AN ABSTRACT OF THE THESIS OF

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Abstract approved

J. Boone Kauffman

Biomass burning associated with deforestation and land use in the Amazon, greatly contributes to emissions as well as depletion of elemental pools. We identified deforestation and pasture formation and quantified resultant burning emissions and terrestrial aboveground elemental pool losses during a period of early colonization in the vicinity of Jamari, Rondônia. Using the Tasseled Cap transformation on multi-date Landsat Thematic Mapper (TM) data (1984-1992), we developed a land cover and change map for a 94,372-ha study area. Between 1984 and 1992, 8,250 ha of primary and 828 ha of regenerating forest were cleared (18% and 1% of the study area, respectively). Cleared land increased from 1,231 ha to 2,692 ha (1% to 3% of the study area) by 1986 and this area was in a cleared state in 1992. We developed equations using realistic pasture management burn scenarios to model cumulative area of pasture burned (19,008 ha). We computed emissions using a model that scaled up ground-based data (biomass, combustion factors, and flaming and smoldering combustion emission factors) by land cover type (primary forest, regenerating forest, and pasture). Over the extent of the entire study area, an average of 138, 920, and 16 kg C ha^{-1} yr^{-1} in CO, CO_2, and CH_4, respectively, were generated from primary forest slash burns. Regenerating forest slash fires contributed 4, 31, and <1 kg C ha^{-1} yr^{-1} in CO, CO_2, and CH_4, respectively. The cumulative area of pasture burned produced 46, 316, and 4 kg C ha^{-1} yr^{-1} in CO, CO_2, and CH_4, respectively. Site estimates of C, N, and S pools

and losses were applied to the land cover change map to estimate elemental pool losses from slash and pasture fires during the study period. In 1984, study area elemental pools were 163,297 kg C ha^{-1}, 1,971 kg N ha^{-1}, and 199 kg S ha^{-1}. Between 1984 and 1992, 5%, 7%, and 5% of the terrestrial aboveground C, N, and S pools, respectively, were lost from primary forest slash fires. Also, cumulative pasture burning contributed to losses of 1%, 2%, and 1% of the C, N, and S pools, respectively.

Detection of Deforestation and Land Conversion and Estimation of Atmospheric Emissions and Elemental Pool Losses from Biomass Burning in Rondônia, Brazil

by

Liane S. Guild

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APPROVED:

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Major Professor, representing Wildlife Science

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Chair of Department of Fisheries and Wildlife

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Dean of Graduate School

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Liane S. Guild, Author

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CONTRIBUTION OF AUTHORS

Dr. J. Boone Kauffman and Dr. Warren B. Cohen were involved in the design analysis, guidance, and writing of each manuscript.

TABLE OF CONTENTS

TABLE OF CONTENTS (Continued)

LIST OF FIGURES

LIST OF FIGURES (Continued)

Figure | Page

LIST OF FIGURES (Continued)

LIST OF TABLES

LIST OF TABLES (Continued)

Detection of Deforestation and Land Conversion and Estimation of Atmospheric Emissions and Elemental Pool Losses from Biomass Burning in Rondônia, Brazil

Chapter 1

Introduction

Statement of Purpose

A considerable global source of atmospheric emissions is from biomass burning of tropical forest slash and pasture. Furthermore, forest conversion and pasture management deplete terrestrial carbon and nutrient pools. Despite numerous estimates of deforestation in the Amazon using satellite imagery, there are conflicting and limited data on rates and areal extent of deforestation and land conversion. As a result there are tremendous uncertainties in estimates of resultant emissions and losses from elemental pools from regional biomass burning. This research identified deforestation and cattle pasture formation in a 94,372-ha study area where early colonization and agricultural expansion was occurring in the vicinity of Jamari, Rondônia between 1984 and 1992. In addition, we report the impacts of forest conversion and land use burning practices of this area on emissions and elemental pools.

The novelty of this research, is that it combines: 1) multi-date high resolution satellite imagery to provide reliable land cover mapping (e.g., areal extent of forest clearing and land conversion) through time; 2) ground-based estimates (e.g., aboveground biomass, carbon, and nutrient dynamics); and 3) tower data (e.g., combustion and emissions) to derive landscape-scale biomass burning emissions and carbon and nutrient pool losses. Previous deforestation estimates were based on single-date satellite image analysis and rarely differentiate other land cover types such as regenerating forest, cultivation, and pastures with much accuracy.

Unlike single-date analysis, multi-date analysis of satellite imagery could indicate the frequency of subsequent clearing and assist in identification of the likely land use scenario. For example in the Amazon, pasture sites burn on the order of every 2-3 years whereas shifting cultivation sites are cleared and burned every six years. We used TM data for June 1984, July 1986, and July 1992 to characterize these land use patterns. These dates corresponded to the dry season in Rondônia when much of the clearing for burning occurs. With this series of TM data, we compared spectral transformation and change detection techniques (Tasseled Cap, Principal Components of Tasseled Cap indices, and Tasseled Cap Image Differencing) to map land cover type and change during the study period. Our land cover change map indicated deforestation events, clearing of regrowth areas (for conversion to pasture or shifting cultivation), and areas managed as pasture (remained in a state of clearing).

Due to the six-year gap (1986-1992) in TM data selected for this research, the timing and area estimates of forest clearing and pastures burned could not be identified directly. Therefore, based on known land use scenarios of the region, we developed equations to model cumulative area of pasture burned following conversion of forest during the period of 1986 to 1992. Emissions from biomass burning were estimated by integrating interdisciplinary local ground-based research data on: 1) pre and postfire biomass, carbon, and nutrient dynamics, and 2) flaming and smoldering fire chemistry (i.e., emission factors and combustion efficiency) with the TM-derived land cover change map and the 1986-1992 modeled area estimates of forest conversion and pasture burning. Emissions of CO, CO_2, CH_4, and hydrocarbon trace gases were computed following a standard emissions model, based on the rate at which fuel carbon burns per unit area and the weighted sum of emissions factors for flaming and smoldering combustion. The emissions equation included delineation of land cover types (primary forest, regenerating forest, and pasture) and partitioning of biomass categories for land cover types. During the study period, emissions from forest slash fires and burning of pastures were estimated to quantify the importance of pastures as subsequent sources of site emissions following forest conversion.

Site estimates of carbon, nitrogen, and sulfur pools and resultant losses associated with burning of primary and regenerating forests and pastures were applied to the TM-derived land cover change map and the 1986-1992 modeled area estimates of forest conversion and pasture burning to estimate site pools and losses of carbon, nitrogen, and sulfur at a landscape scale for the study area. Original primary forest area elemental pools were estimated and compared with elemental losses by 1984 and by the end of the study period in 1992. This comparison was conducted to quantify the magnitude of original forest pools becoming sources of biomass burning emissions from forest conversion and pasture management. Additionally, the estimates of carbon pools and losses for the study area were utilized to validate carbon losses derived from modeling carbon emissions from biomass combustion for the study period.

Hypotheses

1. With the knowledge of land use dynamics, patterns of deforestation and land use/land cover change can be quantified using a time series of imagery to measure both the spatial and temporal extent of fire activity, carbon pools, and atmospheric emissions.

2a. Substantial sources of emissions include conversion of primary forest to pasture and shifting cultivation;

b. Additionally, abandoned agricultural sites are considerable, albeit ephemeral sinks of carbon and nutrients, but;

c. Emissions arising from regenerating forests and pasture fires are increasingly important sources of emissions.

Chapter 2

Detection of Deforestation and Land Conversion to Estimate Biomass Burning in Rondônia, Brazil

Liane S. Guild, Warren B. Cohen, J. Boone Kauffman

Abstract

Fires associated with tropical deforestation, land conversion, and land use greatly contribute to emissions as well as the depletion of carbon and nutrient pools. Between 1984 and 1992, deforestation and land conversion rates in Rondônia, Brazil were remarkably high. The objective of this research was to identify deforestation and cattle pasture formation where early colonization and agricultural expansion was occurring during this time period in the vicinity of Jamari, Rondonia along the BR-364 highway. A time series of Landsat Thematic Mapper (TM) data between 1984 and 1992 was examined in a 94,372-ha area of active deforestation to map land cover change. Spectral transformations were used for enhancement and data compression. The Tasseled Cap (TC) transformation enhanced the contrast between forest, cleared areas, and regrowth. TC images in the time series were stacked into a composite multi-date TC and used in a Principal Components (PC) transformation to identify change components. In addition, consecutive TC image pairs were differenced and stacked into a composite multi-date differenced image. The multi-date TC composite, TC composite with PC, and TC differenced image composite were each compared in a maximum likelihood classification for identification of land cover change between 1984 and 1992. The land cover change maps resulting from the three procedures were evaluated for accuracy. The multi-date TC composite classification had the best overall accuracy at 79.3% (Kappa Statistic of 78.1%). This map showed that at the beginning of the study (1984), only 5% (4,738 ha) of the 94,372-ha area had been cleared. By 1992,

15% (14,310 ha) of the area had been deforested and an additional 7% (7,086 ha) of the forest was lost due to flooding of the Jamari River, associated with completion of the Samuel Hydroelectric Dam. Analysis of the map indicates that cleared forest was converted to pasture or shifting cultivation. Frequent burning (i.e., every two to three years) is characteristic of pasture maintenance and clearing and burning on a six-year cycle is typical of shifting cultivation in the Amazon region. The frequency of clearing identified gave an indication as to whether the land cover type was pasture or shifting cultivation. Areas exhibiting regrowth through time with infrequent clearing in the land cover change map, were interpreted to be in shifting cultivation, but represented less than 1% of the study area. Pasture area increased from 1,231 ha (1% of the study area) in 1984 to 2,692 ha (3% of the study area) by 1986 and remained in pasture until 1992. Primary forest clearing between 1984 and 1992 was 9,944 ha (11% of the study area) with 192 ha of this becoming flooded due to the Samuel Dam in 1989. Since pasture formation is the primary reason for deforestation in the region, the 9,944-ha area of primary forest cleared during the study period was likely cleared for conversion to pasture. From this multi-date analysis of land cover and change, coupled with results of ground-based studies, we quantified emissions and elemental losses at the landscape scale.

Introduction

Before 1975, deforestation in the Brazilian Amazon was less than 1%, however, the deforestation rate increased exponentially between 1975 and 1987 [*Moran*, 1993]. Road building, colonization projects, and fiscal incentives supplied the impetus for rapid deforestation. Anthropogenic burning of primary forests and pastures in the Amazon contributes major sources of emissions and depletes terrestrial pools of carbon and nutrients [*Crutzen and Andreae*, 1990; *Fearnside*, 1997; *Guild et al.*, 1998; *Houghton*, 1990; *Huges et al.*, in press; *Kauffman et al.*, 1995; 1998].

According to *Fearnside* [1997], approximately 11% (421,600 km^2) of the forested area of Amazonia had been cleared as of 1991. Deforestation rates increased in the 1980s due to colonization projects that created fiscal incentives for agricultural expansion in remote areas of the Amazon [*Browder and Godfrey*, 1997; *Hecht*, 1993; *Molion*, 1991; *Moran*, 1993]. Rondônia, a western state in Brazil, has had a recent history of high rates of deforestation. Rondônia comprises an area of 243,000 km^2 of the Brazilian Legal Amazon's 5,000,000 km^2. By 1997, nearly 24% (50,529 km^2) of the original 215,000 km^2 Rondônian forest area had been deforested [*INPE*, 1998]. Annual deforestation rates (excluding forest loss from hydroelectric dams) in Rondônia were 2,100 km^2 (1978-1988), 1,400 km^2 (1988-1989), 1,700 km^2 (1989-1990), and 1,100 km^2 (1990-1991); at annual rates of 1% or less [*Fearnside*, 1997]. Recent estimates by *INPE* [1998] suggest that average annual deforestation estimates were 2,767 km^2 (1994-1997). Deforestation rates in Rondônia have remained high relative to other Amazonian states [*Fearnside*, 1997].

In 1960, with the tin rush and the opening of the unpaved BR-364 highway providing an overland route between Rondônia and the Atlantic Coast, prospectors and settlers migrated to Rondônia [*Browder and Godfrey*, 1997]. Paving of the BR-364 highway was completed in 1984 with the intent to increase immigration and stimulate markets for agriculture and forest products [*Browder and Godfrey*, 1997; *Stone* et al., 1991; *Tucker et al.*, 1984].

In spite of numerous estimates of deforestation using satellite imagery and other sources, several uncertainties exist including the rate and extent of deforestation and land uses mapped with satellite imagery. Improved estimates on the fate of deforested lands through time using satellite imagery could improve estimates of emissions flux and terrestrial carbon and nutrient pool losses. In most studies, a single date of satellite imagery is used to map deforestation and land clearing [*Tardin*, 1980; *Woodwell et al.*, 1986; *Skole and Tucker*, 1993; *Tucker et al.* 1984; *Fearnside*, 1997]. In general, forest areas in a state of clearing, vegetation regrowth, and water can be distinguished visually in Landsat data. It is difficult, however, to differentiate land cover types such as regenerating forest, cultivation, and young pastures in a single date of imagery [*Tucker et al.*, 1984; *Woodwell et al.*, 1986; 1987]. Hence, there are conflicting and limited data on rates and the areal extent of deforestation and land conversion. In 1978, the extent of deforestation in Rondônia was estimated in the range of 4,200 km^2 [*Tardin*, 1980] and 6,300 km^2 [*Fearnside*, 1997]. By 1982, the estimates of the exent of deforestation ranged from 9,200 km^2 [*Tucker et al.*, 1984] to 11,400 km^2 [*Woodwell et al.*, 1987]. Deforestation extent as of 1988 has been reported as 24,000 km^2 [*Skole and Tucker*, 1993], 29,600 km^2 [*Fearnside*, 1997], and 37,200-37,900 km^2 [*Stone et al.*, 1991].

In other regions, change detection techniques in a time series of imagery have been used to monitor changes in land use, shifting cultivation, vegetation phenology, pasture development and to assess deforestation, crop stress, and damage [*Singh*, 1989; *Collins and Woodcock*, 1996; *Coppin and Bauer*, 1996; *Cohen et al.*, 1998]. Digital change detection allows quantification of temporal phenomena in multi-date satellite imagery [*Coppin and Bauer*, 1996]. Change detection techniques using multi-date Tasseled Cap, Principal Components Analysis, and image differencing integrate spectral transformations to enhance change in land cover features [*Richards*, 1984; *Fung*, 1990; *Cohen et al.*, 1998].

The Tasseled Cap (TC) linear transformation, or data plane rotation, is used to compress the spectral redundancy of the Landsat Thematic Mapper (TM) visible and infrared bands to create the vegetation indices of brightness, greenness, and wetness. The weights of the TC transformation are fixed and are not scene dependent [*Schowengerdt*, 1997]. Brightness is a weighted sum of the reflective visible (TM bands 1, 2, 3) and infrared bands (TM bands 4, 5, 7) and as such is sensitive to changes in total reflectance [*Crist and Cicone*, 1984]. Greenness is a contrast between the visible and infrared bands and is sensitive to chlorophyll absorption in the visible bands and high infrared reflectance associated with leaf cellular structure. Wetness is a contrast of the visible and near-infrared bands (TM bands 1, 2, 3, 4) against the shortwave infrared bands (TM bands 5, 7). The wetness band is sensitive to both soil and vegetation moisture content. The TC transformation lends itself to physically-based interpretation where the output TC image is interpretable vegetation indices.

Variation in the TC indices is influenced by the combination of the variation of vegetation, soil, and physical factors. Brightness is a useful index of overall vegetation brightness compared to soil. In general, vigorous vegetation is brighter than less vigorous vegetation and lightly colored soils are brighter than dark soils. Vegetation can be brighter or less bright than soils. For example, dry soils and light colored soil types can be brighter than vigorous vegetation and the soils can have a greater influence on the brightness index than bright vegetation. Vegetation shadowing or wet soils can decrease brightness. Greenness generally gives and indication of the amount of vegetation (percent canopy closure and leaf area index) and vigor [*Crist and Cicone*, 1984]. In general, high leaf area has a higher greenness. Senescing vegetation has a lower greenness than vigorous vegetation. Wetness gives an indication of moisture status (soil and vegetation) and forest structure [*Cohen et al.*, 1995; *Crist and Cicone*, 1984].

Cohen et al. [1995] reported that as canopies develop in old-growth forests of the Pacific Northwest, USA, although leaf area index becomes higher, shadowing increases. Therefore, we infer that brightness and greenness will generally be lower for mature or primary forests than for vegetation regrowth, due to shadowing from variation in canopy heights in later stages of succession. Old-growth forests generally are lower in brightness and higher in greenness and wetness than in clear-cut areas [*Cohen et al.*, 1998]. In contrast, a deciduous forest stand would have higher brightness and greenness, but lower wetness than evergreen forest [*Cohen et al.*, 1995]. Inferred from *Cohen et al.* [1995, 1998], primary forest and areas of well established vegetation regrowth would likely have higher values of wetness than areas maintained for pasture (dry soils). In cleared areas undergoing vegetation regrowth, greenness will generally increase to the point of canopy closure. Depending on soil color and moisture content, brightness will either increase or decrease during regrowth. If the vegetation regrowing is brighter than the soils and the soils are not relatively dark, brightness would likely increase during regrowth. Darker soil types and moist soils will decrease brightness. Brightness, however, may be decreased initially during vegetation regrowth due to shadowing, but increases as vegetation cover density increases, eliminating shadowing on soil. *Fung* [1990] reported that changes in greenness were the most accurate for detecting changes in crop types and vegetation to non-vegetation changes. For example, high accuracy was reported for land cover changes from corn to bare soil and from bare soil to pasture, but changes from bare soil to grain were not distinguished due to low greenness in mature grain crops. *Collins and Woodcock* [1996] found that in general, changes in brightness, greenness, and wetness were significantly associated with conifer mortality and that changes in wetness had the strongest and consistent association with mortality.

The Principal Components (PC) transformation is also a data compression technique, but unlike TC, it is not physically-based. The output principal component images have new coordinate axes that are orthogonal to each other and explain decreasing levels of variance with each successive component (similar to TC) [*Richards*, 1984; *Singh*, 1989; *Collins and Woodcock*, 1996; *Coppin and Bauer*, 1996; *Schowengerdt*, 1997]. Unlike TC, the weights in PC transformation matrix are not fixed and are scene dependent [*Schowengerdt*, 1997]. The difficulty in interpreting PC is that methodical analysis of the eigen-structure of the data along with the input image and output components is essential for making valid conclusions [*Coppin and Bauer*, 1996]. Analysis of the eigenvectors (linear combinations of the original bands for the PC axes rotation) is complex. By combining multi-date TM data in a PC analysis, a spectral-temporal transformation results creating some components indicative of change over time. *Collins and Woodcock* [1996] selected change components for classification and further analysis, leaving out stable components (commonly the first component), which usually accounts for spatial scene variation and not variation between dates represented in higher components. However, stable components can provide a frame of reference and improve classification results [*Cohen and Fiorella*, 1998]. *Richards* [1984] combined two MSS scenes in a PC analysis to examine change from fire damage to vegetation regrowth. Richards found that the higher order components highlight changes from vegetation in the first date and burn scar in the second date or burn scar on the first date and vegetation regrowth on the second date. In addition, *Richards* [1984] found that PC analysis was important for discriminating fire scar from mixed water/vegetation pixels, which are generally confused in classification. *Collins and Woodcock* [1996] found that a higher order component was associated with identification of tree mortality, generally marked by an increase in the TM mid-infrared bands. Inspection of the eigenvector matrix showed that this component was heavily weighted in the mid-infrared TM bands. *Singh* [1985] used multi-date MSS data in several change detection techniques including a PC analysis to detect shifting cultivation effects on tropical forest cover. The PC analysis gave a lower accuracy for change classification than a simple image differencing approach.

Image differencing is a common change detection approach that has been used to map changes in forest cover and agricultural areas [Cohen et al., 1998; *Coppin and Bauer*, 1996; *Fung*, 1990; *Singh*, 1989; *Woodwell et al.*, 1986]. Image differencing is a simple approach whereby coincident bands of spatially registered date pairs are subtracted. The output image of positive and negative values represent change and the values close to zero represent no change. There is difficulty in interpreting the differenced imaged because different input values can have the same result after subtraction and the original pixel value information is not retained [*Cohen and Fiorella*, 1998; *Singh*, 1989]. Further confusion could be associated with very different pixel values in two dates that yield a non-zero value indicating change, but are actually the same land cover type. Such variation in pixel values may be associated with atmospheric conditions or sun angle differences between dates. Also, it is difficult to determine where to select threshold boundaries of change and no change [*Singh*, 1989]. In Rondônia, *Woodwell et al.* [1986, 1987] used change detection techniques of visible red and near infrared band differencing between image date pairs using the Landsat Multispectral Scanner (MSS) data. Woodwell found that since healthy vegetation absorbs red light in the visible red band and reflects in the near infrared band, deforested areas would show an increase in reflectance in the visible red band and a decrease in reflectance in the near infrared band. Therefore for a deforested area, the difference between the visible red bands between dates would be on a positive scale and the difference between the near infrared bands would be on a negative scale, provided the earlier date was subtracted from the later date. *Fung* [1990] used image differencing for six bands of TM (excluding the thermal-infrared band) between two dates. The differenced image for the near-infrared band gave high accuracy (100%) in detection change from corn to bare soil, pasture to grain, and bare soil to pasture, but had less accuracy in detecting change from pasture to corn. Also, the near-infrared differenced image could not detect change from bare soil to grain since both cover types have low near-infrared reflectances, but could also have been related to timing of date acquisition and phenology. *Cohen et al.* [1998] found that both merged image differencing and simultaneous image differencing yielded high accuracy (>90%) for clear-cut harvest activity. Also, simultaneous image differencing is more feasible since it uses only a single classification of all differenced images whereas the merged image differencing entails a classification for each differenced image with subsequent merging.

The objective of this research was to use multi-date TM data for the period of 1984 to 1992 to compare change detection techniques for creating a land cover and change map. The purpose of this map is reliable identification of forest clearing and pasture in an area of Amazon forest in Rondônia, Brazil. This area was undergoing active deforestation and land cover change associated with early colonization. To develop a method to map land cover and change, we compared change detection techniques of TC, PC analysis of TC images, and image differencing of TC images to detect change in land cover types (primary forest, regenerating forest, and cleared/pasture). The resultant land cover change map identified deforestation (an indication of forest slash burning events), regrowth (an indication of regenerating forest/shifting cultivation and carbon sequestration), and areas in a state of clearing (an indication of areas maintained as pasture and continued carbon sources to the atmospheric sinks).

Methods

Study Area

We examined land use/land cover change from 1984-1992 over a 94,372-ha area of primary forest in Rondônia, Brazil. The study area is located along the BR-364 highway in an area of recent colonization resulting in active deforestation during the period of our study. The BR-364 highway, which bisects the state, was opened in 1960 and was paved in 1984. This highway provided a route for colonists to migrate and claim land within the state and provided an overland transportation route to the Atlantic Coast [*Browder and Godfrey*, 1997]. Areas along the BR-364 highway have been subject to intense deforestation for cultivation, cattle pastures, timber exploitation, and mining.

We chose to conduct our study in this region because of its relatively recent colonization and deforestation activity at the onset of our analysis (since the early 1980's) and this allowed quantification of deforestation and conversion to pasture or agriculture that occurs when tropical forest landscapes are colonized in this region. In addition, the presence of relevant long-term ground-based data and knowledge of land use in the area was available [*Guild et al.*, 1998; *Hughes et al.*, in press; *Kauffman et al.*, 1995, 1998]. Other related research in the region has occurred south of Jamari from Ariquemes to Vilhena along BR-364 highway where more long-term deforestation, agriculture, and mining have occurred [*Fujisaka et al.*, 1998; *Moraes et al.*, 1998; *Neill et al.*, 1997; *Tucker et al.*, 1984]. Our study area and study period results come from a landscape in the early stages of disturbance and may provide some interesting deforestation and land use comparisons with landscape studies of areas following many years of intense disturbance.

