 1.34 g cm^{-3}. In the top 10-cm soil layer, N mass ranged from 2513 to 3659 kg ha^{-1} and C mass ranged from 32.2 to 52.1 Mg ha^{-1}, (Table 7). Carbon at the 10–30 cm depth at the Francisco site was 20.7 Mg ha^{-1} or only ≈30% of the 0–30 cm soil C pool. The 0–2.5 cm surface layer composed > 30% of the 0–10 cm soil nutrient pool of N, C, S, and P. At the older João site, the K and Ca pool in the 0–2.5 cm layer exceeded that of the 2.5–10 cm layer; this was unlike the conditions at the younger Durval pasture where the K and Ca pool in the underlying layer was higher.

The mass of N, S, P, and Ca soil pools were far greater than aboveground pools (Tables 5 and 7, Fig. 1). For example, aboveground N pools only composed 10–15% of the ecosystem pool made up of the total aboveground and the 0–10 cm surface layer of soils. Aboveground P pools composed 6–11% of the total P pool. In contrast, C and K pools were approximately equivalent between these aboveground and belowground components. Aboveground C composed 41–53% of this ecosystem pool. As a proportion of this ecosystem pool, fires resulted in the loss of 6–9% of N, 10–34% of C, 4.5–4.7% of S, 2.4–0.06% of P, 11.7–6.6% of K, and 3.5–1.9% of Ca (Fig. 1).

Discussion

Biomass and nutrient losses by pasture fire

The pastures in this study have similar preburn TAGB values (53–119 Mg ha^{-1}) to those reported for a cattle pastures in the eastern Amazon (52 Mg ha^{-1}) (Uhl and Kauffman 1990). However, this does not likely represent the total range of biomass of Amazonian pastures. The most recent fire at the Francisco pasture reduced residual wood to ≈8 Mg ha^{-1}. In the future, the mass of wood will likely be lower than that of the grasses and litter at this site, a likely scenario for many older pastures throughout the Amazon Basin. In contrast, residual wood debris was as high as 207 Mg ha^{-1} on recently deforested and burned primary forest sites to be established as pastures (Kauffman et al. 1995). With a range of residual wood debris of 8–207 Mg ha^{-1} and a range in non-wood biomass of 16–29 Mg ha^{-1}, we can conservatively project a pasture biomass range in the Amazon Basin of 24–236 Mg ha^{-1}. These values exceed the pasture/agricultural biomass estimates used in many studies to describe C dynamics of deforestation and land use in the tropics (e.g., Detwiler and Hall 1988; Houghton et al. 1991; Fearnside 1992, 1996), largely because these studies did not include the dominant biomass component in the ecosystem – residual wood debris.

In contrast to pastures formed from tropical forest, biomass of fuels in Brazilian tropical savannas (Cerrado) ranged from 7 to 10 Mg ha^{-1} (Castro 1995; Kauffman et al. 1994). The large differences in biomass between Amazonian cattle pastures and naturally occurring tropical savannas are due principally to the residual wood debris. It follows that the C and other nutrient losses due to fires in cattle pastures are dramatically higher than losses due to natural savanna fires. For example, C losses were 2.6–3.2 Mg ha^{-1} in Cerrado fires compared to 11–21 Mg ha^{-1} for pasture fires in this study (Table 5, Kauffman et al. 1994). N losses from fire in Amazonian cattle pastures were ≈10-fold greater than losses from fires in these natural savannas.

There are a number of land-use activities where fire is used in Amazonia and the loss of C and nutrients (i.e, the flux of emissions to the atmosphere) varies depending upon the land-use history of the sites. For example, biomass and nutrient losses from fires in Amazon cattle pastures are substantially lower than fires of primary slash. The mean biomass loss from primary slash fires was 172 Mg ha^{-1} compared to the mean biomass loss in pasture fires of 34.5 Mg ha^{-1} (Table 1, Kauffman et al. 1995). Average N loss by primary slash fires was 1218 kg ha^{-1} compared with 239 kg ha^{-1} from the pasture fires in this study.