The 94,372-ha study area is centered at approximately 9°11' S and 63°10' W and about 100 km southeast of the state's capital, Pôrto Velho (Figure 2.1). The small town of Jamari is located along the Jamari River, a blackwater tributary of the Madeira River. The study area consists of both small landholdings of subsistence farms and large ranches. It is adjacent to the Jamari National Forest, which contains the Santa Bárbara Tin Mine.

The primary forest type of Rondônia and the study region is submontane open forest consisting of overstory broad-leaved canopy and subcanopy with an abundance of palms and vines [*Cummings*, 1998; *Departmento Nacional de Produção Mineral*, 1978; *Instituto Brasileiro de Desenvolvimento Florestal (IBDF) and Instituto Brasileiro de Geographia e Estatística (IBGE)*, 1993]. Soil types include red-yellow podzolic latosols and red-yellow latosols [*Neill et al.*, 1997].

Climatological data come from Pôrto Velho, Rondônia about 100 km north of the region. Mean annual precipitation is 2,354 mm [*Departmento Nacional de Meteorologia, Brasil*, 1992]. During the dry season, between June and September, mean precipitation is typically <100 mm per month. Dry season mean temperature is ~25°C ranging from a minimum of ~21°C to a maximum of 31°C with a mean relative humidity of 85%.

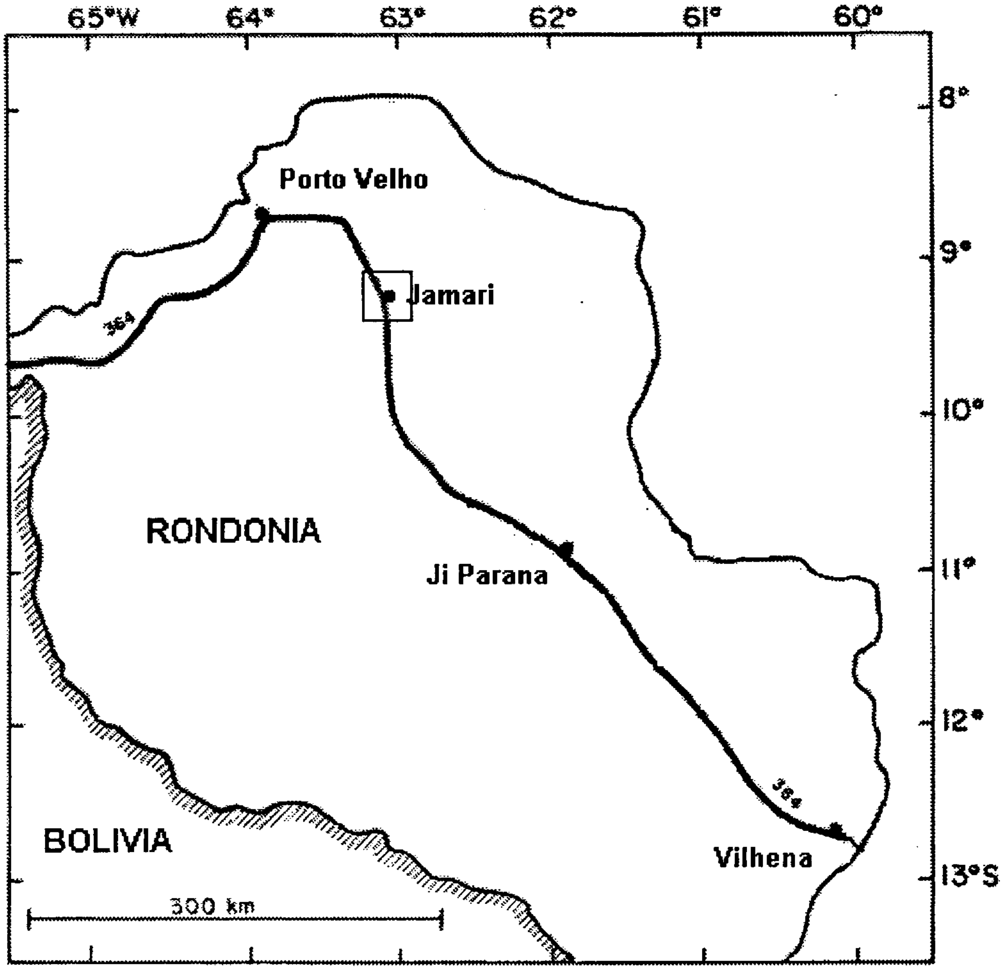


Figure 2.1. The Brazilian state of Rondônia comprises an area of 243,000 km^2. The location of the study area is 100 km southeast of Pôrto Velho in the vicinity of Jamari along the BR-364 highway. The study area is 94,372 ha and is centered at 9°11' S and 63°10' W with the approximate extent oulined on the map.

Data

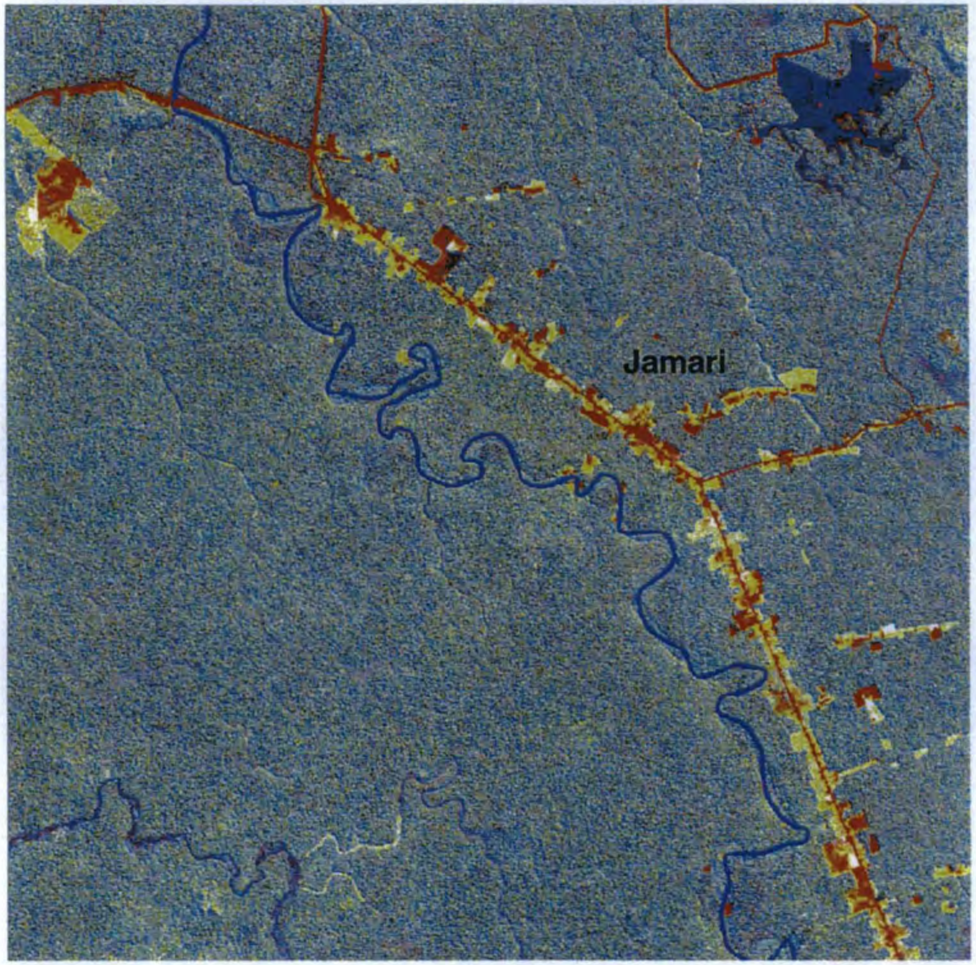
Available Landsat TM data for path 232, row 66 was selected from the National Aeronautics and Space Administration (NASA) Landsat Data Collection at the Earth Resource Observation System (EROS) Data Center, Distributed Active Archive Center (DAAC). This scene extends from Pôrto Velho in the north and south to just north of Ariquemes, Rondônia following the Jamari River and the BR-364 highway. The study area corresponds to a 1024 x 1024 pixel subsection of approximately 94,372 ha with a center point at approximately 9°11'S and 63°10'W. From the archive, dry season dates were selected for minimizing spectral variability associated with phenological differences. Therefore, as close to anniversary dates in a multi-temporal sequence were chosen. The following cloud-free images were selected: 24 June 1984, 16 July 1986, and 24 July 1992. Scenes were co-registered using an automated tie point and area correlation technique [*Kennedy and Cohen*, submitted]. The procedure locates tie points by maximizing an index of normalized cross-correlation for small subsets of the two images to be matched. Required user input is limited: pixel size, relative rotation of the two images, an initialization point in the two images, and the desired density of the output grid of tie points. Radiometric correction was considered but not performed because preliminary analysis indicated that the spectral change associated with deforestation and land clearing is far greater than changes associated with sun angle and atmospheric variation [*Cohen et al.*, 1998]. Moreover, because we analyzed the digital numbers in a set of statistical analyses for land cover classification, calibration was not important.

Change Detection

We tested three methods of change detection to map deforestation and land cover change between 1984 and 1992. We chose to use the Tasseled Cap (TC) transformation in variations of all change detection techniques because the TC images show sharp contrasts between forest, regrowth, and clearing (Figures 2.2a, 2.2b, and 2.2c). Making these features more obvious spectrally will strengthen the capability of the classification algorithm to delineate these land cover change classes of interest.

Tasseled Cap

Brightness, greenness, and wetness indices were generated for the 1984, 1986, and 1992 TM images using Landsat 4 and Landsat 5 TM Tasseled Cap coefficients (Table 2.1). The three TC images were stacked to create a nine-band multi-date composite. To improve classification performance on the multi-date composite, a classification of the 1992 TC image using unsupervised techniques in a maximum likelihood classification was used to delineate primary forest and non-forest. This classification was used to create a mask of primary forest and non-forest. Since the primary forest in 1992 was an area not undergoing change, we used the forest mask to eliminate this area from further analysis in the multi-date composite. Using the masked multi-date composite, unsupervised techniques were used to define spectral cluster signatures to train a second-level maximum likelihood classification. The classification produced a 60-class image. Although ground-truth data was not available, familiarity with the site from field visits assisted visual interpretation of the original TC data for comparison with the classification. Groupings of similar classes were determined and an 18-class land cover change map for the 1984-1992 time period was generated. Change class



Jamari, RO
June 1984
(9°11' S, 63°10'W)

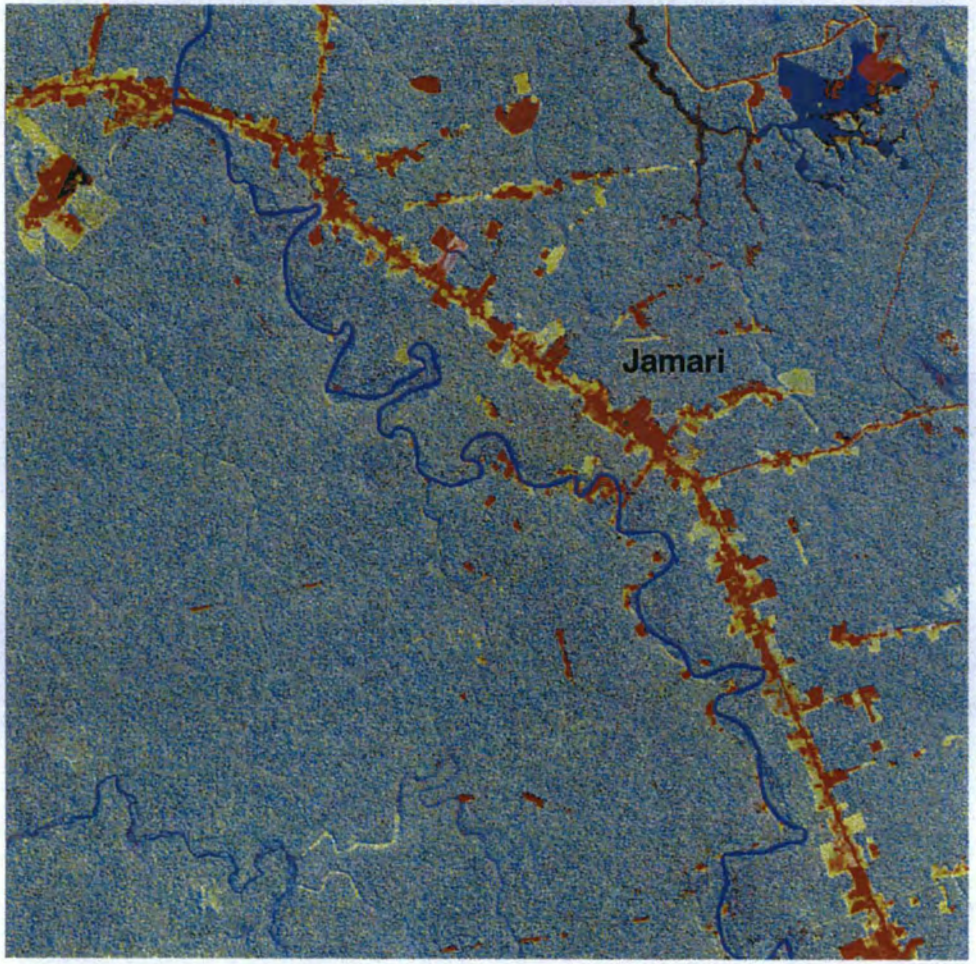
Landsat TM
Tasseled Cap

Approximate Scale

0 10 km

Study Area = 94,372 ha

Figure 2.2a. Tasseled Cap image from Landsat TM 24 June 1984 data for the Jamari, Rondônia study area. The BR-364 highway bisects the study area. Forest conversion for pasture and cultivation is present along the highway. Red indicates cleared areas with little or no vegetation, yellow represents areas of vegetation regrowth, green/blue designates primary forest, and blue areas are water.



Jamari, RO
July 1986
(9°11' S, 63°10'W)

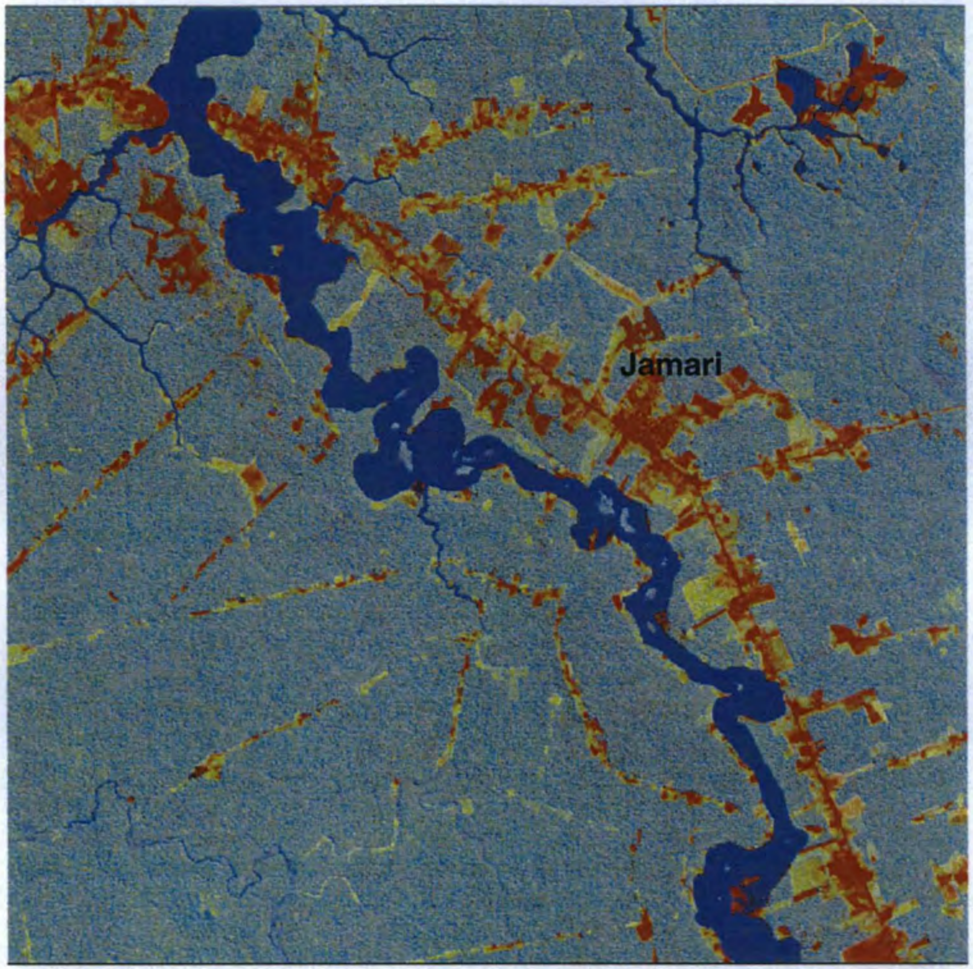
Landsat TM
Tasseled Cap

Approximate Scale

0 10 km

Study Area = 94,372 ha

Figure 2.2b. Tasseled Cap image from Landsat TM 16 July 1986 data for the Jamari, Rondônia study area. Forest conversion for pasture and cultivation has expanded along the BR-364 highway and radiates out from the highway.



Jamari, RO
July 1992
(9°11' S, 63°10'W)

Landsat TM
Tasseled Cap

Approximate Scale

0 10 km

Study Area = 94,372 ha

Figure 2.2c. Tasseled Cap image from Landsat TM 24 July 1992 data for the Jamari, Rondônia study area. The Jamari River extent increased due to completion of the Samuel Hydroelectric Dam in 1989. Some forest, pasture, and cultivation areas were lost due to flooding of the Jamari River.

Table 2.1. Landsat 4 and 5 TM Tasseled Cap Coefficients adapted from *Crist and Cicone* [1984] and *Crist et al.* [1986].

Index	Coefficients					
	TM1	TM2	TM3	TM4	TM5	TM7
Landsat 4						
Brightness	.3037	.2793	.4743	.5585	.5082	.1863
Greenness	-.2848	-.2435	-.5436	.7243	.0840	-.1800
Wetness	.1509	.1973	.3279	.3406	-.7112	-.4572
Landsat 5						
Brightness	.2909	.2493	.4806	.5568	.4438	.1706
Greenness	-.2728	-2174	-.5508	.7221	.0733	-.1648
Wetness	.1446	.1761	.3322	.3396	-.6210	-.4186

labels include combinations of forest, cleared, regrowth, dieback, flooded, dry, and water.

Tasseled Cap with Principal Components Analysis

In a second technique to detect land cover change through time, we combined all brightness, greenness, and wetness bands for each of the three dates in one unstandardized Principal Components (PC) transformation. We found that several of the components were dominated by the change in expanse of the Jamari River which made other land cover change features more subtle and not easily delineated. Due to the construction of the Samuel Dam in 1989, there was a substantial increase in the expanse of the Jamari River from the 1984 and 1986 images to the 1992 image. Therefore to eliminate the river expanse change from the PC analysis, a mask of the extent of the Jamari River in 1992 was used on all three dates prior to performing the PC transformation. Nine components, one for each input band, were generated in the PC transformation based on the covariance matrix. Each component was evaluated along with the PC eigenvectors, which are linear combinations for the PC axes rotations. Factor loadings were generated using the correlation matrix. The PC factor loadings, describing the correlation between the original bands and the PC bands, were graphed for each PC against the corresponding TC indices. The graphs indicated contrasts occurring between dates as well as within and among indices. This analysis assisted inspection of single PC band images for change, but did not indicate the type of change. Band by band, the PC images were analyzed with the original TC images, the eigenvectors, and the factor loadings to identify component bands exhibiting change through time due to deforestation, clearing, and regrowth. Three resultant principal components indicating change during the 1984 – 1992 study period together with the first component were used in a maximum likelihood classification using unsupervised techniques. The first component was a stable component that was used as a reference for change to improve the classification [*Cohen and Fiorella*, 1998; *Collins and Woodcock*, 1996]. Without the inclusion of the first component or stable component in the classification, there appeared to be no frame of reference and the spatial integrity of the image was lost. The original TC images for each of the three dates were used to interpret 60 classes in the classification. Subsequent grouping of similar classes generated a 13-class land cover change map using the same land cover labels used in the TC classification previously described.

Tasseled Cap Image Differencing

TC image date pairs were subtracted one from the other to create TC differenced images. Subtracted image date pairs included, 1) 1984 and 1986, and 2) 1986 and 1992. The two TC differenced images along with the 1992 TC image, which served as a reference image for change, were combined into one image creating a nine-banded composite image. A maximum likelihood classification using unsupervised techniques was performed on this nine-band image creating 60 classes. Following inspection of the classes with the original TC images, 18 unique change classes were identified with the land cover labels previously discussed in the TC classification.

Accuracy Assessment

An equalized stratified random sampling approach was used to assess the accuracy of each of the three land cover change classifications. Fifteen random points were selected for each class and visually compared with the original TC images. In addition, validation of the land cover change classes came from knowledge of the history of the land use in the area based upon experiences of ground-based research in this region. Based on the number of classes, between 195 and 270 points per classification were verified for classification accuracy. An error matrix was used to calculate producer's accuracy (indication of omission errors), user's accuracy (indication of commission errors), and the overall accuracy of the classifications. In addition, a more conservative approach, the Kappa Statistic, was calculated. The Kappa Statistic is a maximum likelihood estimate from a multinomial distribution that measures the actual agreement of the classification output with what is observed in the information or data used for "truthing" minus the chance agreement. Essentially the Kappa Statistic is the difference between the observed accuracy and the chance agreement divided by one minus the chance agreement [*Lillesand and Kiefer*, 1994]. Finally, the Kappa Statistic was calculated using only the 12 classes that corresponded to all three change detection classifications.

Results

Clearing of primary forest was evident and easily detected in the land cover change classification approaches. All three classification techniques yielded acceptable accuracy given the number of unique classes generated for each land cover change map and for a multi-date analysis (Tables 2.2, 2.3, and 2.4). The PC classification had a fewer number of classes (13 classes) than the TC (18 classes) and TC difference (18 classes) classifications because areas associated with flooding and the Jamari River were masked out prior to classification. Masking out these areas eliminated classification of these features. The PC classification missed classifying an area that was in regrowth between 1984 and 1986 and was cleared by 1992 (Class 16). The TC difference classification missed capturing two classes identified in the TC classification. Both of these classes involved regrowth or cleared areas that were flooded by 1992 (Classes 2 and 6). The TC classification, however, missed identifying a class delineated in both the TC difference and PC classification that indicated areas in a state of clearing between 1984 and 1986, but

Table 2.2. Error matrix for the Tasseled Cap land cover change classification. User's accuracy and producer's accuracy are reported for each class. Overall accuracy was 79.3% and Kappa Statistic was 78.1%.

Class	Reference Data																		
	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	User's (%)
0 Forest 84-92	15																		100
1 Water 84-92		11		1	3														73
2 Cleared 84-86, Flooded 92			13																100
3 Flooded 84-86, Dry 92		1		14															93
4 Forest 84-86, Flooded 92		1			14														93
5 Forest 84, Cleared 86, Flooded 92			1			14													93
6 Regrowth 84-86, Flooded 92		1	1				12									1			80
7 Forest 84-86, Cleared 92								10						4	1				67
8 Cleared 84-92									7					2		2		2	54
9 Forest 84, Cleared 86-92										13						1			93
10 Forest 84, Regrowth 86, Cleared 92									2		10				2	1			67
11 Forest 84, Cleared 86, Regrowth 92												15							100
12 Forest 84-86, Regrowth 92													14						100
13 Regrowth 84-92														12	1	2			80
14 Forest 84, Regrowth 86-92														7	9				56
15 Regrowth 84, Cleared 86-92									1	2						13			81
16 Regrowth 84-86, Cleared 92									1							6	8		53
17 Cleared 84, Regrowth 86-92								1	6							1		7	47
Producer's Accuracy (%)	100	79	87	93	82	100	100	91	41	87	100	100	100	46	69	50	100	78	
Overall Accuracy (%)																			79.3
Kappa Statistic (%)																			78.1
Kappa Statistic (classes 0, 3, 7, 8, 9, 10, 11, 12, 13, 14, 15, 17)																			72.5

Table 2.3. Error matrix for the Tasseled Cap with Principal Components land cover change classification. User's accuracy and producer's accuracy are reported for each class. Overall accuracy was 68.4% and Kappa Statistic was 65.8%.

	Reference Data																			
Class	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	User's (%)
0 Forest 84-92	14												1		1					88
1 Water 84-92																				
2 Cleared 84-86, Flooded 92																				
3 Flooded 84-86, Dry 92				14																100
4 Forest 84-86, Flooded 92																				
5 Forest 84, Cleared 86, Flooded 92																				
6 Regrowth 84-86, Flooded 92																				
7 Forest 84-86, Cleared 92								14					1							93
8 Cleared 84-92									7					1		2		2	1	54
9 Forest 84, Cleared 86-92									1	5		7	2		1					31
10 Forest 84, Regrowth 86, Cleared 92									3		4				6			2		27
11 Forest 84, Cleared 86, Regrowth 92										1		13				1				87
12 Forest 84-86, Regrowth 92													15							100
13 Regrowth 84-92									2		1			8		2		1		57
14 Forest 84, Regrowth 86-92	3										1		4		8					50
15 Regrowth 84, Cleared 86-92														2		13				87
16 Regrowth 84-86, Cleared 92																				
17 Cleared 84, Regrowth 86-92									3					1				11		73
18 Cleared 84-86, Regrowth 92									4			1				1		2	6	43
Producer's Accuracy (%)	82			100				100	35	83	67	62	65	67	50	68		61	86	
Overall Accuracy (%)																				68.4
Kappa Statistic (%)																				65.8
Kappa Statistic (classes 0, 3, 7, 8, 9, 10, 11, 12, 13, 14, 15, 17)																				64.9

Table 2.4. Error matrix for the Tasseled Cap Image Differencing. User's accuracy and producer's accuracy are reported for each class. Overall accuracy was 71.4% and Kappa Statistic was 66.7%.

Class	Reference Data																				
	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	User's (%)
0 Forest 84-92	13												1								93
1 Water 84-92		13		1																	93
2 Cleared 84-86, Flooded 92																					
3 Water 84-86, Dry 92				15																	100
4 Forest 84-86, Flooded 92	1				14	1															88
5 Forest 84, Cleared 86, Flooded 92		1			5	9															60
6 Regrowth 84-86, Flooded 92																					
7 Forest 84-86, Cleared 92								15	1												94
8 Cleared 84-92									14	1											93
9 Forest 84, Cleared 86-92									5	7						3					45
10 Forest 84, Regrowth 86, Cleared 92					1	2			1		6			2		1	1	1			40
11 Forest 84, Cleared 86, Regrowth 92										3		8				2					62
12 Forest 84-86, Regrowth 92												2	14								88
13 Regrowth 84-92														15							100
14 Forest 84, Regrowth 86-92									1		1			4	5			3	1		33
15 Regrowth 84, Cleared 86-92									5							8			2		53
16 Regrowth 84-86, Cleared 92														6		2	7				45
17 Cleared 84, Regrowth 86-92									1									9	6		56
18 Cleared 84-86, Regrowth 92												1							5		83
19 Water 84, Cleared 86, Water 92									1	4										8	62
Producer's Accuracy (%)	93	93		94	70	75		100	48	45	86	73	93	56	100	50	88	69	36	100	
Overall Accuracy (%)																					71.4
Kappa Statistic (%)																					66.7
Kappa Statistic (classes 0, 3, 7, 8, 9, 10, 11, 12, 13, 14, 15, 17)																					67.9

had changed to regrowth by 1992 (Class 18). The TC difference had a high accuracy for Class 18, but the PC classification only had moderate accuracy in delineating this class. Interpretation of this class is likely due to confusion with areas in a state of clearing for the study period. A second class that was missed by the TC classification, but was captured by the TC difference classification, was water in 1984, cleared (bare soil) in 1986, and water in 1992 (Class 19). This class is associated with the variation in water levels in the tin mine reservoir and is not a change of interest for this study. The TC classification performed better than the other classifications in capturing clearing of both primary forest and areas in a stage of regrowth.

The TC classification gave the highest accuracy of all three approaches with an overall accuracy of 79.3% and a Kappa Statistic of 78.1% (Table 2.2 and Figure 2.3). Additionally, the Kappa Statistic for the 12 classes that corresponded to all three change detection approaches was 72.5%. The TC classification gave high producer's accuracies (87%-100%) and user's accuracies (67%-100%) for classes of primary forests converted to clearing (Classes 5, 7, 9, 10, 12). For classes of primary forest that are cleared and followed by regrowth (Classes 11 and 14), the producer's accuracies were 69% and 100% and user's accuracies were 56% and 100%. For the class that were interpreted as being in a state of sustained clearing (Class 8), the producer's accuracy was 41% and the user's accuracy was 54%. The low accuracy is associated with spectral confusion of regrowth areas. We suspect that some of the uncertainty in the classification is associated with sprouting of surviving vegetation and the establishment and rapid growth of secondary broad-leaved plants in pastures. This situation could be exhibiting relatively high brightness and greenness similar to vegetation regrowth in shifting cultivation sites. Therefore, some of the error may be overestimated. Similarly, a class interpreted as clearing initially, but was allowed to regenerate (Class 17) gave a producer's accuracy of 78% and a user's accuracy of 47%. In other words, although 78% of all areas cleared and allowed to regrow were correctly identified as this class, only

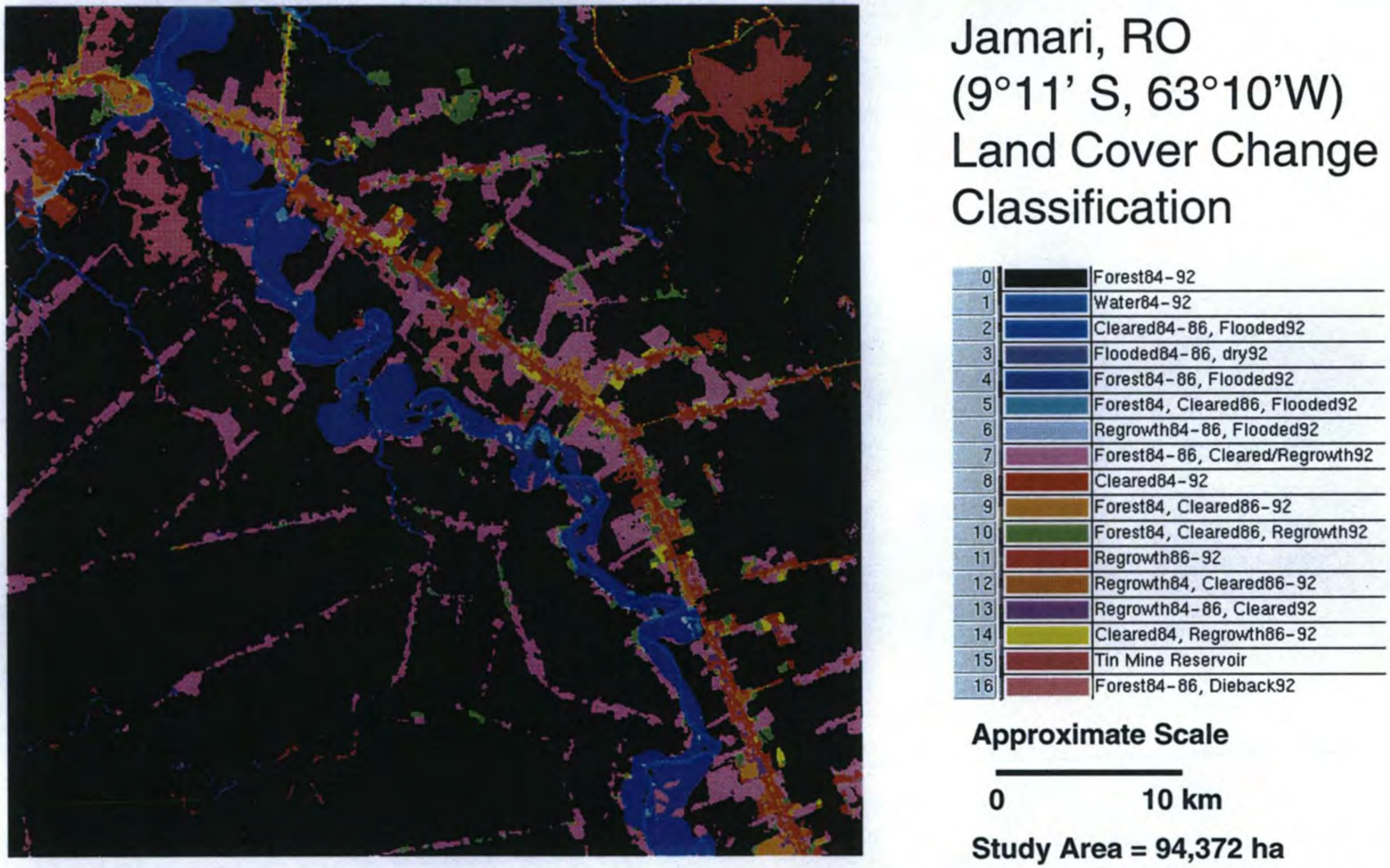


Figure 2.3. Classification of theTasseled Cap composite image (1984, 1986, and 1992 TC indices) for the Jamari, Rondônia study area. Land cover change classes include primary forest, regrowth (regenerating forest/cultivation), cleared areas (pasture/cultivation), flooded (Samuel Dam), dieback (prior inundation), water (Jamari River and other tributaries), and reservoir (tin mine).

45% of the areas identified as "clearing followed by regrowth" within the classification were actually that class. The spectral confusion associated with this class was with assignment of regrowth areas to clearing. Because it is impossible to ground-truth historic satellite, data there could be some error in our visual interpretation of this class in the TC images through time. Perhaps some of the error is overestimated. Additionally, this class represented less than 1% (351 ha) of the study area and hence does not lessen the high overall accuracy of this classification.

The TC difference classification had an overall accuracy of 71.4% and a Kappa Statistic of 66.7% (Table 2.4). The Kappa Statistic for the 12 classes present in the three approaches was 67.8%. The TC Principal Components classification was the most computationally intensive and time consuming technique, both in terms of time for computation of components for multiple dates and for interpretation of components, eigenvectors, and factor loadings, but was the least reliable for capturing and identifying the type of change. The TC Principal Components classification yielded a 68.4% overall accuracy and a 65.8% Kappa Statistic (Table 2.3). The Kappa Statistic for the 12 classes present in the three classifications was 64.9%.

The TC land cover change map was selected for land cover change analysis due to the classification's high accuracy and number of classes (Table 2.2 and Figure 2.3). Four of the 16 classes exhibited no change and consisted of primary forest, regrowth, cleared status, or water (including the Jamari River and the tin mine reservoir) for all dates. The classification was successful in delineating the original extent of the Jamari River as well as the extent following the completion of the Samuel Dam in 1989. The classification shows that there were 5,443 ha of primary forest lost from the flooding and rise in the river level and associated tributaries (Table 2.5). This area remained under water in 1992. In addition, an area of 1,270 ha was lost due to inundation, but the water had receded leaving an area mostly devoid of green vegetation with relatively dry, bare soil resembling clearing in the 1992 data.

As of 1984, ~4,738 ha or nearly 5% of the 94,372 ha study area had been cleared (including areas in regrowth) for agriculture or for the tin mine (Table 2.5). Between 1984-1992, 16,658 ha of additional primary forest were lost due to conversion to agriculture, pasture, or inundation. This loss of primary forest represents nearly 18% of the study area. This contributed substantially to the study area's primary forest conversion (i.e., cleared, regrowth, inundation, and tin mine) total of 23% (21,396 ha) as of 1992. In 1984, ~1,231 ha, or 1% of the study area, was in a state of clearing and remained cleared by 1986. This area remaining in a cleared state is likely pasture, as shifting cultivation sites would have evidence of regrowth due to infrequent clearing. By 1986, the area in clearing/pasture increased to ~2,692 ha, or 3% of the study area, and remained cleared as pasture until 1992. In addition, we found that between 1984-1992, only ~615 ha of regenerating forest had been cleared and even less (~213 ha) between 1986-1992. For each of these time periods, clearing of regenerating forest accounted for less than 1% of the study area.

In 1984, both areas in regrowth and in clearing each represented 2% of the study area (Table 2.5). In 1986, new areas in regrowth and areas remaining in regrowth were each 2% of the study area. Forest areas cleared between 1984 and 1986 represented over 3% of the study area and these cleared areas that remained cleared were over 1% of the study area. New areas in regrowth increased at a lower rate than new areas in clearing increased between 1984 and 1986, but the total area of each was essentially equal and each represented about 4% of the study area. By 1992, areas previously cleared then allowed to regenerate represented less than 2% of the study area and areas remaining in regrowth between 1986 and 1992 dropped to 1% of the study area. Between 1986 and 1992, 8% of the study area was cleared and 3% remained in a state of clearing. During this period, the total area in a state of clearing increased whereas the area in regrowth decreased.

Table 2.5. Land cover change classes interpreted from the Tasseled Cap (TC) composite classification which included TC indices for years 1984, 1986, and 1992. Land cover classes include primary forest, regrowth (regenerating forest/shifting cultivation), cleared areas (pasture/shifting cultivation), flooded (Samuel Dam), dieback (prior inundation), water (Jamari River and other tributaries), and reservoir (tin mine). The Jamari, Rondônia study area encompasses 94,372 ha.

Land Cover Change Class	Area (ha)	Percent of Study Area
Forest 1984		
No change	71,881	76.2
Flooded in 1992	5,443	5.8
Flooded dieback 1992	1,270	1.3
Cleared/regrowth by 1992	7,352	7.8
Cleared/regrowth by 1986		
Flooded in 1992	192	0.2
Regrowth in 1992	1,503	1.6
Cleared in 1992	898	0.9
Regrowth 1984		
No change	1,305	1.4
Flooded in 1992	130	0.1
Cleared by 1992	213	0.2
Cleared by 1986		
Cleared in 1992	615	0.7
Cleared 1984		
Regrowth 1986-1992	352	0.4
Cleared 1986, Flooded in 1992	51	0.1
No change	1,180	1.2
Reservoir 1984-1992 (tin mine)	892	0.9
Water 1984-1992	1,095	1.2
Total	94,372	100.0

We used three years of Landsat TM data from the same path and row locations between 1984 and 1992 during the dry season to detect extent of deforestation and the extent and frequency of clearing in the Amazon forest region of Rondônia. Our knowledge of land use in Rondônia from fieldwork and interviews with landowners is that sites continuously cleared during the study period were indicative of pastures, whereas, clearings interrupted by regrowth through time indicated shifting cultivation. Classes indicating pasture and shifting cultivation land use sequences emerged from the TC classification. If a site remained cleared between two dates analyzed during the study period, the site was presumed burned by each date to maintain pasture. For example, the class interpreted as forest in 1984, cleared/regrowth in 1986, and cleared in 1992 indicated that primary forest had been cleared and burned between 1984 and 1986 and that there was subsequent burning between 1986 and 1992 to maintain a state of clearing (Table 2.5). Since this class had been cleared by 1986 and remained in a state of clearing by 1992, we assumed that this is an example of pasture maintenance. Cultivation is possible during the 1986-1992 period. However, because clearing was evident in the 1992 image and the timing of the 1992 image was early in the dry season and likely before clearing for the season occurred, the clearing identified corresponded to 1991. Therefore, only a five-year shifting cultivation cycle would have occurred. We assumed that it was more likely to have remained cleared for pasture. Under the typical pasture burning practices of this region, there were likely two to three pasture burning events for this class between 1986 and 1992. Another class likely following characteristics of pasture maintenance is the class that was in a state of regrowth in 1984 and then was in clearing by 1986 and remained clear by 1992. This class was likely a fallow shifting cultivation site in 1984 with regenerating forest. By 1986, this site was cleared and presumably converted to pasture in 1986 and maintained as pasture through 1992. This class represents less than 1% of the study area. This site experienced a regenerating forest slash burn followed by a minimum of two pasture burns. A clearer example of a pasture site scenario is for the class that was in a state of clearing for all dates (i.e., 1984, 1986, and 1992) and here we assumed a minimum of three burning events under a two to three year pasture burning scenario.

If regrowth was prevalent during the study period, shifting cultivation practices or regenerating forest was assumed and site burning only occurred when clearing was evident. The class depicted as regrowth between 1984-1986 and then was cleared in 1992 indicates a regenerating forest slash burn between 1986 and 1992. Due to the infrequency of clearing, we assume this class likely represents a shifting cultivation scenario. Another possible example of shifting cultivation was found in the class that was forest in 1984, was in a state of clearing or regrowth in 1986, and continued to be in a state of vegetation regeneration by 1992. This particular shifting cultivation class had an initial primary forest burn, but was likely not followed by subsequent burning but to cultivation or second-growth forest establishment.

Deforestation of primary and regenerating forest was easily detected in the 30-m resolution classification of the multi-date TC composite. Most of the inaccuracy of the classification was associated with spectral confusion of some regrowth areas that were classified as being in a cleared state. If these regrowth areas were actually young pastures with recently planted pasture grasses of fairly dense regenerating forest vegetation cover, these conditions could exhibit both relatively high brightness and greenness and give an indication of regrowth.

During the study period, the area of primary forest that was cleared comprised 8,250 ha and represented 9% of the study area (Table 2.5). We assume this scenario to correspond to primary forest conversion to pasture. In contrast, primary forest area cleared and allowed to regrow as second-growth forest or become cultivated represented less than 2% (1,503 ha) of the study area. Regenerating forest areas were about equally likely to be cleared again or continue forest establishment. Each scenario represented about 1% of the study area.

Finally, cleared areas remaining cleared (pasture) during the study period included 1,180 ha and represented over 1% of the study area whereas areas cleared then allowed to regrow represented less than 1% (352 ha) of the study area. Additional areas that were forest or in a state of clearing prior to 1992, but were flooded by 1992 due to the completion of the Samuel Dam in 1989 were not interpreted as one of the above land use scenarios due to their flooded condition. These areas lost to flooding comprised an area of 7,086 ha and represented 8% of the study area. This time series of imagery supports observations that primary forest was more commonly converted to pasture than allowed to establish as second-growth forest in this area of subsistence agriculture in Rondônia.

Discussion

Previous studies in the Amazon have used single-date and multi-date satellite data to quantify deforestation, but most have neglected to identify shifting cultivation and pasture clearing [*Alves and Skole*, 1996; *Frohn et al.*, 1996; *Moraes et al.*, 1998; *Rignot et al.*, 1997; *Skole and Tucker*, 1993; *Stone et al.*, 1991; *Woodwell et al.*, 1986]. A shortcoming of much of this previous research is that the research emphasis is deforestation. In terms of quantifying regional and global emissions, elemental pools, and losses more accurately, land clearing and burning estimates are needed other than from conversion of primary forest. Many of the methods of previous research could be useful in identifying land clearing and burning. *Frohn et al.* [1996] classified forested and cleared areas in Rondônia in Landsat MSS data for 1978 and 1980 and TM data for 1986. These classifications were compared with modeled simulations of clearing to describe the pattern of clearing in the study area, but did not classify type of clearing (i.e., pasture or cultivation). *Stone et al.* [1991] used 1980 MSS, 1986 TM, and 1988 AVHRR data to map the area and rate of deforestation in Rondônia to look at spatial trends of clearing and for comparison with information in federal statistical atlases. The multi-date analysis of *Stone et al.* [1991] could have indicated additional types of land cover conversion (i.e., areas remaining cleared versus areas allowed to regenerate). *Moraes et al.* [1998] used a single date of TM data for 1991 to map forests and pastures in a study area south of Ariquemes in Rondônia. *Moraes et al.* [1998] was able to map different ages of pastures with at least 59.5% accuracy (Kappa Coefficient). The objective was to assign vegetation and soil carbon stocks to land cover types for estimating carbon pools and fluxes. A multi-date TM analysis may improve the accuracy of identifying pasture age for this study. *Alves and Skole* [1996] used a time-series of Systeme Pour L'Observation de la Terre (SPOT) data (1986, 1988, 1989, 1991, and 1992) to map deforestation near Ariquemes, Rondônia and to separate secondary vegetation regrowth within the deforested areas. Although the accuracy of these classifications was not reported, it was stated that a limitation was that cacao plantations could not be separated from secondary vegetation. The methods of *Alves and Skole* [1996] go beyond estimating deforestation to estimate secondary vegetation regrowth. It is likely that areas that remained cleared following deforestation, indicative of pasture, could have been identified using these methods. *Rignot et al.* [1997] analyzed multi-resolution data of Spaceborne Imaging Radar C (SIR-C) for 1994, TM data for 1993, SPOT data for 1986, 1988, 1989, 1991, 1992, and 1994, and Japan Earth Resources Satellite (JERS-1) radar data for 1994 and 1995 to map deforestation and secondary growth in Rondônia. The combination of the radar data classifications with the TM classification allowed discrimination of forest, non-forest with no woody biomass, recent clearings with slash of high woody biomass, initial regrowth, intermediate regrowth, flooded dead forest, and open water. The accuracy of this combined classification gave a Kappa Coefficient of 93%. The researchers add that there was substantial variability of woody biomass within classes and they suggested that the classification was not appropriate for estimating biomass inputs for carbon models. These methods, however, likely identify land conversion rates and land cover change through time *Woodwell et al.* [1986] used multi-date MSS data for 1976, 1978, and 1981 to map forest to non-forest change in Rondônia. The methods could not determine change from bare ground to agriculture or pasture. In addition, the 1981 MSS forest clearing was compared to a 1982 AVHRR scene to develop methods to scale deforestation to the lower resolution AVHRR data. Finally, *Skole and Tucker* [1993] visually interpreted deforestation in black and white photos of TM mid-infrared data for 1988 Amazon deforestation. Deforestation for 1978 was digitized from deforestation maps derived from single-channel MSS data. Deforestation, forest fragments, and edge effects by Amazon state were reported for both dates, however, the nature of this comprehensive project and the state of computational capabilities wouldn't have allowed a feasible means for analysis of land cover beyond forest and non-forest.

We found that analysis of a third date, during an eight-year period, provided the information on clearing needed to delineate a pasture from a shifting cultivation scenario. Additional TM around the year 1989 could have improved the analysis since the six-year gap in our selected data from 1986 to 1992 may have confused interpretation of land cover change classes. The ability to identify clearing events and the type of land cover type cleared using our methods is valuable as clearing gives an indication of timing of burning. The value of predicting the extent and frequency of burning events and the type of land cover burned is that this data can be used to improve estimates of emissions from biomass burning and site elemental pool losses.