Virtually all deforested areas are eventually converted to pasture in the Amazon. Large ranchers normally convert deforested areas directly to pasture, whereas small landowners often utilize areas in one or two shifting crop rotations prior to pasture conversion. Fires are the most common means of pasture maintenance (Uhl and Buschbacher 1985). Based upon our interviews with landowners we found that pastures are purposefully burned every 2 or 3 years during the first 10 years of the pasture's existence. Uhl et al. (1988) and Fearnside (1992) also reported that moderately to heavily utilized pastures with a lifetime of 6–12 years were re-burned one to five times. Because of the widespread use of fire for all aspects of pasture and forest slash disposal, accidental fires are also extremely common in the Amazon (Uhl and Buschbacher 1985). We have observed that approximately 40% of the fires set in slash and pastures escaped into adjacent areas of pasture, cropland, or slash.

Based upon our measurements and observations, the cumulative fluxes of nutrients and carbon to the atmosphere during the lifetime of a pasture must be quite significant. For example, we conclude that total N loss from fires during the first 6 years of pasture use (i.e., the slash fire and pasture fires at years 2, 4, and 6) would amount to 1935 kg ha^{-1}. This is equivalent to 94% of the mean aboveground N pool for primary forests reported by Kauffman et al. (1995).

C inputs into the atmosphere by pasture fires

The contribution of pasture fires as a source or sink of greenhouse gasses is largely unknown. Fearnside (1992) suggested that under a typical scenario of three fires following the initial slash fire, a total of 35% of the biomass C would be released from combustion processes and ≈62% would be released through decomposition processes. Furthermore, Fearnside (1992) used a mean TAGB for primary forest of 300 Mg ha^{-1} indicating that ≈53 Mg C ha^{-1} was released by fire in his estimates of greenhouse emissions from deforestation in the Brazilian Amazon. Kauffman et al. (1995) reported that the mean combustion factor of four primary Amazon slash fires was ≈50%. The mean TAGB of these forests was 345 Mg ha^{-1}. Based upon the mean biomass loss of the three pasture fires in this study we estimate that three reburns in a pasture would consume ≈41 Mg ha^{-1} of residual wood originating from the forest. This is equivalent to a total release of 12% of the forest TAGB. The total biomass consumed in the initial fire and the three pasture fires is estimated to equal 214 Mg ha^{-1} or 62% of the mean TAGB of primary forests. Our results indicate that more C and N is released via combustion processes and less is released via decomposition processes than predicted by Fearnside (1992). The significance lies in the additional quantities of more radiatively active aerosols, black carbon and trace gas emissions (e.g., CH_4, CO, N_2O, NO_x, nonmethane hydrocarbons) arising from biomass burning compared to decomposition (Shine et al 1990).

Pasture burns are likely the most common type of fire currently occurring in the Amazon. As such they are likely to be significant sources of C to the atmosphere. If the deforestation rate is ≈15,000–19,000 km^2 year^{-1} in the Amazon, and pastures burn (purposely or accidentally) on the average of every 2 years, then pastures fires would occur on 60,000–80,000 km^2 of land in a given year in the Amazon. With a mean C loss of ≈16 Mg ha^{-1} in pastures (Table 5), pastures fires in the Amazon would release 96–128 Tg C year^{-1} into the atmosphere. Mean C loss by primary slash fires was ≈86 Mg ha^{-1} (Kauffman et al. 1995). If deforestation rates are 15,000–19,000 km^2 year^{-1}, C losses associated with the initial primary forest slash fires would total 130 to 160 Tg year^{-1}. Gross C inputs of 226–288 Tg year^{-1} under this scenario are equivalent to 14–18% of the IPCC (International Panel on Climate Change) value (1.6 GT C year^{-1}) for emissions associated with deforestation and land use (IPCC 1996). This potential contribution clearly indicates that pasture fires are a significant atmospheric C source.

Response of nutrient pools to conversion from forest to pasture

Although there are significant losses in aboveground nutrient pools upon the conversion of forests to pastures, the effects of forest conversion on soil nutrient pools are not so clear. C concentration of surface soils in pastures ranged from 3.55 to 5.78% (Table 6) compared to 3.01–6.27% for paired primary forests of the same regions (Kauffman et al. 1995). With conversion of forest to pasture, we found no significant changes in either concentration or mass of soil C (Fig. 2). In contrast, Neill et al. (1996) reported increases in soil C upon conversion to pasture. Interestingly, their estimates of pasture soil C pools (0–10 cm) ranged from 16.8 to 27.4 Mg ha^{-1} compared with our estimates of 35–53 Mg ha^{-1} (Table 7).