The utility of our methods in expanding the analysis to the state of Rondônia could be tested in other areas of the state experiencing the range of deforestation from early colonization to extensive agricultural expansion. The analysis of the full TM scene for each of the dates in our study period would be a logical next step for analysis and testing whether the methods are appropriate at the regional scale. Since the TC transformation reduces the dimensionality of the data, a TC composite of dates covering a full TM scene is appropriate. The computational limit in a TC composite classification would likely be due to the number of dates in the analysis. In addition, the date interval for the time series might be appropriate at two to three year intervals, whereas, intervals of four to six years in areas of less rapid change may be appropriate. Both date interval selection of data and length of the time series are considerations for computational limitations. Based on the utility of our methods, we suggest that mapping forest clearing and areas in pasture/agriculture could improve estimates of deforestation, regrowth (regenerating forest/cultivation), and pasture at a regional scale. Additionally, as TM data are move available with the launch of the Landsat 7 satellite in 1999 with a repeat cycle offset from Landsat 5 by eight days, TM data is available every eight days for as long as Landsat 5 is operational. The use of high resolution TM data has recently become more feasible for regional analysis of multi-date imagery and will likely improve a variety of regional estimates associated with land cover/land use and change.

Conclusion

The importance of this multi-date TC classification was the reliable detection of clearing and the ability to predict the type of land cover that was cleared given the clearing rate. Deforestation fires can be estimated along with the minimum number of subsequent site burns. It is important to distinguish primary forest burns from fires used to maintain pasture and to burn fallow fields for cultivation since the nature of the emissions and the biomass, carbon, and nutrient contents are very different. Together, knowledge of land use pattern and multi-date TM imagery, instead of single TM date analysis, led to better interpretation of land cover types, contributed to resolving problems of interpreting the land use following clearing, and assisted quantification of burning frequency. Hence, estimates of resultant emissions and terrestrial carbon and nutrient pools and losses associated with deforestation and land conversion are likely improved.

Land cover change detected from this multi-date TC analysis can be useful in studies to quantify sources of atmospheric emissions associated with various land use burning practices and assessing their overall contribution of emissions to the region. This could be accomplished by a combination of ground-based data of biomass, carbon, and nutrient dynamics with remote sensing data to better quantify local and regional elemental sinks, sources, and atmospheric emissions.

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Chapter 3

Modeling Biomass Burning Emissions for Jamari, Rondônia, Brazil

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Abstract

Biomass burning emissions associated with deforestation in the Amazon are considerable. Fire is a ubiquitously used tool in land management and once deforested, these lands continue to be sources of burning emissions for many years. For example, pastures are maintained through burning every two to three years and produce considerable emissions. Using results from ground-based measurements combined with Landsat TM data, we estimated carbon emissions from three sources. Fires associated with: 1) primary forest slash; 2) regenerating forest slash; and 3) pastures. The study area was a 94,372-ha section of land surrounding Jamari, Rondônia, Brazil. In the emissions computation, we integrated the site-specific, ground-based data (biomass, emission factors for flaming and smoldering combustion, and combustion factors for land cover types) with a Landsat TM-delineated land cover change map for 1984-1992 for the study area. This map was used to delineate changes in land cover for time periods 1984-1986 and 1986-1992 based on TM image dates used in the analysis and corresponding to a period of early colonization and rapid deforestation in Rondônia. Flaming and smoldering combustion emissions of CO, CO_2, CH_4, and other hydrocarbon trace gases were calculated. On a study area basis between 1984 and 1992 primary forest slash burns generated 104,294 Mg C (138 kg C $ha^{-1}yr^{-1}$) in CO, 694,379 Mg C (920 kg C $ha^{-1}yr^{-1}$) in CO_2, and 12,409 Mg C (16 kg C $ha^{-1}yr^{-1}$) in CH_4. Regenerating forest slash burning contributed 3,226 Mg C (4 kg C $ha^{-1}yr^{-1}$) in CO, 23,436 Mg C (31 kg C $ha^{-1}yr^{-1}$) in CO_2, and 376 Mg C (<1 kg C $ha^{-1}yr^{-1}$) in CH_4. Whereas pasture burning during this same period produced 34,666 Mg C (46 kg C $ha^{-1}yr^{-1}$) in CO,

238,180 Mg C (316 kg C $ha^{-1}yr^{-1}$) in CO_2, and 2,993 Mg C (4 kg C $ha^{-1}yr^{-1}$) in CH_4. Primary forest slash burns contributed to 73% of the C emissions arising from fire in the study area for the eight-year study period. Regenerating forest slash burning produced only 2% of the C emitted and pasture burning generated 25% of the C emissions. While primary forest slash fires are the predominant source of emissions, during the period of colonization in Rondônia, annual emissions from pasture burning increases as the area in pasture increases, hence, pastures are becoming a more important source of biomass burning emissions in the Amazon.

Introduction

Since the early 1980s, it has been recognized that biomass burning, particularly in the tropics, contributes tremendous amounts of greenhouse gases and aerosols to the troposphere [*Andreae et al.*, 1988; *Crutzen and Andreae*, 1990; Houghton, 1990; Ward *et al.*, 1992; *Allen and Miguel*, 1995; *Fearnside*, 1997]. Increased concentrations of trace gases, such as CO_2, CH_4, N_2O, and O_3, from biomass burning are potentially involved in the warming effect of Earth by preferentially absorbing thermal infrared light in the troposphere and redirecting this energy back to the Earth [*Baird*, 1999]. Although the quantity of CH_4 emissions from burning are about 0.5-1.5% of CO_2, CH_4 has 25 times the radiative effect [*Houghton*, 1990]. Important tropical sources of CH_4 include deforestation, cattle ranching, burning of pastures, grassland burning, and burning of fuelwood, and rice cultivation [*Houghton*, 1990]. CO emissions, generated by smoldering combustion, are not radiatively active, but interact with the hydroxyl radical (OH) and reduce the oxidizing capacity of OH to remove gases such as CH_4 from the troposphere [*Baird*, 1999; *Fearnside*, 1997]. Aerosols, generated during the combustion process, modify climate by intercepting and reflecting sunlight and also act as nuclei in the formation of water droplets (cloud condensation nuclei) that also reflect solar radiation [*Allen and Miguel*, 1995]. The prevalence of aerosols associated with biomass burning may augment a cooling effect in the troposphere.

Fuel moisture content, fuel mass, and fuel chemistry, along with fire intensity which is a reflection of vegetation type, topography, and weather, determine the composition and quantity of emissions from biomass burning [*Ward*, 1990]. Furthermore, the amount of biomass consumed and the resultant composition of gaseous and particulate emissions are determined by the combustion factor and combustion efficiency of fuels, which in turn depend on ecosystem characteristics, environmental characteristics (humidity, temperature, windspeed), and ignition type [*Lobert and Warnatz*, 1993; *Ward et al.*, 1992]. Flaming and smoldering phases of combustion are also influenced by fuel factors of placement, size class and quantity, moisture, and chemistry [*Ward*, 1990].

Flaming combustion is a rapid oxidation reaction where volatile fine fuels are consumed [*Ward*, 1990]. The flaming phase of combustion occurs at higher temperatures and is a more complete combustion of fuels (i.e., a higher combustion efficiency and amount of CO_2 produced) [*Lobert and Warnatz*, 1993; *Ward*, 1990]. Smoldering combustion is a low-temperature process and is a less efficient combustion resulting in increased amounts of aerosols (incompletely oxidized compounds) and CO. In the tropics, smoldering combustion predominates as the primary mode of combustion for coarse wood within forest slash while flaming combustion predominates for grasses and fine fuels of pastures (*J.B. Kauffman*, personal communication). The more coarse particles of flaming and smoldering combustion will be deposited close to the burning source potentially contributing to local site nutrient losses, whereas finer particles and gases can be transported great distances from the burning site and would represent ecosystem losses of nutrients [*Allen and Miguel*, 1995; *Kauffman et al.*, 1992].

Uncertainties exist in the net release of carbon to the atmosphere in the tropics [*Houghton*, 1991]. Factors contributing to these uncertainties include: 1) the estimates of carbon stocks of tropical forests especially those subject to thinning and degradation, regenerating forests, and pastures; and 2) deforestation rates, and subsequent use (i.e., permanent or temporary clearing) of deforested areas. *Houghton* [1991] reported that some of the variation in deforestation rates was attributed to measurement, but primarily the variation appears to be from year-to-year deforestation influenced by the length of the dry season and, hence, length of the burning season. Additionally, deforestation rates are driven by economic factors [*Browder and Godfrey*, 1997; *Hecht*, 1993; *Molion*, 1991]. Deforestation for conversion to cattle pastures, a means of claiming land, is economical and considered low risk [*Hecht*, 1993]. There is less labor involved in cattle ranching than in producing crops of rice, beans, corn, manioc, or tree crops. Also, cattle pastures are in continuous production longer (~10 years) than crops (~2-3 years).

In the Brazilian Amazon, biomass burning emissions increased in the 1980s due to accelerated deforestation rates related to colonization projects that created fiscal incentives for agricultural expansion in remote areas of the Amazon [*Browder and Godfrey*, 1997; *Hecht*, 1993; *Molion*, 1991]. Burning associated with deforestation, land conversion, and pasture management in the Brazilian Amazon decreases the pools of terrestrial carbon and nutrients, and is a source of regional air pollution. Traditionally, biomass burning is used to convert tropical forests to shifting agriculture, permanent agriculture, and/or cattle pasture. Shifting cultivation is typically a cycle of a two-year cultivation period followed by a four-year fallow period [*Guild et al.*, 1998]. Based on interviews with landowners, this crop/fallow cycle is generally repeated up to two times before the land is rendered unproductive for crops at which time it is abandoned or converted to pasture. In contrast to shifting agriculture that results in slash fires every six years, pastures are burned purposefully or accidentally every two to four years [*Guild et al.*, 1998; *Kauffman et al.*, 1998]. Because of the frequency and extent, pasture burning could be a considerable source of carbon and other emissions to the atmosphere [*Kauffman et al.*, 1998]. Unfortunately, there are only a few studies that have contributed information on the impacts of land use in the Amazon [*Fearnside*, 1990; *Kauffman et al.*, 1995, 1997, 1998; *Uhl and Kauffman*, 1990; *Uhl et al.*, 1988]. Particularly limiting are data on losses of carbon and nutrients from burned sites. Collectively, these site losses could combine to become substantial regional sources of greenhouse emissions, other radiatively active trace gasses, and aerosols [*Andreae,* 1991, 1993; *Crutzen and Andreae*, 1990; *Crutzen et al.*, 1985; *Ward et al.*, 1992].

In Para and Rondônia, Brazil, mean prefire total aboveground biomass of primary forest slash and livestock pasture is 349 Mg ha^{-1} and 74 Mg ha^{-1}, respectively [*Guild et al.*, 1998; *Kauffman et al.*, 1995, 1998]. In addition, mean nutrient losses from primary forest slash fires of this region are 88 Mg C ha^{-1}, 1181 kg N ha^{-1}, and 107 kg S ha^{-1}. In livestock pasture fires of this region, mean nutrient losses are 14 Mg C ha^{-1}, 199 kg N ha^{-1}, and 16 kg S ha^{-1} [*Guild et al.*, 1998; *Kauffman et al.*, 1995, 1998].

The objective of this research was to develop an approach to quantify biomass burning emissions at a local scale. From these methods we then quantified emissions for a forest area around Jamari, Rondônia, Brazil for 1984-1992; a period of early colonization and active deforestation in this region. To estimate carbon gas species emissions, we integrated and scaled up local-level ground-based data to the landscape level corresponding to a TM-derived land cover change map (Chapter 2).

Methods

Study Area

We examined land use/land cover change and fire emissions from 1984-1992 over a 94,372 ha area of primary forest in Rondônia, Brazil. The study area is located along the BR-364 highway in an area of recent colonization resulting in active deforestation during the period of our study. The BR-364 highway, which bisects the state, was opened in 1960 and was paved in 1984. This highway provided a route for colonists to migrate and claim land within the state and provided an overland transportation route to the Atlantic Coast [*Browder and*

Godfrey, 1997]. Areas along the BR-364 highway have been subject to intense deforestation for cultivation, cattle pastures, timber exploitation, and mining.

We chose to conduct our study in this region because of its relatively recent colonization and deforestation activity at the onset of our analysis (since the early 1980's) and this allowed quantification of emissions that occur when tropical forest landscapes are colonized. In addition, the presence of relevant ground-based data on prefire and postfire biomass, elemental pool dynamics, and fire chemistry were available [*Guild et al.*, 1998; *Hughes et al.*, in press; *Kauffman et al.*, 1995, 1998]. Other related research in the region is occurring south of Jamari from Ariquemes to Vilhena along BR-364 highway where more long-term deforestation, agriculture, and mining have occurred [*Fujisaka et al.*, 1998; *Moraes et al.*, 1998; *Neill et al.*, 1997; *Tucker et al.*, 1984]. Our study area and study period results from a landscape of early stages of disturbance may provide some interesting emissions comparisons with results from the landscapes of intense disturbance as they become available.

The 94,372-ha study area is centered at approximately 9°11' S and 63°10' W and about 100 km southeast of the state's capital, Pôrto Velho. The smaller town of Jamari is located along the Jamari River, a blackwater tributary of the Madeira River. The study area consists of both small landholdings of subsistence farms and large ranches. It is adjacent to the Jamari National Forest, which contains the Santa Bárbara Tin Mine.

The primary forest type of Rondônia and the study region is submontane open forest consisting of overstory broad-leaved canopy and subcanopy with an abundance of palms and vines [*Cummings*, 1998; *Departmento Nacional de Produção Mineral*, 1978; *Instituto Brasileiro de Desenvolvimento Florestal (IBDF) and Instituto Brasileiro de Geographia e Estatística (IBGE)*, 1993]. Soil types include red-yellow podzolic latosols and red-yellow latosols [*Neill et al.*, 1997].

Climatological data come from Pôrto Velho, Rondônia about 100 km north of the region. Mean annual precipitation is 2,354 mm [*Departmento Nacional de Meteorologia, Brasil*, 1992]. During the dry season, between June and September, mean precipitation is typically <100 mm per month. Dry season mean temperature is ~25°C ranging from a minimum of ~21°C to a maximum of 31°C with a mean relative humidity of 85%.

Emissions Model

Biomass burning emissions of CO, CO_2, CH_4, and hydrocarbon trace gases were computed following a model modified from that given by *Chatfield et al.* [1996] for biomass burning in Africa. This model is based on the rate at which fuel carbon burns per unit area and the weighted sum of emission factors for flaming and smoldering combustion [*Lobert et al.*, 1991]. We modified the emissions computation to include delineation of land cover types and partitioning of biomass or fuel categories by land cover type. Trace gas emissions from biomass burning are calculated as follows:

$$E_i = \sum_{l,k=1}^{lmax,\, kmax} (\eta_{il}^{\mathrm{flame}} f_{lk}^{\mathrm{flame}} + \eta_{il}^{\mathrm{smolder}} f_{lk}^{\mathrm{smolder}})\, m_i\, c_l\, b_{lk}^{\mathrm{TAGB}}\, f_{lk}^{\mathrm{burned}}\, a_l^{\mathrm{burned}}$$

Where

E_i: Mass of C in gas species i emitted (Mg);

$\eta_{il}^{\mathrm{flame}}$, $\eta_{il}^{\mathrm{smolder}}$: flaming and smoldering emission factors for gas species i by land cover type l (g emission/g fuel);

f_{lk}^{flame}, $f_{lk}^{\mathrm{smolder}}$: fraction of biomass of land cover type l burned during flaming and smoldering combustion of fuel class k;

m_i: mass of carbon in gas species i (g C/g emission)

c_l: carbon concentration (%) of biomass for each land cover type;

b_{lk}^{TAGB}: total aboveground biomass (g/ha) of fuel classes for land cover types;

f_{lk}^{burned}: fraction of biomass or fuel burned by land cover type and fuel class; and a_l^{burned}: area of burn (ha) by land cover type.

The emissions flux, E_i, of chemical species i is calculated as the sum over distinct land cover types l. These land cover types include primary forest, regenerating forest, and pasture that were identified as the land cover type prior to detection of clearing in a Landsat derived land cover change map for 1984-1992 (Chapter 2).

Emission factors for flaming and smoldering combustion, $\eta_{il}^{\mathrm{flame}}$ and $\eta_{il}^{\mathrm{smolder}}$, for CO, CO_2, and CH_4 from two primary forest slash sites and one pasture site were acquired by the SCAR-B project following methods of *Babbitt et al.* [1996] and *Ward et al.* [1992]. Hydrocarbon emission factors measured for SCAR-B additionally follow methods used in Africa [*Babbitt et al.*, 1996; *Hao et al.*, 1996]. This emissions measurement effort was in collaboration with an additional SCAR-B project that sampled biomass prior to the fire and collected postfire biomass and ash in these three sites [*Guild et al.*, 1998]. Two to three Fire-Atmosphere Sampling System (FASS) towers at each site collected flaming and smoldering combustion phase gases using real-time analyzers and gas sampling in canisters [*Babbitt et al.*, 1996]. *Ward et al.* [1992] provides a thorough description of the FASS tower systems used in Brazil. Real-time sensors measured CO and CO_2. A gas sampler was placed at the height of 8-14 m above the ground depending on the anticipated fire conditions, including flame height. Flaming and smoldering combustion phases were measured separately. Emission factors included CO, CO_2, CH_4, C_2H_2, C_2H_4, C_2H_6, C_3H_4, C_3H_6, and C_3H_8 (Table 3.1). Emission factors for regenerating forests were not sampled from the SCAR-B experiment. Therefore, we used the primary forest emission factors for regenerating forests. *Ward et al.* [1992] measured emission factors for one second-growth forest in Para in eastern Amazon, however, we chose not to use these emission factors because the relative emission factors estimated for this region of the Amazon are lower than those estimated for Rondônia.

It is important to weight emission factors differently for flaming and smoldering combustion since the carbon release rate differs greatly for these phases

Table 3.1. Emission factors for flaming and smoldering combustion in primary forest slash and pasture burning. Values are mean emission factors and standard errors from unpublished data collected during the 1995 Smoke, Clouds, and Radiation – Brazil (SCAR-B) Experiment (D. Ward, personal communication, USFS Missoula, MT). Flaming phase combustion emission factors are based on tower measurements recorded in the early stages of the fire and smoldering emission factors of fuel consumed were recorded in the later stages of the fire.

	Primary Forest		Pasture	
Emission	Flaming $g\ kg^{-1}$	Smoldering $g\ kg^{-1}$	Flaming $g\ kg^{-1}$	Smoldering $g\ kg^{-1}$
CO	95.21±5.32	169.21±2.81	79.39±5.97	174.10±2.87
CO_2	1664.52±9.60	1527.95±4.95	1697.52±10.66	1529.49±4.95
CH_4	5.46±0.39	11.91±0.21	2.93±0.37	9.03±0.18
C_2H_2	0.27±0.01	0.26±0.01	0.24±0.01	0.29±0.01
C_2H_4	1.20±0.06	1.53±0.03	1.02±0.07	1.45±0.03
C_2H_6	0.66±0.05	1.53±0.04	0.32±0.05	0.95±0.04
C_3H_4	0.06±0.01	0.07±0.01	0.04±0.00	0.08±0.01
C_3H_6	0.57±0.04	1.01±0.02	0.39±0.05	0.86±0.02
C_3H_8	0.21±0.02	0.51±0.02	0.08±0.01	0.27±0.02

of combustion [*Ward et al.*, 1992]. Weighting the emission factors for flaming and smoldering combustion yields a more representative estimate of emissions released from fire ignition to the end of the smoldering phase of combustion.

Fractions of fuels consumed in flaming and smoldering phases of combustion, f_{lk}^{flame} and $f_{lk}^{smolder}$, were weighted by fuel category (k) for each site. Based on *Ward et al.* [1992] and observations of fuels burning in the Amazon (*J.B. Kauffman*, personal communication), we assume that 90% of fine fuels (litter, dicot seedlings, rootmat, live and dead grass, attached foliage, and woody fuels < 2.55 cm in diameter) are consumed by flaming phase combustion and 10% in the smoldering phase. Intermediate fuels of 2.55 to 7.62 cm in diameter are equally (50%) consumed in the flaming and smoldering phases. For coarse woody fuels (fuel diameter ≥ 7.63 cm) and palms, 10% and 90% of the mass is consumed by flaming and smoldering combustion, respectively.

The term for the mass of carbon, m_i, in gas species i was used to convert each emission to the mass of C emitted per gas species. This term is calculated per emission as the mass of total C in the gas species divided by the total mass of the gas species.

Based on averages of measurements in this region and other research in the Amazon [*Fearnside*, 1991; *Guild et al.*, 1998; *Hughes et al.*, in press; *Kauffman et al.*, 1994, 1995, 1998; *Ward et al.*, 1992], the carbon concentration, c_l, per unit of biomass is ~50%. The emissions factors used in this research were estimated using the carbon mass balance method [*Babbitt et al.*, 1996; *Ward et al. 1992*]. As the emission factors used in the emissions calculation account for the 2:1 biomass/carbon ratio, there is no need to factor the carbon concentration into the computation separately. If, however, the emission factors do not account for the 2:1 biomass/carbon ratio and are reported as g C emission per kg C burned, then the carbon content term is used to multiply the weighted sum of emissions factors for flaming and smoldering combustion by a factor of two. This will give emission factors in terms of g C in kg of biomass or fuel burned.

Mean TAGB from ground-based data collected between 1992 and 1995 in the Jamari, Rondônia study area was used for the aboveground biomass term, b_{lk}^{TAGB}, in the emissions computation [*Kauffman et al.*, 1994, 1995, 1998; *Guild et al.*, 1998; *Hughes et al.*, in press] (Table 3.2). These ground-based measurements were conducted in five primary forest slash sites, five regenerating forest sites, and four pastures. The biomass measured in each of these sites (or land cover types, *l*) was partitioned into size classes and composition that differ in their influence on fire behavior. Biomass size classes, *k*, included litter, grass, dicot seedlings, rootmat, attached foliage, fine woody debris (< 7.63 cm in diameter), and coarse woody debris (≥ 7.63 cm diameter). Coarse woody debris was further distinguished as sound or rotten. Following fire, these fuel classes were again measured along with ash. This facilitated measurements of biomass consumption and site loss of nutrients and C pools. Mean TAGB for primary forest sites in Rondônia is 343 Mg ha^{-1}, 116 Mg ha^{-1} for regenerating forests, and 79 Mg ha^{-1} for pastures [*Guild et al.*, 1998; *Hughes et al.*, in press; *Kauffman et al.*, 1994, 1995, 1998](Table 3.2). In primary forests, the coarse fuels compose 69-89% of TAGB. However, in regenerating forests coarse fuels represent 35-70% of TAGB, whereas 62-82% of TAGB is coarse fuels in pastures. The coarse fuels of primary forests and pastures, therefore, are the source of a greater quantity of the emissions generated than from fine fuels. The impact of coarse fuels on emissions from burning regenerating forest slash is more variable since there is much variation in these types of fuels at different stages of regrowth. In pastures, the coarse fuels are the remnant logs from original forests that remain in the sites and decompose slowly. This remnant coarse woody debris dominates by mass over the fine fuel components in pastures. Through pasture reburning and decomposition, the remnant logs continue to contribute to site emissions through time. Details of the biomass measurements are described in *Kauffman et al.* [1995, 1998].

The proportion of biomass that is actually burned, f_{lk}^{burned} (or the combustion factor), was measured at sites for the study area between 1992 and 1995 [*Guild et al.*, 1998; *Hughes et al.*, in press; *Kauffman et al.*, 1994, 1995, 1998]. The mean

Table 3.2. Mean total aboveground biomass (TAGB) and combustion factors for selected land cover types in Rondonia, Brazil (1992-1995).

Land Cover Type	TAGB, Mg ha^{-1}	Combustion Factor, %	Reference
Primary Forest Slash			
Range	290-399	38-57	Guild et al. [1998], Kauffman et al. [1994, 1995]
Mean (n=5)	343±19	48±3	
Regenerating Forest Slash			
Range	71-178	47-63	Hughes et al. [in press]
Mean (n=5)	116±17	56±4	
Pasture			
Range	60-119	21-47	Guild et al. [1998], Kauffman et al. [1994, 1998]
Mean (n=5)	79±14	37±7	

combustion factor for all sites for each land cover type, *l*, was calculated as: 48% for primary forest slash; 56% for regenerating forest slash; and 37% for pastures (Table 3.2).

A multi-date land cover change map was derived from a time series of Landsat Thematic Mapper (TM) data for the study area (Chapter 2). The TM data were used in change detection analysis and interpretation of the resultant classification produced a land cover change map to identify land cover type and change. The TM data used included dry season images for June 1984, July 1986, and July 1992 that decreased variability associated with phenology. The land cover change map indicated change for a two-year period from 1984-1986 and a six-year period from 1986-1992. For example, primary forest in one date followed by either regenerating forest or a site in a cleared state identifies a deforestation event. A site remaining in a cleared state was indicative of pasture. Also, if a site was in a cleared state on an early date and then is showing regrowth on subsequent dates this would likely be a regenerating forest site or cultivation. There are numerous land cover change possibilities, but a time series of TM data along with knowledge of the land use practices allows reasonable identification of the land cover type and subsequent land use path. The analysis of the land cover change map identified changes in land cover and indicated deforestation and clearing (i.e., burning) events for regenerating forests and pastures. From this map, area estimates of primary forest slash, regenerating forest slash, and pasture burns were quantified (Table 3.3).

Assumptions and Approaches to Model Emissions

Because of the selection of TM data (1984, 1986, and 1992) used, the change detection methodology does not allow detection of the actual timing of burning events within the six-year period between 1986 and 1992. These burning events included subsequent burns to maintain areas in pasture as of 1986. Precise timing of deforestation and subsequent burning was also missed during this time period. What can be quantified from the land cover change identified in the map is the following: 1) the total area of primary forest in 1986 that was deforested by 1992; 2) the area of regenerating forest in 1986 that was deforested by 1992; and 3) the amount of area that remained in pasture (cleared) since 1986 (Table 3.3). Estimating emissions based only on total areas cleared and burned for the 1986-1992 time period is likely an underestimate since pastures are maintained by frequent burning (approximately every two to three years) [*Guild et al.*, 1998]. In addition, consideration of subsequent burning following deforestation events is needed since these areas are deforested primarily for conversion to pasture, otherwise, emissions associated with reburning of pasture are overlooked.

To address the problem of underestimating pasture burning emissions, we developed two approaches of pasture burning scenarios that account for the pasture burning practices representative of the region. The first approach used to predict burning events for land cover types during the 1986-1992 time period was based on the assumptions that 1) deforestation of both primary and regenerating forest occurred at a linear rate; and 2) that pasture burning frequency occurred every two years to maintain pasture or from accidental burnings. Therefore, based upon the total area of primary forest cleared (7,352 ha) between 1986 and 1992, we assume that deforestation occurred at an annual rate of 1,225 ha or a little over 1% of the study area was deforested each year. In the same manner, regenerating forest area was cleared at the annual rate of 36 ha. Since pasture is the predominant reason for deforestation in the region, we made the assumption that both primary and regenerating forests were converted to pasture following cutting and burning and that these new pasture areas were burned every two years. To quantify this we assumed that of the total area in pasture each year, half of this area was burned. Forest area converted to pasture for the 1984-1986 time period is included in the area of pasture for the 1986-1992 time period. Additionally, due to the completion of the Samuel hydroelectric dam in 1989, there was forest and pasture lost to flooding of the Jamari River which runs adjacent to the BR-364 Highway.

Table 3.3. Primary and regenerating forest conversion and areas remaining in pasture in the Jamari, Rondonia study area (94,372 ha) in Brazil (1984-1992) (Chapter 2).

Land Cover Type	Area (ha)	Percent of Study Area
1984-1986		
Primary Forest Clearing	2592	3
Regenerating Forest Clearing	615	<1
Area Remained Clear (Pasture)	1231	1
1986-1992		
Primary Forest Clearing	7352	8
Regenerating Forest Clearing	214	<1
Area Remained Clear (Pasture)	2692	3
Cumulative Area of Pasture Burned (2 yr)	17,777	n/a
Cumulative Area of Pasture Burned (3 yr)	11,851	n/a

n/a: Not applicable.

Therefore, it was necessary to exclude these flooded areas as a source of potential burning emissions after 1988. Pasture area burned annually is calculated as:

$\boldsymbol{P_t = \frac{1}{2}(PA_{t-1} + PF_{t-1} + RF_{t-1}) - PAF_t - PFF_t - RFF_t}$

Where

P_t: area of pasture burned at year t;

PA_{t-1}: area of pasture burned the previous year, $t-1$;

PF_{t-1}: area of primary forest cleared the previous year, $t-1$;

RF_{t-1}: area of regenerating forest cleared the previous year, $t-1$;

PAF_t: area of pasture lost to flooding at year, t;

PFF_t: area of primary forest flooded at year, t; and

RFF_t: area of regenerating forest flooded at year, t.

For the 1986-1992 time period the sum of the area of pasture burned, P_{tot}, is expressed as:

$\boldsymbol{P_{tot} = \sum P_t}$, where t represents years 1986 to 1992.

In contrast, in the second approach we assumed that pastures were maintained by burning every three years. Deforestation rates remained linear as in the first approach. However, of the total area in pasture each year, we assumed that a third of this area burned and the equation for pasture area burned annually, P_t, is modified to divide by three instead of two.

Results

On a per hectare basis, primary forests have the potential to emit the greatest mass of carbon emissions, whereas, pastures emit the least quantity of carbon emissions. Based only on a per hectare basis in each land cover type, primary forest slash burning emissions were 11 Mg C ha^{-1} in CO, 69 Mg C ha^{-1} in CO_2, and 1 Mg C ha^{-1} in CH_4. For regenerating forests burned, emissions were 4 Mg C ha^{-1} in CO, 28 Mg C ha^{-1} in CO_2, and <1 Mg C ha^{-1} in CH_4. Finally, emissions from pastures burned were 2 Mg C ha^{-1} in CO, 13 Mg C ha^{-1} in CO_2, and <1 Mg C ha^{-1}

in CH_4. Factors influencing the potential mass of carbon emissions are primarily weighted by the amount of biomass and the combustion factor. The emission factors for flaming and smoldering combustion influence the distribution of carbon emitted per gas species.

For the extent of the Jamari, Rondônia study area (94,372 ha), during the period of 1984 and 1986, emissions from burning cut primary forest (2,592 ha) were 27,188 Mg C (144 kg C $ha^{-1}yr^{-1}$) in CO, 181,013 Mg of C (959 kg C $ha^{-1}yr^{-1}$) in CO_2, and 3,235 Mg C (17 kg C $ha^{-1}yr^{-1}$) in CH_4 emitted (Figures 3.1 and 3.2). Study area emissions from burning regenerating forest (615 ha) that had been cut and left as slash were 2,395 Mg C (13 kg C $ha^{-1}yr^{-1}$) in CO, 17,398 Mg C (92 kg C $ha^{-1}yr^{-1}$) in CO_2, and 279 Mg C (2 kg C $ha^{-1}yr^{-1}$) in CH_4 emitted. Also, for these areas to remain in a cleared state, they had to be burned, otherwise there likely would have been evidence of regrowth. Regrowth would be an indication of regenerating forests or cultivation. Therefore, the entire area representing pasture (1,231 ha), burned once during this two-year period yielding 2,244 Mg C (12 kg C $ha^{-1}yr^{-1}$) in CO, 15,420 Mg of C (82 kg C $ha^{-1}yr^{-1}$) in CO_2, and 194 Mg C (1 kg C $ha^{-1}yr^{-1}$) in CH_4 emitted. Additional trace gas estimates of C_2 and C_3 hydrocarbon emissions for these land cover types include ethyne (C_2H_2), ethene (C_2H_4), ethane (C_2H_6), propyne (C_3H_4), propene (C_3H_6), and propane (C_3H_8) (Figure 3.2). Trends in these hydrocarbon trace gas estimates follow those of CO, CO_2, and CH_4, whereby emissions from primary forest burning dominate those from burning regenerating forests and pastures.

Primary forest slash (7,352 ha) emissions for the extent of the Jamari, Rondônia study area for 1986-1992 were 77,107 Mg C (136 kg C $ha^{-1}yr^{-1}$) in CO, 513,366 Mg of C (907 kg C $ha^{-1}yr^{-1}$) in CO_2, and 9,174 Mg C (16 kg C $ha^{-1}yr^{-1}$) in CH_4 emitted (Figures 3.3 and 3.4). The approximate annual flux of emissions from primary forest slash for this period is slightly lower than the 1984-1986 time period due to a 5% decrease in the average annual rate of deforestation from 1,296 ha yr^{-1} to 1,225 ha yr^{-1} in the 94,372 ha study area. Regenerating forest slash (213 ha) emissions were 831 Mg C (2 kg C $ha^{-1}yr^{-1}$) in CO, 6,038 Mg of C (11 kg C $ha^{-1}yr^{-1}$)

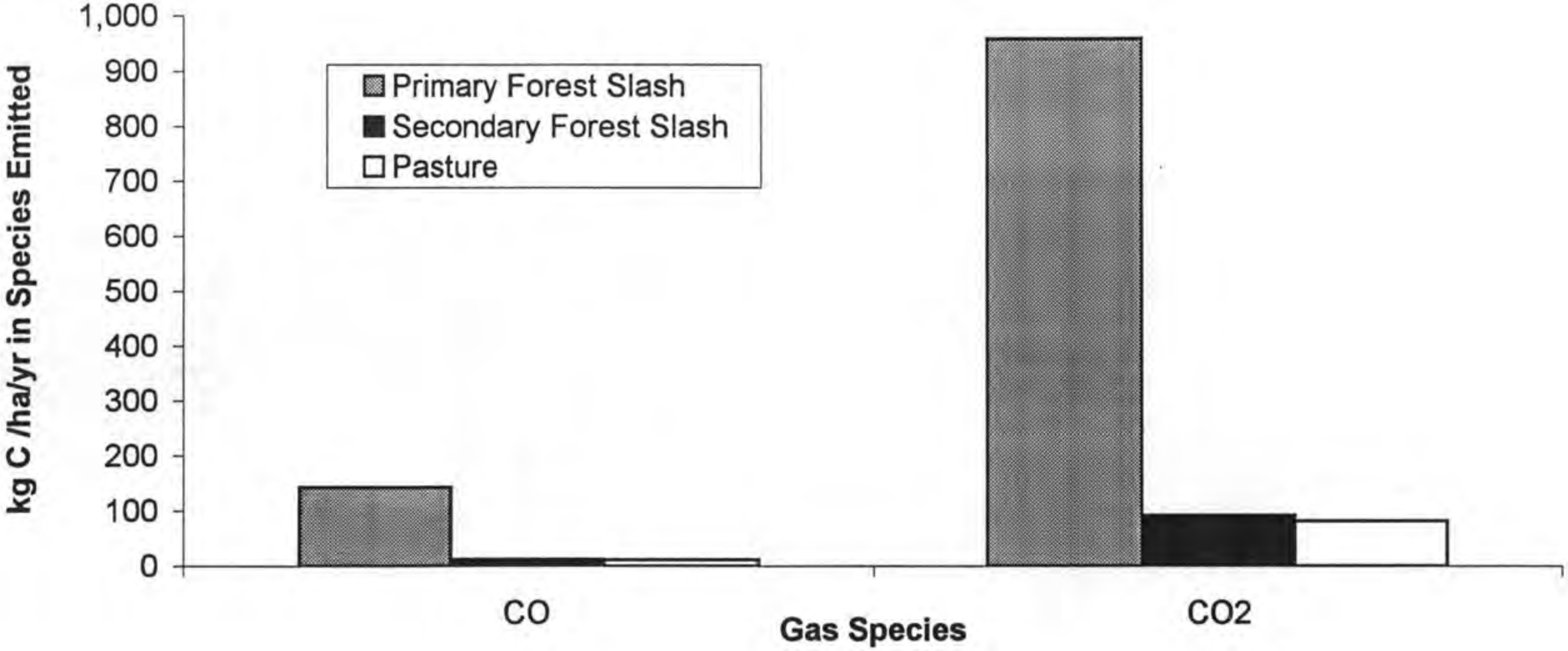

Figure 3.1. Estimated emissions of CO and CO_2 from fires in primary forest slash, regenerating forest slash, and pastures for the extent of the Jamari, Rondônia study area (1984-1986).

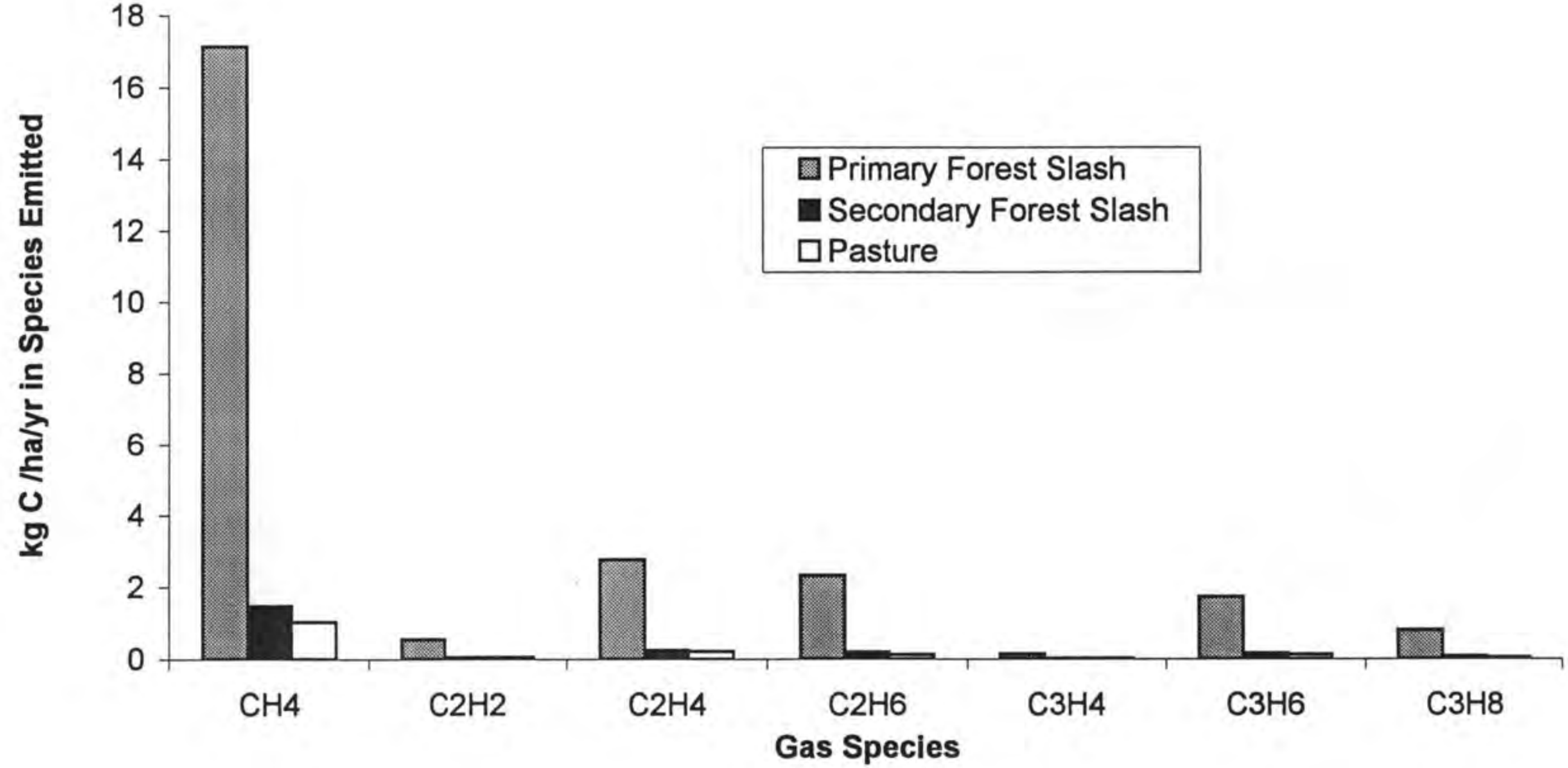

Figure 3.2. Emissions of CH_4 and trace gases arising from fires in primary forest slash, regenerating forest slash, and pastures for the extent of the Jamari, Rondônia study area (1984-1986).

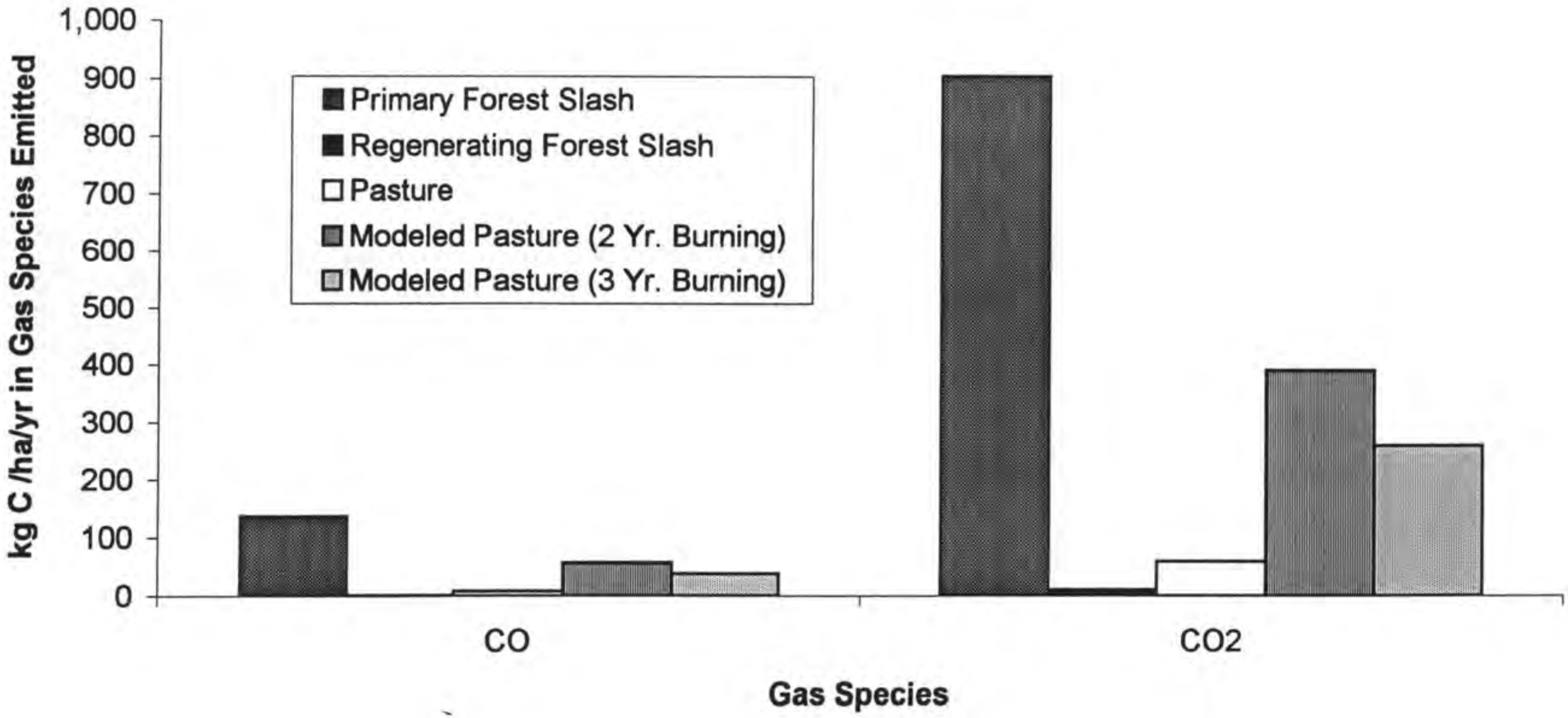

Figure 3.3. Emissions of CO and CO_2 arising from burning primary forest slash, regenerating forest slash, and pastures (burned once) for the extent of the Jamari, Rondônia study area (1986-1992). Modeled pasture emissions include pastures reburned on a frequency of two or three years during the study period.

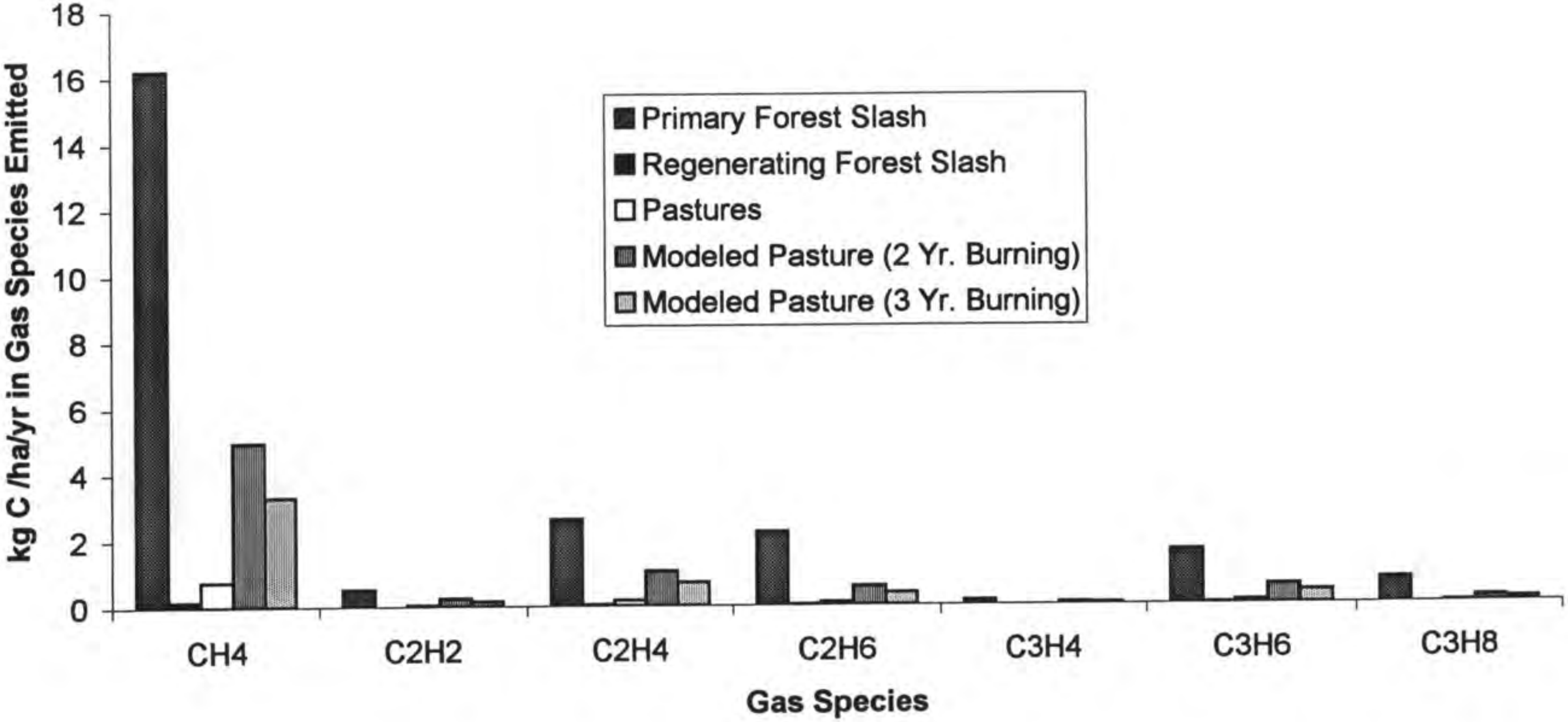

Figure 3.4. Emissions of trace gases from burning primary forest slash, regenerating forest slash, and pastures (burned once) for the extent of the Jamari, Rondônia study area (1986-1992). Modeled pasture emissions include pastures reburned on a frequency of two or three years during the study period.

in CO_2, and 97 Mg C (<1 kg C $ha^{-1}yr^{-1}$) in CH_4 emitted. Regenerating forest burning emissions on an annual basis were over eight-fold less than in 1984-1986. The contribution of emissions from fires in regenerating forests was substantially lower than from primary forest slash fires. This is attributed to the area in regrowth, which did not represent a large proportion (< 1%) of the study area. This supports evidence of land use practices of the region of primary forests being deforested predominately for the creation of cattle pastures. Finally, for the area that was in pasture in 1986 and remained in pasture from 1986 to 1992 (2,692 ha) and assuming this pasture area burned only once, resultant emissions were 4,910 Mg C (9 kg C $ha^{-1}yr^{-1}$) in CO, 33,738 Mg of C (60 kg C $ha^{-1}yr^{-1}$) in CO_2, and 424 Mg C (1 kg C $ha^{-1}yr^{-1}$) in CH_4 emitted.