Comparing our estimates of soil C and nutrient pools between forests and pastures is complicated by the fact that there was a 46% increase in soil bulk density in pastures compared to forests (a mean of 1.31 g cm^3 in pastures and 0.90 g cm^3 in adjacent primary forests). To compensate for differences in bulk density, we calculated the soil C and nutrient pools in pastures based on the equivalent soil mass (0–10 cm) to that of the adjacent primary forest (Neill et al. 1996). Soil C and nutrient pools of pastures and forests in Fig. 2 were calculated using a soil mass of 1069 Mg ha^{-1} for the Pará and 750 Mg ha^{-1} for the Rondônia sites.

When forests are converted to pastures, dramatic shifts in the partitioning of ecosystem nutrient pools occur. In contrast to primary forests, the aboveground component of pastures did not comprise as significant a proportion of the ecosystem nutrient pool (Fig. 2). This was particularly true of the older Francisco and João pastures where lower amounts of residual wood from the forest remained. In primary forests, aboveground pools of C, K, and Ca composed ≈76–89% of this ecosystem pool. In pastures, aboveground pools of C, K, and Ca were 53%, 46% and 20% of the total pool, respectively. Aboveground pools of N and S in forests composed ≈50% of the total but < 16% of the total in pastures. Aboveground P composed ≈65% of the ecosystem pool in primary forest but only ≈9% in pastures.

With the exception of Ca, we did not find any dramatic losses or gains in surface soil nutrient pools when comparing pastures to primary forest sites (Fig 2). This