Using the two-year burning scenario, the total area of pastures burned was 17,777 ha. Pasture emissions for this six-year time period were 32,422 Mg of C (57 kg C $ha^{-1}yr^{-1}$) in CO, 222,760 Mg of C (393 kg C $ha^{-1}yr^{-1}$) in CO_2, and 2,800 Mg C (5 kg C $ha^{-1}yr^{-1}$) in CH_4 emitted (Figures 3.3 and 3.4). This approach, which considers subsequent pasture burning following conversion of forest to pasture, indicates there was over a six-fold increase in the flux of C via CO, CO_2, and CH_4 emissions from pastures burned only once. In comparison to annual pasture emissions from 1984-1986, there was nearly a five-fold increase on an annual basis during the 1986-1992 time period. The pasture area burned during this period was over twice the area deforested, but the total quantity of pasture emissions was half that from primary forest slash burning. The relatively higher TAGB and the combustion factor of primary forest slash sites of the region are the primary factors influencing the higher mass of carbon emissions from these sites.

If pastures were burned on a three-year cycle, calculated as one-third of the area in pasture is burned each year, the total area of pasture reburned was 11,851 ha. For this burning cycle, emissions from pasture fires were 21,614 Mg of C (38 kg C $ha^{-1}yr^{-1}$) in CO, 148,507 Mg of C (262 kg C $ha^{-1}yr^{-1}$) in CO_2, and 1,866 Mg C (3 kg C $ha^{-1}yr^{-1}$) in CH_4 emitted (Figures 3.3 and 3.4). This three year pasture burning scenario represents about a three-fold increase in mean C fluxed annually through CO, CO_2, and CH_4 emissions than from just considering the area in pasture burning once and excluding forest conversion to pasture. In comparison to the 1984-1986 pasture emissions on an annual basis, the mean emissions during the 1986-1992 period were about three times greater. The area of primary forest slash burned was about 60% that of pasture burned in this three year reburning scenario, however, the quantity of pasture burning C emissions were less than 30% of primary forest slash emissions.

Using the two-year burn assumption to estimate pasture emissions during the 1986 to 1992 time period, the contribution of emissions from reburning areas in pasture were clearly more important as a source of emissions from burning in the study area. It is apparent from our calculations that the magnitude of emissions is likely underestimated if there is no consideration of pasture areas reburning. Additionally, by including deforested areas into the pasture area pool for reburning, a supplementary source of emissions contributed to the pasture emissions estimates.

Trace gas estimates for C_2 and C_3 hydrocarbon emissions, from primary and regenerating forest slash and pasture burning during the 1986 to 1992 time period, maintain the trends (primary forest dominating) in the magnitude of the flux of these emissions between land cover types (Figure 3.4). Hydrocarbon emissions of ethene (C_2H_4), ethane (C_2H_6), and propene (C_3H_6) were higher than ethyne (C_2H_2), propyne (C_3H_4), and propane (C_3H_8). These hydrocarbon gases remain at trace levels in comparison to CO, CO_2, and CH_4.

During the entire time period of this study (1984-1992), total C released in the 94,372 ha study area from primary forest slash burning (9,944 ha) was 104,294 Mg C (138 kg C $ha^{-1}yr^{-1}$) in CO, 694,379 Mg C (920 kg C $ha^{-1}yr^{-1}$) in CO_2, and 12,409 Mg C (16 kg C $ha^{-1}yr^{-1}$) in CH_4 (Figures 3.5 and 3.6). Regenerating forest slash burning (827 ha) contributed 3,226 Mg C (4 kg C $ha^{-1}yr^{-1}$) in CO, 23,436 Mg C (31 kg C $ha^{-1}yr^{-1}$) in CO_2, and 376 Mg C (<1 kg C $ha^{-1}yr^{-1}$) in CH_4. For the forests converted to pasture under the two-year burn return interval, total area of pasture burned was 19,008 ha between 1984-1992. The C flux from pasture burning was 34,666 Mg C (46 kg C $ha^{-1}yr^{-1}$) in CO, 238,180 Mg C

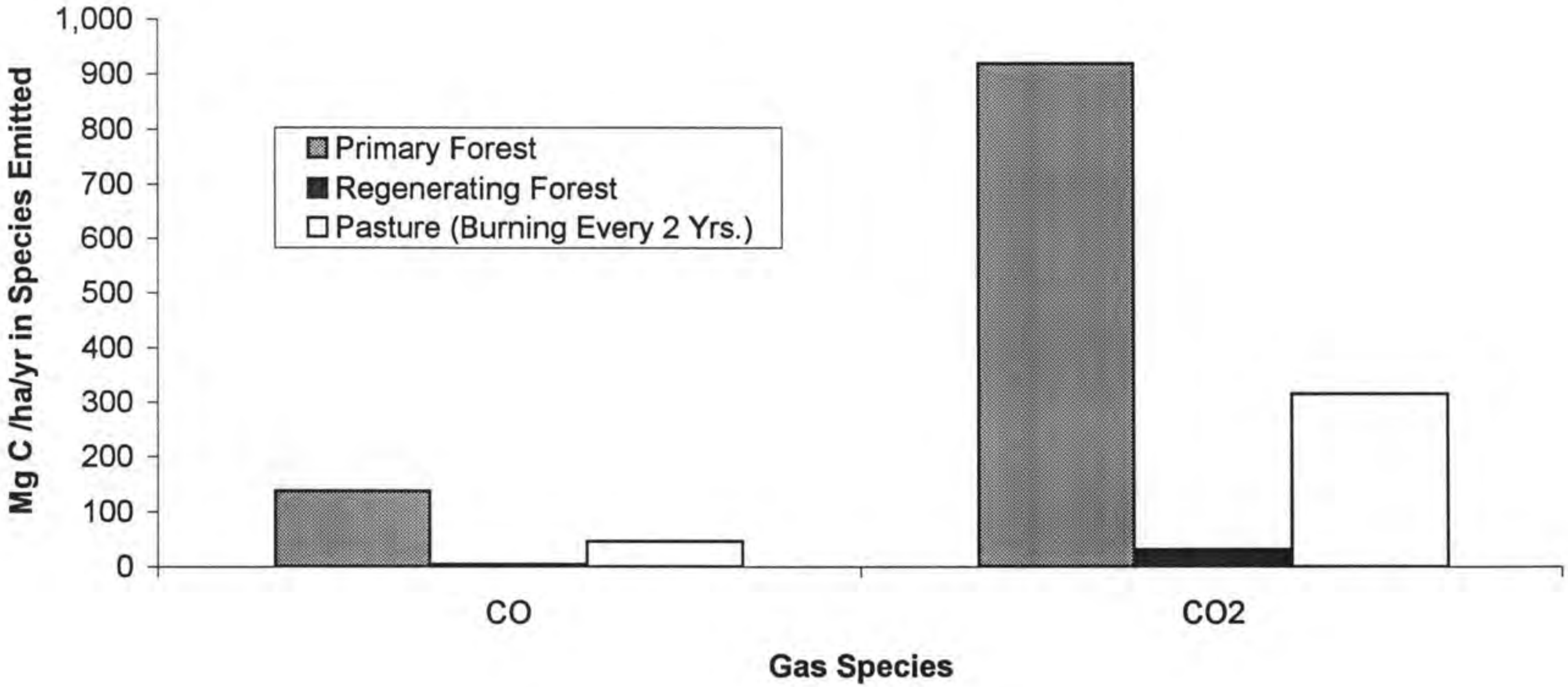

Figure 3.5. Emissions of CO and CO_2 arising from burning primary forest slash, regenerating forest slash, and pastures for the extent of the Jamari, Rondônia study area (94,372 ha) during the study period (1984-1992). Pasture emissions include pastures burned between 1984 and 1986 and modeled pastures reburned on a frequency of two years for the 1986-1992 period.

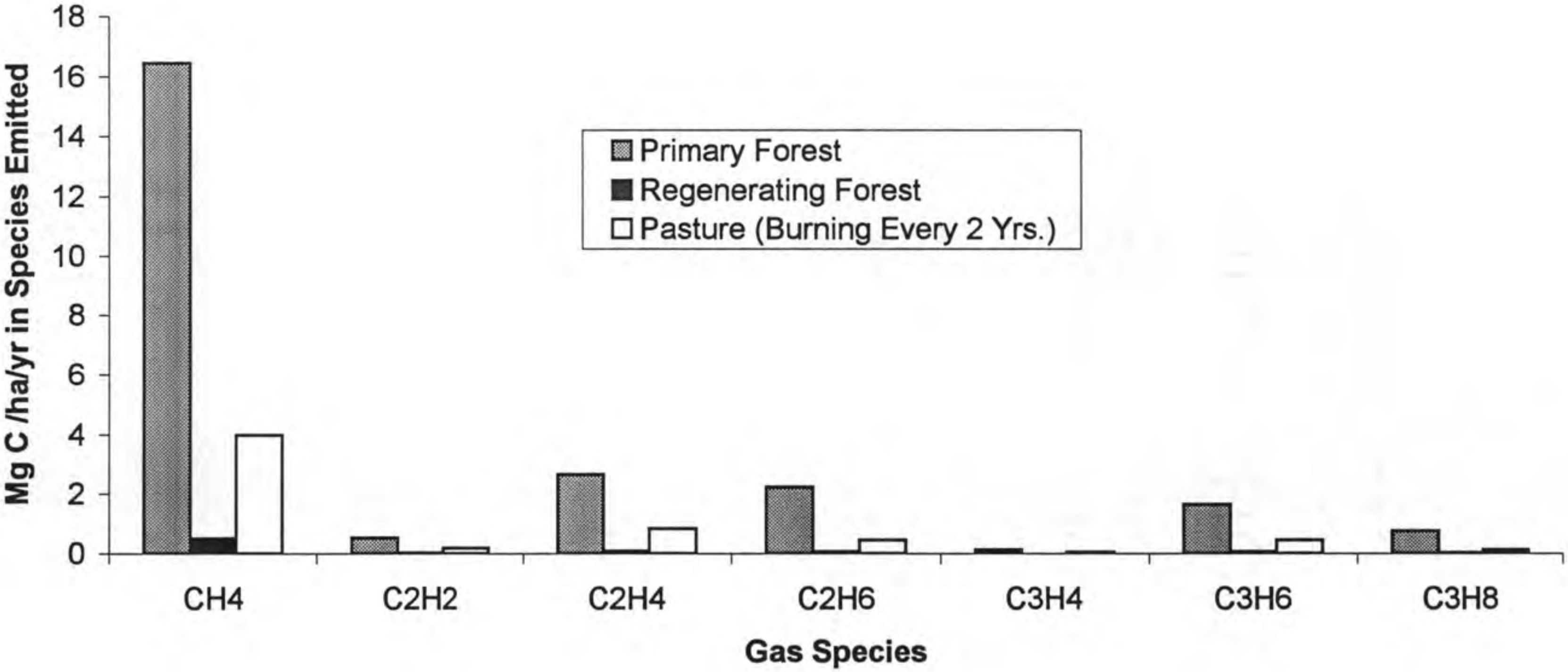

Figure 3.6. Emissions of trace gases arising from burning primary forest slash, regenerating forest slash, and pastures for the extent of the Jamari, Rondônia study area (94,372 ha) during the study period (1984-1992). Pasture emissions include pastures burned between 1984 and 1986 and modeled pastures reburned on a frequency of two years for the 1986-1992 period.

(316 kg C $ha^{-1}yr^{-1}$) in CO_2, and 2,993 Mg C (4 kg C $ha^{-1}yr^{-1}$) in CH_4. The magnitude of the primary forest emissions were three times that of burning emissions from pastures. However, the area of primary forest area burned for the study period was 9,944 ha, whereas 19,008 ha of pasture burned, nearly double the area of primary forest burned.

Discussion

Our estimates of carbon emissions from deforestation and pasture burning are based on an area of early colonization (i.e., low proportion of landscape deforested) in Rondônia. The expansion of land cover in cattle pasture occurred steadily as active deforestation converted forests to pasture during the time period of this study. With increasing area in pasture as well as frequent reburning of pastures, the mounting importance of pastures in the contribution of emissions is revealed in our results. In other areas of Rondônia that are further along in the process of agriculture expansion, pasture burning emissions could potentially rival those from deforestation.

Annual rates of deforestation for Rondônia, excluding areas of forest flooded by hydroelectric dams, were 2,100 km^2 (1978-1988), 1,400 km^2 (1988-1989), 1,700 km^2 (1989-1990), and 1,100 km^2 (1990-1991) [*Fearnside*, 1997]. These deforestation rates were 4%, 8%, 12%, and 10%, respectively of the total for the Legal Amazon's annual deforestation rates. This indicates an increasing trend of annual deforestation on an Amazon-wide scale in the state of Rondônia between 1978 and 1990. Deforestation rates reported for 1990-1991 dropped from 1989-1990 levels perhaps reflecting climatic conditions, local economic conditions, or political changes in the nation's tax structure. As of 1991, 16% (34,200 km^2) of Rondonia's original forest area (215,000 km^2) had been deforested [*Fearnside*, 1997]. In comparison, nearly 11% (421,600 km^2) of the Legal Amazon's orginal forest area (3,996,000 km^2) had been cleared. By using Fearnside's annual deforestation rates and assuming that 1,100 km^2 was also the annual deforestation rate for 1991-1992 we calculated the deforestation area corresponding to the same time period of our study (1984-1992). We estimated that 12,700 km^2 (1,270,000 ha) of primary forest in Rondônia was cleared between 1984 and 1992. Using this deforestation area estimate in the emissions computation, the mass of C emitted was 13 Tg (Tg = 10^{12} g) in CO, 89 Tg in CO_2, and 2 Tg in CH_4. These emissions would be generated from deforestation alone in the state of Rondônia during the period of 1984-1992. To calculate emissions from pasture fires in Rondônia, we assumed that each year during the 1978-1992 time period, the forest area cleared was converted to pasture and half the area in pasture was burned each year. We summed the total area of pasture burned for 1984-1992 (corresponding to our study period). C flux was estimated as 15 Tg in CO, 106 Tg in CO_2, and over 1 Tg in CH_4. Here it is evident that pasture emissions rivaled emissions from deforestation between 1984-1992.

Although emissions from fires of slashed primary forest decreased due to declining deforestation rates in Rondônia between 1978-1992, the quantity of emissions generated from fires to maintain pasture likely increased with the continued expansion of the areal extent of pastures each year. Hence, the conversion of forests to pastures represents a potential long-term source of emissions that has received little attention. Consideration of these anthropogenic sources of emissions is essential for reporting regional and Amazon emissions estimates.

Early estimates of Amazon burning emissions by *Setzer and Pereira* [1991] use a basic method adapted from *Seiler and Crutzen* [1980] to quantify the mass of dry biomass burned. This calculation multiplied area of forest and cerrado (Brazilian savanna) burned by average biomass and combustion factor for each vegetation type. The mass of emissions is assumed to be 45% of the mass of biomass burned. Using emission ratios relative to CO_2 from *Andreae* et al. [1988] the Amazon wide biomass burning emissions for 1987 were estimated as 94 Tg CO, 1,700 Tg CO_2, and 10 Tg CH_4. These estimates were not presented as fluxes of carbon per gas species. As these estimates correspond to the entire Amazon and include savanna burning emissions, we only address comparing Setzer and Pereira's parameters and our input parameters into the respective emissions calculations. Since Setzer and Pereira use the Advanced Very High Resolution Radiometer (AVHRR) at 1.1 km resolution to estimate area burned by land cover type, there could be some error associated with the resolution of the AVHRR in estimating area burned. Better accuracy is achieved with TM, but for burn area estimates for the entire Amazon, AVHRR is more feasible [*Setzer and Pereira*, 1991]. This use of emission ratios based on aircraft sampling over fires was a novel approach. However, aircraft sampling could represent a mixture of emissions from adjacent or nearby fires. Our use of emission factors from ground-based tower measurements likely eliminates most of this error associated with mixing of emissions at higher altitudes. In addition, delineating the contribution of flaming and smoldering combustion to emissions in our approach may provide more accurate emissions estimates.

The shortcomings of our research are that there are additional sources of emissions that arise from the land use/land cover change areas besides those directly from combustion not estimated by our techniques. For example, decomposition of uncombusted biomass from the original forest and regrowth found in pastures and regenerating forests is included in Amazon-wide deforestation emissions of *Fearnside* [1997]. *Fearnside* [1997] suggested that microbes and termites decomposed most of the remaining biomass in the first 10 years following deforestation. During this period, 42% of the aboveground biomass carbon was released through combustion and 56% through decomposition. The remaining biomass carbon was found in charcoal remains (< 2% of prefire carbon pools) from burning slash and result in long-term carbon pools in the soil. Additionally, reburning of coarse wood in pasture sites increased the on-site charcoal carbon pools. Not directly related to biomass burning, but associated with pastures is the production of CH_4 by cattle. Globally, ruminants produce 15-20% of the total CH_4 emissions and; therefore, are an important source of CH_4 emissions

[*Leng*, 1993]. However, CH_4 production by ruminants is not addressed in this research.

In addition to aboveground carbon, there are carbon emissions estimates associated with belowground carbon [*Fearnside*, 1997; *Fujisaka* et al., 1998]. We did not include belowground carbon emissions from decomposition and processes influenced by burning and heating of soil. *Fearnside* [1997] assumed that all carbon released from soils is CO_2 and that the greatest release occurred near the surface (i.e, top 20 cm of soil). An additional belowground carbon source for CO_2 emissions included in *Fearnside's* [1997] emissions is belowground decay. Fearnside's approach to estimating emissions for the Legal Amazon is based on the carbon stock per hectare and the partitioning of carbon released as the unit area is transformed from forest to the resultant land use of pasture over a period of 10 years. Fearnside termed these emissions as 'net committed emissions' where sources of carbon emissions include, initial forest burn, pasture reburning, termites and decomposition (aboveground and belowground), soil carbon, and cattle. Considering only burning emissions, Fearnside estimated net committed emissions of 26 Mg CO, 283 Mg CO_2, and 1 Mg CH_4 in the 13,800 km^2 area of forest cleared in the Legal Amazon in 1990 based on pasture land use for the following 10 years. Although Fearnside's approach and scale of emissions estimates is different and not appropriate for comparison with our methods, the discussion of the inputs into Fearnside's calculation is important to address. For example, in quantifying burning emissions Fearnside used TAGB, burning efficiency, releases by reburning, and flaming and smoldering burn releases (for trace gases). Some of the ground-based measurements that we used were also used by Fearnside. In contrast, we include flaming and smoldering burn release for all gases estimated and additionally our methods use emissions factors.

Fujisaka et al. [1998] estimated carbon emissions based on aboveground (trees, understory, and charcoal) and belowground (roots and soil) carbon stock losses associated with deforestation in the area of Theobroma, Rondônia. These carbon emissions are calculated as the carbon stock of the original forest area subtracted by the carbon stock of the area estimates of replacement land use vegetation (i.e., pasture, fallow, and annual crops). In the 216,500 ha study area, 93,211 ha (43% of the study area) was deforested by 1993 with carbon emission estimate of 14 Tg over a 20-year period (1973-1993). On an annual basis for the Theobroma study area, carbon emissions associated with deforestation and land use were approximately 3 Mg C $ha^{-1}yr^{-1}$. For our 94,372 ha study area, of which only 14% was deforested by 1992, our total carbon emissions estimate was nearly 2 Mg C $ha^{-1}yr^{-1}$ and this estimate includes reburning of pasture areas. The forest and fallow aboveground carbon stocks used in the Theobroma study were within the range for our primary and regenerating forest measurements, but the aboveground pasture carbon stocks for our estimates were five-fold greater than those used in the Theobroma study. In the Theobroma study, belowground carbon stocks were 16-45% of the total forest, annual crop, and fallow crop carbon stocks. However, belowground carbon stocks in pasture were 69% of the total pasture carbon stock. The differences between the Theobroma study and ours is likely due to the amount of area deforested in the study area and not from the estimates of carbon stocks. The total of aboveground and belowground carbon stocks in the Theobroma study were within the range of, or close to our estimates of aboveground carbon stocks by land cover type. Theobroma is an area subjected to wide-spread land cover/land use change for a longer time period. Also, the Theobroma study assumed that the differences between carbon stocks of forest and resultant land uses were carbon emissions.

The methods presented in this research could be applied to other states in Amazonia to scale-up local ground-based measurements to estimate state biomass burning emissions and ultimately improve Amazon Basin biomass burning emissions associated with land use. To accomplish this, appropriate ground-based data on emission factors, TAGB, and combustion factors would need to be integrated with regional deforestation and land clearing estimates. We assumed that for this study, the ground-based estimates of TAGB, combustion factors, and emission factors were representative of the primary forests, regenerating forests, and pastures of the study area. Less important are the estimates of regenerating forests since this area represents only a small part (<3%) of the study area. However, this may not be the case in other areas of the Amazon or other tropical forests of the world where agricultural areas are regenerating as secondary forest.

As measurements of TAGB and combustion factors will likely change with age of pasture and age of regenerating forest, for example, we assumed the different ages of pastures and second-growth and third-growth forests sampled in the ground-based measurements captured these differences. This could be an area of further investigation.

We found that area estimates of deforestation and land clearing are better captured in a temporal series of satellite data at a frequency to adequately capture change rather than from single-date analysis (Chapter 2). For Rondônia and other areas of the Amazon where pasture burning is persistent, frequent intervals of data selection could improve estimates. In other areas of the Amazon where forest conversion and pasture burning is less frequent, the time interval for data selection might be on the order of every four to six years or longer. In areas where shifting cultivation and/or cattle ranching is intense, a well documented knowledge of the land use and burning practices would decrease uncertainties in burning predictions when adequate satellite or aircraft data is unavailable. Agricultural agencies of the region and landowner interviews could provide the land use history needed to make predictions land use practices that influence C flux (e.g., burning scenarios). Addressing the contribution of biomass burning emissions associated with agricultural practices at an appropriate scale could improve the accuracy of the contribution to changes in the atmospheric C balance.

Conclusion

For our study area describing a period of early colonization, the greatest emissions sources were attributed to those arising from fires in primary forest slash, but the frequency of pasture burning also produced a considerable flux of emissions. Using 1978-1991 deforestation estimates of *Fearnside* [1997], we found that in cattle ranching dominated land use systems following deforestation, pasture burning emissions rival deforestation burning emissions. Following colonization, agricultural expansion in Rondônia resulted in forests primarily converted to pastures. Emissions from frequent pasture reburning may be as significant as those arising from forest slash fires. Also with areas in pasture increasing, forested area would become increasingly fragmented. Forest edges and fragments become more susceptible to surface fires escaping from pasture and slash fires contributing to additional elemental pool losses and burning emissions. Increased disturbance elevates losses in carbon and nutrient pools, decreases the capacity of the ecosystem to function as a carbon sink, and in time renders the system unproductive.

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Chapter 4

Landscape-Scale Carbon and Nutrient Pool Losses Associated with Biomass Burning from 1984-1992 for Jamari, Rondônia, Brazil

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Abstract

Rates of land use/land cover change, which is primarily the conversion of tropical evergreen forest to cattle pasture, remain high in the Brazilian Amazon. As a result, terrestrial carbon and nutrient pools continue to be depleted. Original primary forest pools become sources of biomass burning emissions during land use and for management of pastures. We applied site estimates of C, N, and S pools and resultant losses associated with burning of primary and regenerating forests and pastures to a 94,372 ha study area in northern Rondônia for the time period of 1984 to 1992. Scaling of site measurements to the landscape level allowed quantification of losses from elemental pools in an area experiencing early colonization during this time period. In 1984, the carbon pool for the study area was 163,297 kg ha^{-1}. In a period of eight years (1992), 5% of the C pool or 8,936 kg C ha^{-1} had been lost from burning of primary forest. Burning and reburning of pastures and including forested areas converted to pasture on a two year burning frequency between 1984 and 1992, contributed to additional C losses of 2,438 kg ha^{-1} or 1% of the total C pool. The N pool comprised 1,971 kg ha^{-1} and lost 129 kg N ha^{-1} or 7% of the N pool from burning of primary forest slash during the eight-year study period. Cumulative pasture burning on a two-year frequency resulted in a loss of 36 kg N ha^{-1} or 2% of the N pool. The 1984 S pool was 199 kg S ha^{-1}, but by 1992, 10 kg S ha^{-1} was lost by burning of primary forest slash. This accounts for a 5% loss of the S pool. Cumulative pasture fires (two year burning scenario) resulted in 3 kg S ha^{-1}

or 1% loss from the S pool. The area in regenerating forest that was burned during the study period contributed to less than a 1% loss from the 1984 C, N, and S pools. As forested areas were slashed and burned during the study period, these sites continued to contribute to additional elemental losses as new pasture areas were maintained by a two to three year burning frequency. By 1992, elemental losses from pasture burning were seven times greater than in 1984 due to increasing area in pasture from forest conversion. We found that emissions in sites initially are relatively low, but increase as colonization and deforestation rates increase.

Introduction

Anthropogenic burning associated with deforestation and agricultural practices in the Amazon reduces terrestrial carbon and nutrient pools [*Guild et al.*, 1998; *Hughes et al.*, in press; *Kauffman et al.*, 1995, 1998]. Traditional agricultural practices in the Amazon utilize burning to clear cut primary forest slash and regenerating forest slash in fallow shifting agriculture sites. Remnant woody debris not consumed in slash fires remain on site and are subject to decomposition and reburning in shifting cultivation and pasture fires [*Fearside*, 1997; *Graça*, 1999]. Additionally, pastures are burned every 2 to 3 years to eliminate disease, control native plants, and to stimulate growth of fresh palatable forage grass for cattle [*Fearnside*, 1997; *Uhl et al.*, 1988]. Accidental fires are also common whereby fires lit on adjacent sites can burn into pastures and even into disturbed forest edges [*Kauffman et al.*, 1998; *Uhl and Buschbacher*, 1985]. Burning typically occurs during the end of the dry season (July to October). In Rondônia, this period of burning commonly occurs between August and October.

Although regional and landscape scale changes in deforestation and land cover change in the Amazon have received much attention by researchers, there have been few studies of carbon and nutrient dynamics associated with anthropogenic practices; namely burning practices [*Fearnside*, 1997; *Graça*, 1999;

Guild et al., 1998; *Hughes et al.*, in press; *Kauffman et al.*, 1995, 1997, 1998; *Moraes*, 1998; *Uhl and Kauffman*, 1990; *Uhl et al.*, 1988]. *Houghton* [1991] estimated aboveground C stocks for moist forests of tropical America as 176 Mg ha^{-1} based on destructive sampling of biomass and 82 Mg ha^{-1} based on wood volumes. In Amazonas, *Fearnside et al.* [1993] estimated aboveground C stocks for a primary forest site were 130 Mg ha^{-1} and 94 Mg ha^{-1} following fire with a presumed release of 36 Mg ha^{-1}. Based upon a series of ground-based measurements in Rondônia, *Guild et al.* [1998] and *Kauffman et al.* [1995] reported primary forest aboveground C pools ranged from 142-195 Mg ha^{-1}. Following deforestation and burning of forest slash, C losses ranged from 58-102 Mg ha^{-1}. Also in Rondônia, aboveground C pools in regenerating forests range from 34-90 Mg ha^{-1} with C losses of 20-47 Mg ha^{-1} following cutting and burning of slash [*Hughes et al.*, submitted]. Rondônian pasture sites have aboveground C pools ranging from 34-59 Mg ha^{-1} and exhibit C losses of 9-16 Mg ha^{-1} from fire to maintain pastures [*Guild et al.*, 1998; *Kauffman et al.*, 1998]. [*Graca et al.*, 1999] report a primary forest prefire C stock of 141 Mg ha^{-1} for a study area in Ariquemes, Rondônia. Following fire, the C stock decreased to 92 Mg ha^{-1}. The importance of dedicating research to C and nutrient pool losses associated biomass burning is that these losses become substantial atmospheric sources of greenhouse gases, nonradiatively active trace gases, and aerosols, particularly in the troposphere [*Allen and Miguel*, 1995; *Andreae*, 1988, 1991, 1993; *Crutzen and Andreae*, 1990; *Crutzen et al.*, 1985; *Fearnside*, 1997; *Houghton*, 1990; *Ward et al.*, 1992].

The objectives of this research were to: 1) scale site estimates of C, N, and S pools and losses to regional models of land use/land cover change to estimate pools and losses at a landscape scale for the Jamari, Rondônia study area between 1984 and 1992; 2) quantify cumulative elemental pool losses from burning forest slash and pastures; and 3) utilize the results from this approach to validate C losses derived from modeling C emissions from biomass combustion in the study area for the same time period (Chapter 3).

Methods

Study Area

We examined land use/land cover change from fire from 1984-1992 in a 94,372 ha area of primary forest in Rondônia, Brazil. The study area is located along the BR-364 highway in an area of recent colonization resulting in active deforestation during the period of our study. The BR-364 highway, which bisects the state, was opened in 1960 and was paved in 1984. This highway provided a route for colonists to migrate and claim land within the state and provided an overland transportation route to the Atlantic Coast [*Browder and Godfrey*, 1997]. Areas along the BR-364 highway have been subject to intense deforestation for cultivation, cattle pastures, timber exploitation, and mining.

We chose to conduct our study in this region because of its relatively recent colonization and deforestation activity at the onset of our analysis (since the early 1980's) and this allowed quantification of elemental losses that occur when tropical forest landscapes are colonized. In addition, the presence of relevant ground-based data on prefire and postfire biomass, elemental pool dynamics, and fire chemistry were available [*Guild et al.*, 1998; *Hughes et al.*, in press; *Kauffman et al.*, 1995, 1998]. Other related research in the region is occurring south of Jamari from Ariquemes to Vilhena along BR-364 highway where more long-term deforestation, agriculture, and mining have occurred [*Fujisaka et al.*, 1998; *Moraes et al.*, 1998; *Neill et al.*, 1997; *Tucker et al.*, 1984]. Our study area and study period results from a landscape of early stages of disturbance may provide some interesting comparisons with results of elemental pools and losses from the landscapes of intense disturbance.

The 94,372 ha study area is centered at approximately 9°11' S and 63°10' W and about 100 km southeast of the state's capital, Pôrto Velho. The smaller town of Jamari is located along the Jamari River, a blackwater tributary of the Madeira River. The study area consists of both small landholdings of subsistence farms and large ranches. It is adjacent to the Jamari National Forest, which contains the Santa Bárbara Tin Mine.

The primary forest type of Rondônia and the study region is submontane open forest consisting of overstory broad-leaved canopy and subcanopy with an abundance of palms and vines [*Cummings*, 1998; *Departmento Nacional de Produção Mineral*, 1978; *Instituto Brasileiro de Desenvolvimento Florestal (IBDF) and Instituto Brasileiro de Geographia e Estatística (IBGE)*, 1993]. Soil types include red-yellow podzolic latosols and red-yellow latosols [*Neill et al.*, 1997].

Climatological data come from Pôrto Velho, Rondônia about 100 km north of the region. Mean annual precipitation is 2,354 mm [*Departmento Nacional de Meteorologia, Brasil*, 1992]. During the dry season, between June and September, mean precipitation is typically <100 mm per month. Dry season mean temperature is ~25°C ranging from a minimum of ~21°C to a maximum of 31°C with a mean relative humidity of 85%.

Carbon and Nutrient Pools

Carbon and nutrient pool data were summarized from biomass and nutrient loss experiments conducted in 1992, 1993, and 1995 in the study area [*Guild et al.*, 1998; *Hughes et al.*, in press; *Kauffman et al.*, 1994, 1995, 1998]. In brief, prefire and postfire data were collected for four primary forest slash sites, four regenerating forest sites, and three pasture sites. Just prior to burning in each of these sites, samples of each fuel component were collected for nutrient analysis.

Following fire, each of the sites were sampled for postfire biomass and ash was collected for nutrient analysis. For all sites, each fuel component category for prefire and postfire was multiplied by the corresponding nutrient concentrations for estimating total aboveground mass of C, N, and S. Site losses were calculated as the difference of prefire and postfire (including ash) nutrient mass.

Mean elemental pools and losses per hectare for each land cover type were calculated using all sites from the 1992-1995 studies (Table 4.1). In the Jamari area, burning of primary forest slash reduced a site by a mean amount of 85 Mg C ha^{-1}, 1221 kg N ha^{-1}, and 99 kg S ha^{-1} [*Guild et al.*, 1998; *Kauffman et al.*, 1994, 1995]. Mean losses associated with burning regenerating forest slash were 32 Mg C ha-1, 613 kg N ha^{-1}, and 38 kg S ha^{-1} [*Hughes et al.*, in press]. Pasture fires result in mean losses of 12 Mg C ha^{-1}, 178 kg N ha^{-1}, and 13 kg S ha^{-1} [*Guild et al.*, 1998; *Kauffman et al.*, 1994, 1998].

Quantifying C, N, and S Pool Losses from the Study Area

In general, the pattern of land use in the region and the corresponding C and nutrient losses follows a path that begins with deforestation of a site with the ultimate fate of the site being conversion to pasture [*Fearnside*, 1997; *Hecht*, 1993] (Figure 4.1). The greatest loss or pulse of C and nutrients from a given site is from the initial conversion of primary forest to cultivation or pasture [*Guild et al.*, 1998; *Kauffman et al.*, 1995]. The relatively high biomass (290-399 Mg ha^{-1}) and relatively high combustion factors (38-57%) contribute to greater losses in primary forest than in other land cover types of this region. Relative to primary forests, intermediate losses are associated with burning regenerating forest slash in a shifting cultivation land use scenario for either returning to cultivation or conversion to pasture [*Hughes et al.*, in press]. Because relatively low quantities of biomass are consumed, pasture burning results in the lowest losses of C and nutrients per hectare [*Guild et al.*, 1998; *Kauffman et al.*, 1994, 1998]. This is

Table 4.1. The range and mean carbon and nutrient pools and losses for selected land cover types in the Jamari, Rondônia study area (1992-1995).

Component	C Mg ha^{-1}	N kg ha^{-1}	S kg ha^{-1}	Reference
Primary forest slash				Guild et al. [1998], Kauffman et al. [1994, 1995]
Range in pool	142-195	1,825-2,427	178-252	
Mean pool (n=4)	172±10	2,075±114	210±14	
Range in losses	58-102	1,019-1,605	87-122	
Mean loss (n=4)	85±11	1,221±133	99±8	
Regenerating forest slash				Hughes et al. [in press]
Range in pool	34-90	720-1012	48-97	
Mean pool (n=5)	57±8	782±41	65±9	
Range in losses	20-47	479-709	29-44	
Mean loss (n=4)	32±6	613±51	38±3	
Pasture				Guild et al. [1998], Kauffman et al. [1994, 1998]
Range in pool	34-59	295-661	39-78	
Mean pool (n=3)	43±7	443±96	53±11	
Range in losses	9-16	88-240	5-19	
Mean loss (n=3)	12±2	178±46	13±4	

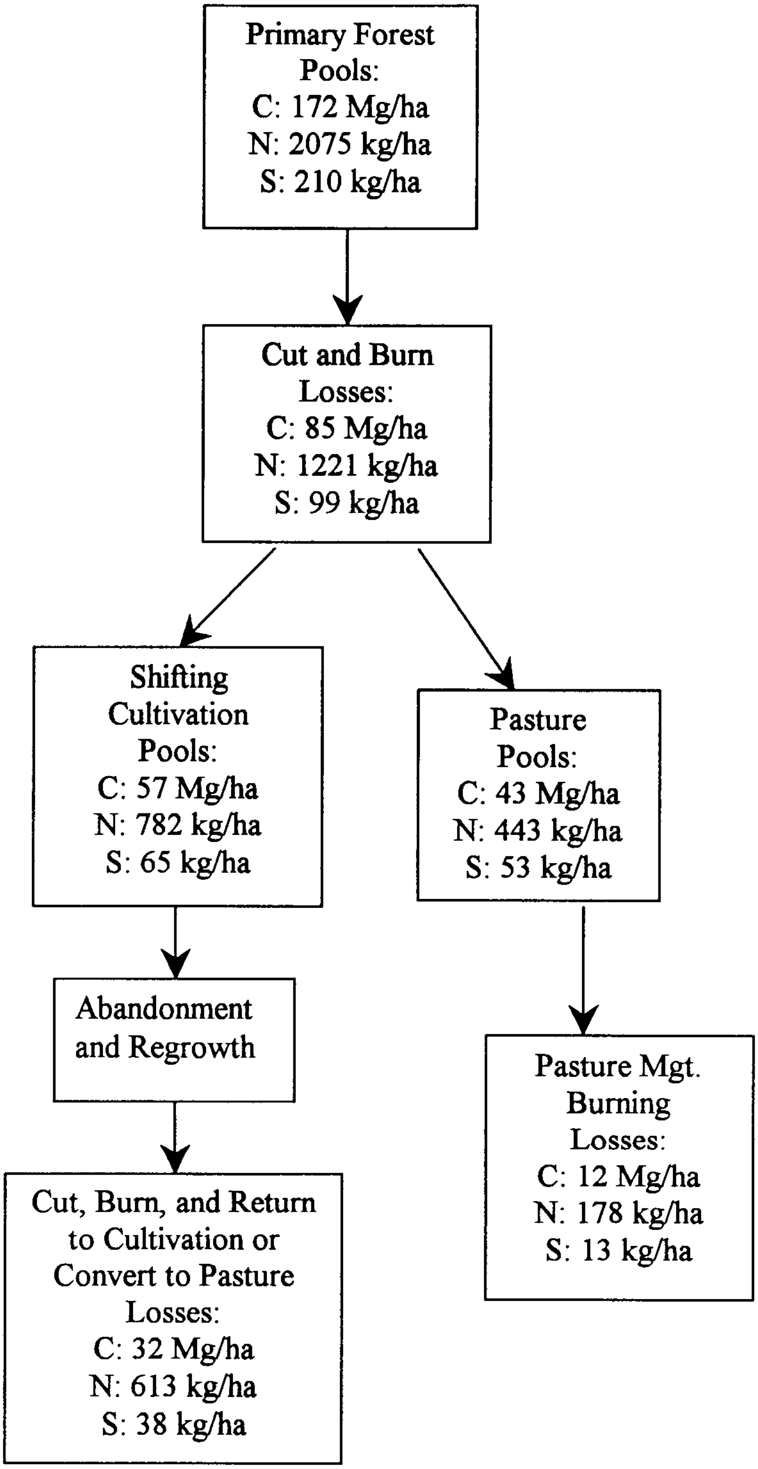

Figure 4.1. Pools of C, N, and S and the losses by burning in primary forest, regenerating forest, and pasture in the Jamari, Rondônia study area. Data are based on studies from *Guild et al.* [1998], *Hughes et al.* [in press], and *Kauffman et al.* [1995, 1998].

attributed to the greatest proportion (67-87%) of biomass in pastures being contained in the remnant woody debris from the original forest [*Guild et al.*, 1998; *Kauffman et al.*, 1994, 1998]. Additionally, the fine fuels are more readily consumed by fire than the remnant wood debris. In general, during the average lifetime of a pasture (~10 years), burning woody debris in pastures continues to contribute to substantial elemental losses with additional C losses associated decomposition of the woody debris [*Fearnside*, 1997].

In this study we scaled C and nutrient losses, resulting from burning primary and regenerating forest slash and fire used to maintain pastures, to the landscape scale of the Jamari, Rondônia study area (94,372 ha) for the time period of 1984-1992. The land cover type and change in land cover type, which indicates burning events during this time period, is based on 1984, 1986, and 1992 Landsat Thematic Mapper (TM) data (Chapter 2). Due to the six-year time gap in TM data between 1986 and 1992, equations for predicting burning events associated with deforestation and pasture management practices were implemented during this period. These predictions assume: 1) deforestation of both primary and regenerating forest occurred at a linear rate during the study period and 2) pastures were burned at a frequency of every two years for pasture management or from accidental burnings. Equations to predict losses were described in Chapter 3 for forecasting timing of burning for each land cover type and land use scenario during the six-year period.

By integrating the area estimates of deforestation slash burning and pasture burning for 1984-1992 from the Landsat TM land cover change map results (Chapter 3, Table 3.2) with the predictions of subsequent burning of pastures following deforestation (Chapter 3), we estimate that 9,944 ha of primary forest slash, 829 ha of regenerating forest slash, and 19,008 ha of pasture were burned (Figure 4.2, Table 4.2). There are three assumptions in these estimates: 1) primary and regenerating forests were all converted to pasture following burning; 2) areas that remained in a state of clearing for the time frame of the study were assumed to

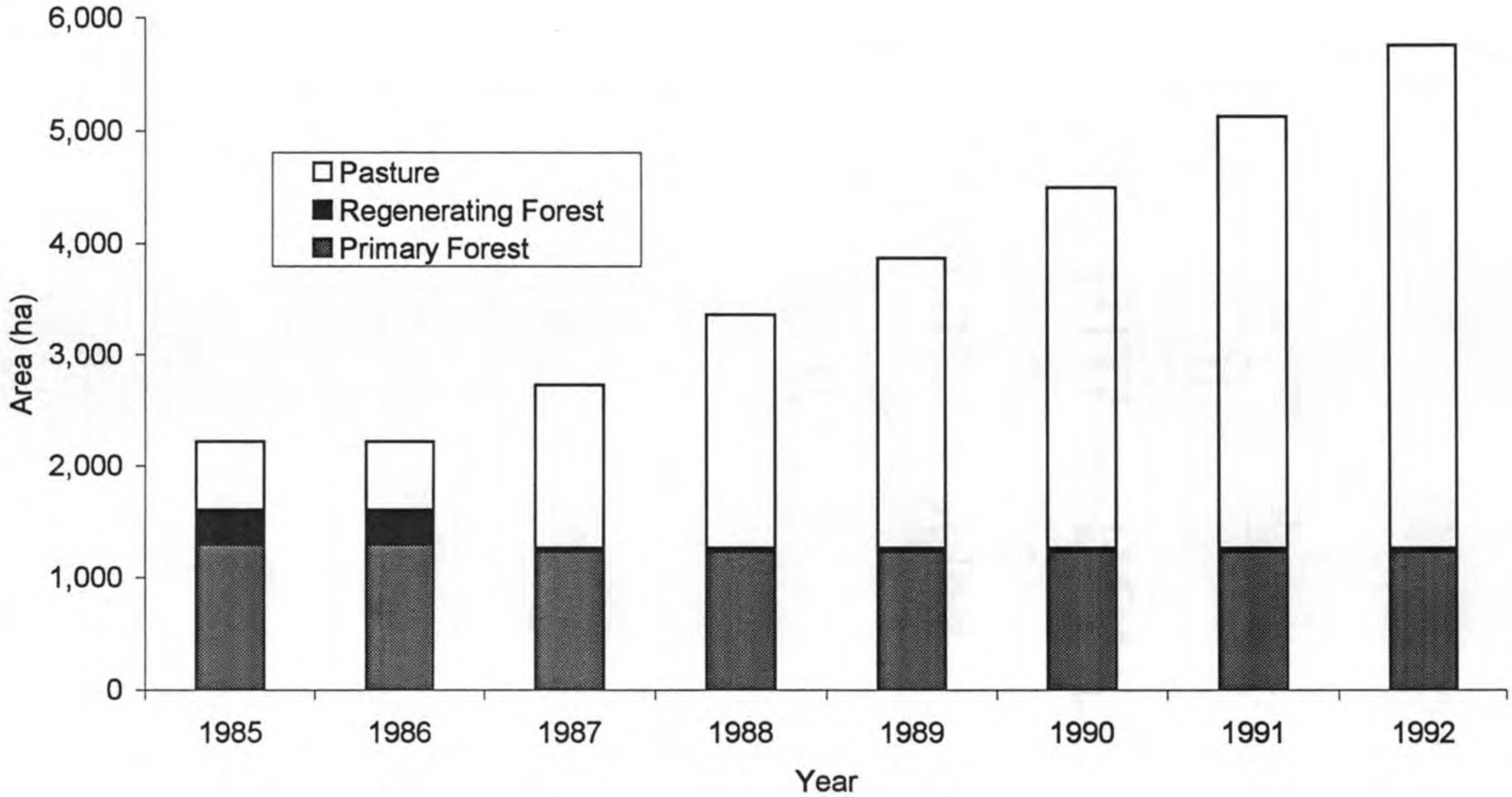

Figure 4.2. Annual anthropogenic biomass burning of primary forest slash, regenerating forest slash, and pastures in the Jamari, Rondônia, Brazil study area (1984-1992).

Table 4.2. Primary and regenerating forest areas slashed and burned and cumulative area of pasture burned in the Jamari, Rondônia study area (94,372 ha) in Brazil (1984-1992).