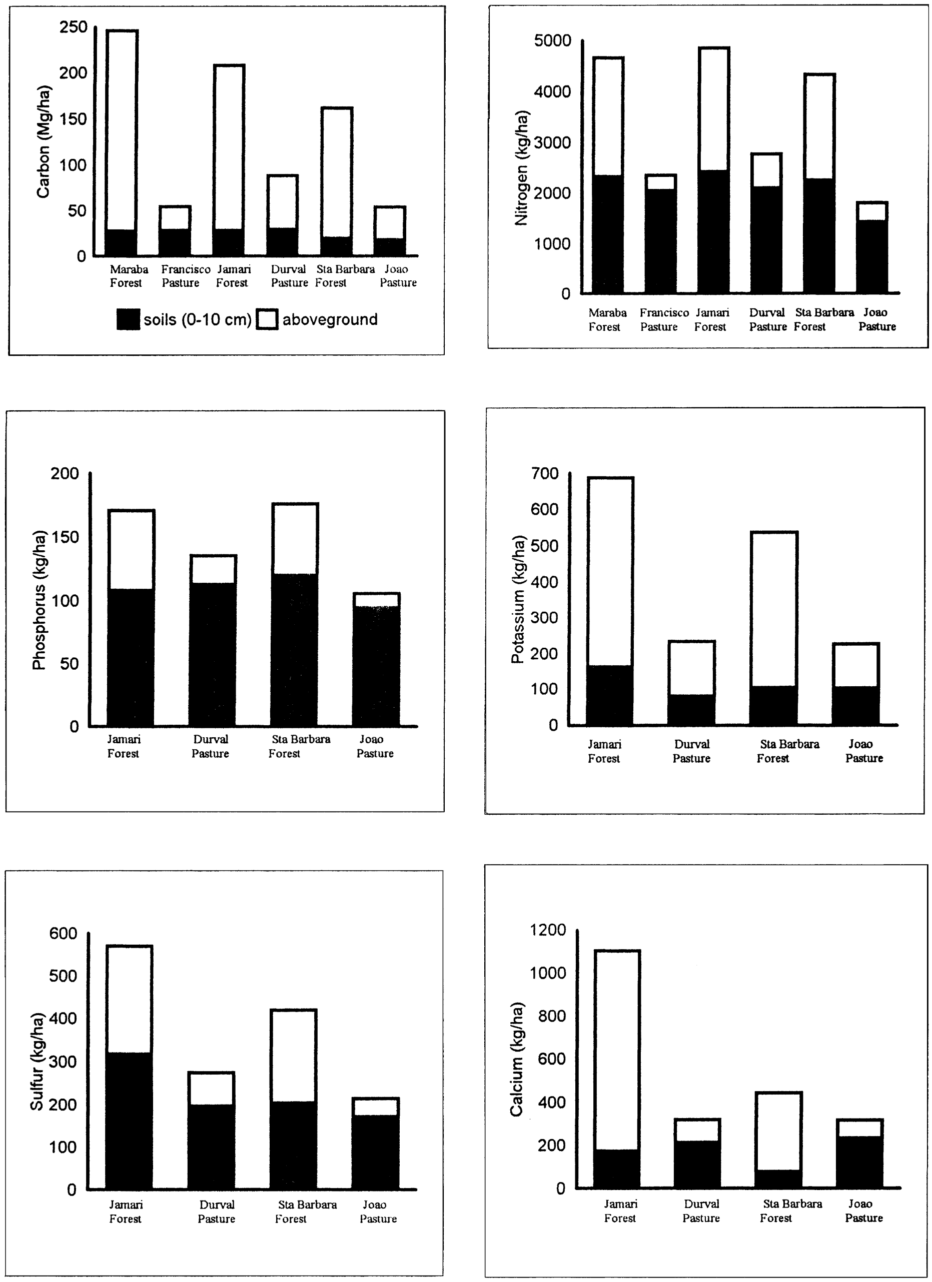

Fig. 2 Ecosystem pools of primary forests and cattle pastures. These nutrient pools comprise the total aboveground biomass and surface soils. Soil nutrient pools are representative of equivalent amounts of soils in pastures and primary forests. This is the mass found in the top 10 cm of soils in primary forests. As soil bulk density is higher in pastures, pools reported in this figure are lower than those of Table 7. The Francisco pasture was adjacent to the Maraba forest site; the Durval pasture was adjacent to the Jamari forest site; and the João pasture was approximately 5 km from the Santa Barbara forest sites. Detailed forest measurements can be found in Kauffman et al. (1995)

is similar to other Amazonian comparisons of recently established pastures (< 10 years) and standing forest (Falesi 1976; Desjardins et al. 1994; Sanchez and Salinas 1981). The concentration and relative mass of Ca was higher in soils of pastures than in primary forests. Apparently, greater proportions of the Ca pool deposited as ash or released through root decomposition is retained in surface soils than other nutrients. However, decreases in available K and Ca due to leaching in Amazon pastures became significant after 6 years of use (Correa 1989; Teixeira 1987).

Because of significant losses in aboveground nutrient pools coupled with minimal changes in soil pools, there were large shifts in the partitioning of nutrient and C pools of the converted ecosystems (Fig. 2). Comparing ecosystem pools of nutrients between the pastures and the paired forests we found that C pools of pastures were 22–43% of those in forests; N pools in pastures were 44–59% of those in forests; S pools were 49–52% of those in forests; K pools were 35–44% of those found in forests; P pools were 63–83% of those found in forests and Ca pools in pastures were 31–84% of those in forests (Fig. 2). While this indicates significant losses associated with pasture conversion, the magnitudes of these losses can not be ascertained because other potentially significant belowground ecosystem pools were not measured (i.e., soils > 10 cm depth and roots).

Data in Fig. 2 are the prefire pools of nutrients in pastures. Because almost all wood was consumed by fire in the Francisco pasture, the postfire aboveground C pool was only ≈3% of the C pool in the adjacent primary forest (i.e., the Maraba site in Kauffman et al. 1995). This small aboveground C pool may be most representative of older pastures as well as the ultimate fate of C pools in actively managed pastures in the Amazon.

Agricultural and cattle ranching activities are unsustainable as practiced and are unlikely to be converted into sustainable systems on sufficiently wide areas (Fearnside 1988, 1993a; Almeida and Uhl 1995). In ecological, societal, and economic contexts, there are a number of negative ramifications of deforestation and pasture conversion. Dramatic losses in biological diversity and site productivity occur with deforestation, fire, and pasture formation because of losses in the native biota, declines in nutrient pools, and increases in soil erosion and compaction. Pasture conversion does not require a large labor force and does little to alleviate employment shortages in rural areas of Amazonia (Fearnside 1993a; Almeida and Uhl 1995). Repeated pasture fires are significant sources of atmospheric C and other greenhouse gasses. Cumulatively over the life of a pasture, atmospheric inputs of radiatively active aerosols and gases arising from pasture fires are likely to be equivalent to those of the slash fires of primary forests. Under current land use patterns, fires burn more areas in this cover type in the Amazon than any other. A better quantification of the global contribution of greenhouse gases from pasture fires and the concomitant influences of nutrient depletion on the capacity for these ecosystems to sequester C in the future is imperative. Of greater importance are the development of sustainable alternatives to current land use practices.

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