Land Cover Type	Area (ha)	Percent of Study Area
1984-1986		
Primary Forest Slash and Burn	2,592	3
Regenerating Forest Slash and Burn	615	<1
Pasture Burned	1,231	1
1986-1992		
Primary Forest Slash and Burn	7,352	8
Regenerating Forest Slash and Burn	213	<1
Cumulative Area of Pasture Burned		
(2 yr. burning frequency)	17,777	n/a
(3 yr. burning frequency)	11,851	n/a
Total 1984-1992		
Primary Forest Slash and Burn	9,944	11
Regenerating Forest Slash and Burn	829	<1
Cumulative Area of Pasture Burned		
(2 yr. burning frequency)	19,008	n/a
(3 yr. burning frequency)	13,082	n/a

n/a: Not applicable.

be pastures; and 3) the pasture burning sequence follows a frequency of every two years. The first assumption that forests are converted to pasture is very likely since conversion of forest to pasture is the dominant land use of the region [*Fearnside*, 1991, 1997; *Guild et al.*, 1998; *Kauffman et al.*, 1998]. In addition based on analysis of the TM data, area in pasture (area in clearing) was greater than area in shifting cultivation (regrowth) in the study area (Table 2.1). The second assumption that areas showing no signs of regrowth, where regrowth is indicative of shifting cultivation, are not difficult to interpret because these areas are quite evident in analysis of satellite imagery selected during the dry season. Areas of regrowth are distinct spectrally from dry pasture grasses during the dry season (Chapter 2). The third assumption that pastures are burned every two years, either purposely or accidentally, is based on interviews with ranchers, observations of the authors, and is also typical of other cattle ranching areas of the Amazon [*Fearnside*, 1991; 1997; *Kauffman et al.*, 1998; *Uhl et al.*, 1988]. Fearnside [*1991*; *1997*] reported that cattle pastures in the Amazon are burned every two to three years to maintain pasture grass growth and to hasten growth of woody vegetation that is unpalatable to cattle. In addition, *Uhl et al.* [1988] reported that 70% of the pastures in the Paragominas region in the state of Para, Brazil were burned every one to three years to control native plants from competing with the pasture grasses. To quantify the pasture scenario of burning every two years, we calculated the sum of the area in pasture each year and divided the total pasture area in half. The total area in pasture each year included the previous year's area in pasture and forest area cleared (conversion to pasture). Half of this total area in pasture burned each year.

In a second approach, we modified our assumptions to where pastures were burned every three years. This burning scenario gave total area estimates of 11,851 ha of pastures burned during the time period of 1986-1992. This estimate was calculated similarly to the two year burning scenario, however for this calculation, a third of the total area in pasture was burned each year. Adding this to the estimate of area burned as pasture for 1984-1986 yields a total of 13,082 ha for 1984-1992. This estimate remains consistent with the other two assumptions for this pasture burning scenario.

We estimated C, N, and S pools for the primary forest area for the study area prior to any disturbance. Essentially, the entire study area excluding the Jamari River was assumed to be primary forest prior to any disturbance. This pre-disturbance primary forest area encompassed 93,277 ha. C, N, and S pool mass per hectare estimates were multiplied by the area of pre-disturbance primary forest to quantify pre-disturbance pools. Similarly, C, N, and S pools were estimated for each area of primary forest, regenerating forest, and pasture for 1984. Based on the area estimates of forest clearing and pasture area burned between 1984 and 1992, corresponding C, N, and S mass estimates of loss per hectare were multiplied. C, N, and S losses were quantified for the period of pre-disturbance to 1984 and from 1984 and 1992.

Results

In the absence of deforestation (prior to 1984), the C, N, and S pools of the 94,372 ha study area were estimated to be 170,004 kg C ha^{-1}, 2,051 kg N ha^{-1}, and 208 kg S ha^{-1}. We estimated that elemental pools for the entire 94,372 ha study area at the beginning of the time frame for this study in 1984, were 161,369 kg C ha^{-1} in primary forest comprising an area of 88,539 ha (Table 4.3). For the 2,264 ha of regenerating forests, the C pool was 1,368 kg C ha^{-1}. Finally, there were 1,231 ha in pasture representing a C pool of 561 kg C ha^{-1}. Therefore by 1984, a total of 6,707 kg of C ha^{-1} was lost from the pre-disturbance pools (Table 4.4). This represents a 4% loss of C by 1984 from the pre-disturbance C pool.

Table 4.3. Mean carbon and nutrient pools as of 1984 and losses as of 1992 for selected land cover types in the Jamari, Rondônia study area (94,372 ha). Losses in forest cover types include deforestation burn and subsequent burning frequency of two or three years under a pasture burning sequence.

Component	1984 C Pool kg ha^{-1}	1992 C Loss kg ha^{-1}	1984 N Pool kg ha^{-1}	1992 N Loss kg ha^{-1}	1984 S Pool kg ha^{-1}	1992 S Loss kg ha^{-1}
Primary forest	161,369	8,936	1,947	129	1197	10
2 yr. pasture burning		1,548		23		2
3 yr. pasture burning		1,032		15		1
Regenerating forest	1,368	277	19	5	2	<1
2 yr. pasture burning		271		4		<1
3 yr. pasture burning		181		3		<1
Pasture	561		6		<1	
2 yr. pasture burning		618		9		<1
3 yr. pasture burning		465		7		<1
Total	163,297		1,971		199	
Forests		9,212		134		11
2 yr. pasture burning		2,438		36		3
3 yr. pasture burning		1,677		25		2

Table 4.4. Pre-disturbance pools of C, N, and S and subsequent losses associated with forest slash and pasture burning for the Jamari, Rondônia study area (94,372 ha) during the study period (1984-1992).

Component	C kg ha^{-1}	N kg ha^{-1}	S kg ha^{-1}
Pre-disturbance pool	170,004	2,051	208
Loss by 1984	6,707	80	8
1984-1992 loss	11,650	170	14
Total loss since pre-disturbance	18,357	250	22

Dynamics of C Pools and Losses

During the study period (1984-1992), 9,444 ha of primary forest were burned (Table 4.2). This represents nearly 11% of the study area and over 11% of the area in primary forest as of 1984. C losses from burning this area of primary forest were 8,936 kg C ha^{-1} (Figure 4.3, Table 4.3). This loss represents 5% of the total C pool present in 1984. Regenerating forest slash fires comprised an area of 829 ha and contributed to losses of 277 kg of C ha^{-1} from the study area. Following clearing, primary forest sites were converted to pasture and subsequent pasture burning (12,075 ha) on these sites contributed to additional losses of 1,548 kg C ha^{-1} during the study period. Additional losses of 271 kg C ha^{-1} from subsequent pasture burning (2,112 ha) were estimated from regenerating forests converted to pasture. These losses assume a two-year pasture burning frequency following forest conversion. On a three-year pasture burning frequency, losses were estimated as 1,032 kg C ha^{-1} from primary forest areas that were converted to pasture (8,050 ha). Regenerating forest areas converted to pasture had additional losses of 181 kg C ha^{-1} from cumulative pasture area burning (1,408 ha).

The total area of pasture burned between 1984 and 1986 was 1,231 ha (Table 4.2). Additionally, the pasture area burned (17,777 ha) between 1986 and 1992 (50% of total area in pasture burned annually) resulted in the cumulative area of 19,008 ha of pastures burned from 1984-1992. Frequent burning of area in pasture on a two-year cycle for the 1984-1992 period resulted in a C loss of 2,438 kg ha^{-1} (Figure 4.3, Table 4.3). This C loss from pasture burning for the time period of the study represents a 1% loss of the 1984 C pool for the study area. Using the three-year burning cycle, the cumulative pasture area burned between 1984 and 1992 was 13,082 ha and resulted in a loss of 1,677 kg C ha^{-1}.

Since much greater biomass and a higher combustion factor (Table 3.2) are associated with primary forest, C losses originating from the burning of primary forest slash were higher than pasture. Although the cumulative area of pasture burning for 1984-1992 is approximately two times higher than for primary forest, C

Dynamics of C Pools and Losses

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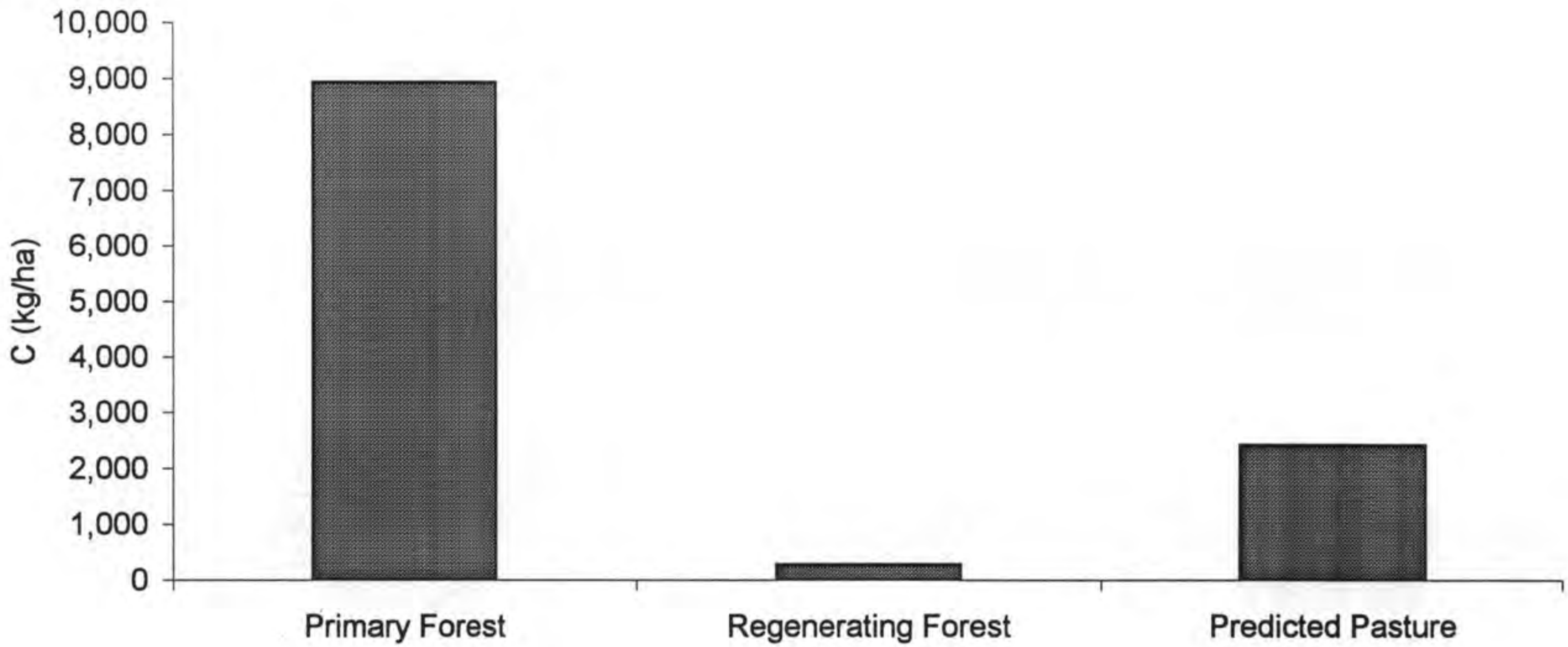

Figure 4.3. Cumulative carbon loss associated with burning of primary forest, regenerating forest, and pastures in the Jamari, Rondônia, Brazil study area (1984-1992). The predicted pasture area includes subsequent burning of the 1984 area in pasture and additionally the primary and regenerating forest areas assumed to be converted to pasture following slashing and burning. Pasture areas were assumed to be burned every two years.

loss from primary forest slash burning was four times greater than from pasture burning (Figure 4.3, Table 4.3). C loss associated with regenerating forest slash (829 ha) was much lower than for pasture (19,008 ha). The cumulative area burned for pasture was 23 times that of regenerating forest slash (Table 4.2, Figure 4.2). While we predicted linear rates of deforestation for 1987-1992 and we assumed these areas were deforested for conversion to pasture, pasture area burned increased annually along with resultant C losses (Figure 4.2, Table 4.5). Annual C loss associated with pasture burning was 79 kg ha^{-1} between 1984-1986 and rose to 577 kg ha^{-1} by 1992. The C loss associated with pasture burning was over seven times greater in 1992 than in 1985. The cumulative C loss from forest conversion and pasture burning in the study area between 1984 and 1992 was 11,650 kg ha^{-1} or a rate of <1% per year. This eight-year study period represents a 7% loss from the pre-disturbance C pool. The cumulative C loss as of 1992 was 18,357 kg ha^{-1}, an 11% loss from pre-disturbance C pools.

Land cover change of primary forest to pasture (1984-1992), assuming a two year pasture burning cycle after conversion, resulted in a 58% loss from the deforested primary forest's prefire C pools. Of the regenerating forest that was cleared during the study period, 55% regenerating forest prefire C pool was lost during the slash fire. Subsequent burning following conversion to pasture on a two-year cycle likely depleted the original regenerating forest C pool. Of the original area in pasture in 1984, this entire pasture C pool was lost by 1992 with additional losses likely depleting C sequestered in new pasture grasses generated following fire. If the area in pasture in 1984 had burned only once by 1992, only 28% of the 1984 pasture C pool would have been lost by biomass combustion alone.

Table 4.5. Annual C, N, and S loss from buring of primary forest, regenerating forest, and pasture in the Jamari, Rondônia study area (94,372 ha) during the study period (1984-1992).

Component	C Loss kg ha^{-1}	N Loss g ha^{-1}	S Loss g ha^{-1}
Primary Forest			
Annual loss (1984-1986)	1,165	16,767	1,362
Annual loss (1986-1992)	1,101	15,851	1,288
Regenerating Forest			
Annual loss (1984-1986)	103	1,996	122
Annual loss (1986-1992)	12	231	14
Pasture			
Annual loss (1984-1986)	79	1,159	86
1987	188	2,765	205
1988	269	3,952	293
1989	334	4,912	364
1990	415	6,100	452
1991	496	7,288	541
1992	577	8,475	629

Dynamics of N Pools and Losses

The 1984 N pools for the study area were estimated as 1,947 kg ha^{-1} in primary forests, 19 kg ha`1 in regenerating forests, and 6 kg ha^{-1} in pastures (Table 4.3). The difference between the total 1984 N pool (1,971 kg ha^{-1}) and the pre-disturbance N pool (2,051 kg ha^{-1}) was 80 kg ha^{-1}, a loss of 4% from the pre-disturbance pool. By 1992, the direct loss of the N pool for the study area in 1984 was depleted by 129 kg ha^{-1} due to burning of primary forest (Figure 4.4). This loss represents 7% of the 1984 N pool for the study area. Following primary forest conversion to pasture, cumulative pasture burning on a two-year frequency contributed to additional N losses of 23 kg ha^{-1} (15 kg ha^{-1} for three-year burning scenario) by 1992. N losses were 5 kg ha^{-1} for burning the areas of regenerating forest for conversion to pasture. Additional N losses from regenerating forest areas associated with subsequent burning as pasture were 4 kg ha^{-1} (two year burning frequency) or 3 kg ha^{-1} (three year burning frequency) by 1992 (Figure 4.4). N losses increased annually as the forest areas were converted to pastures each year between 1984 and 1992. Between 1984-1986, we estimated that annual N losses associated with predicted pasture burning was 1 kg ha^{-1} for the 615 ha of pasture burned (50% of area in pasture each year is burned) (Figure 4.2, Table 4.5). By 1992, the area in pasture had increased to 4,499 ha with an annual loss of 8 kg ha^{-1} from pasture burning. The cumulative N loss from burning and reburning pastures between 1984 and 1992 was 36 kg ha^{-1}. This eight-year time period represents a 2% loss from the 1984 N pool. The cumulative N loss from forest conversion and pasture burning for the study period was 170 kg ha^{-1}, a rate of 1% from the 1984 pool each year. Cumulative N loss during this eight-year period represented an 8% loss from the pre-disturbance N pool. By 1992, the cumulative loss of N was 250 kg ha^{-1}, a 12% loss from the pre-disturbance N pool.

Of the area of primary forest burned during the study period for conversion to pasture, 69% of this primary forest N pool was lost. Slash fires in regenerating forest areas depleted these forest N pools by 78%. Subsequent reburning of these

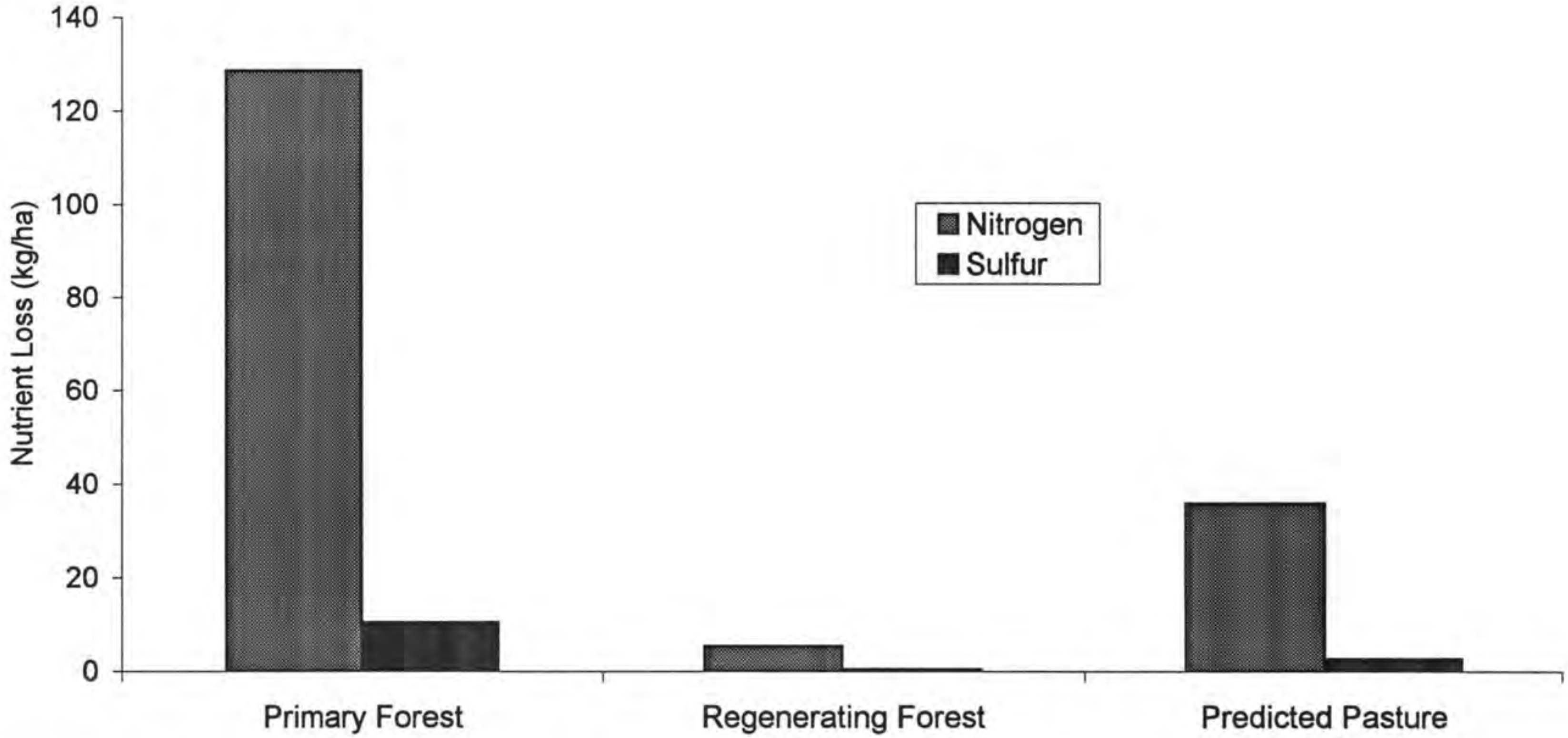

Figure 4.4. Nitrogen and sulfur losses associated with burning of primary forest slash, regenerating forest slash, and pastures in the Jamari, Rondônia, Brazil study area (1984-1992). The predicted pasture area includes subsequent burning of the 1984 area in pasture and additionally the primary and regenerating forest areas assumed to be converted to pasture following slashing and burning. These new pasture areas were burned every two years.

areas as pasture depleted the regenerating forest N pool. Of the 1984 pasture N pool, N loss associated with a two-year burning frequency consumed the 1984 pasture N pool. Even under a three-year pasture burning frequency, 1984 pasture N pools were depleted by the end of the eight-year study period.

Dynamics of S Pools and Losses

In 1984, S pools for the study area were 197 kg ha^{-1} in primary forests, 2 kg ha^{-1} in regenerating forests, and <1 kg ha^{-1} in pasture (Table 4.3). The cumulative S pool for 1984 was 199 kg ha^{-1} and represented a 4% loss from the pre-disturbance S pool. Between 1984 and 1992, S pool losses for the Jamari study area were 10 kg ha^{-1} for primary forest slash representing a 5% loss from the 1984 S Pool (Figure 4.4). Following conversion of these deforested areas to pastures, subsequent burning as pasture (two-year frequency) increased S losses by 2 kg ha^{-1} or 1 kg ha^{-1} (three-year burning scenario). S loss due to burning of the regenerating forest areas was <1 kg ha^{-1} with additional losses following conversion to pasture of <1 kg ha^{-1} on the two-year pasture burning frequency or <1 kg ha^{-1} on a pasture burning frequency of three years. For the cumulative area of pasture burned for the study period, 3 kg ha^{-1} of S were lost for burning on a two-year frequency (Table 4.3, Figure 4.4). This pasture burning scenario represented less than a 1% loss of the 1984 S pool. If the pastures were burning on a three-year frequency, S losses were 2 kg ha^{-1}. Between 1984-1986, we estimated that annual S losses were <1 kg ha^{-1} from burning of pasture on a two-year frequency, however, by 1992 annual S losses were almost 1 kg ha^{-1} (Figure 4.2, Table 4.5). During the eight-year study period, cumulative S losses were 14 kg ha^{-1}; a 7% loss of S during this period at a rate of less than 1% loss per year. The cumulative S loss from the pre-disturbance pool was 22 kg ha^{-1}, an 11% loss from the pre-disturbance S pool.

Of the primary forest cleared for conversion to pasture, 55% of this forest S pool was consumed by the end of the study period. Burning of regenerating forest slash for conversion to pasture, with subsequent burning as pasture, completely depleted the regenerating forest S pool. Burning to maintain the pasture area as of 1984 essentially consumed the entire 1984 pasture S pool levels.

Discussion

Biomass burning in the Amazon contributes to elemental losses through fluxes of emissions and aerosols and erosion of elemental pools in ash. Transport of these elements away from their sources may result in nutrient depletion and hence, decreased ecosystem productivity [*Fearnside*, 1997]. The proportion of the nutrients lost through combustion processes is influenced by factors such as, the distribution of the nutrients in the biomass (or fuel) components, the combustion factor, and the temperature of volatilization of each element [*Kauffman et al.*, 1992, 1998]. The magnitude of C lost compared to N and S is much greater in all of our sites for the reasons of the high proportion of C found in the biomass and a low temperature of volatilization. N also has a low temperature of volatilization, but the concentration of N in the fuel is lower. S has a higher temperature of volatilization and a lower concentration in the fuels. Additionally, between land cover types (primary forest, regenerating forest, and pastures), elemental pools and losses are greatest in forest sites than in pastures due to the high proportion of the biomass represented in the woody components producing high masses of pools on these sites (Figure 4.1). These attributes along with high combustion factors for forest sites (Table 3.1) result in greater losses on forest sites.

The C concentration of fuel components (~50%) did not vary as the concentrations of N and S do with non-wood and wood fuel components. The C concentration of postfire ash is reported to be about half that of uncombusted fuels [*Guild et al.*, 1998; *Hughes et al.*, in press; *Kauffman et al.*, 1995]. Additionally, it was found that ash represents 1-6% of the postfire C pool supporting the evidence that the high proportion of C loss is associated with biomass combustion emissions due to the low temperature of the volatilization of C [*Guild et al.*, 1998; *Hughes et al.*, in press; *Kauffman et al.*, 1995]. Due to residual levels of C associated with postfire ash, substantial site loss of C via erosion of ash is less likely. Slow decomposition of remnant woody debris on sites would contribute to additional site losses of C.

Concentrations of N in non-wood fuels (i.e., rootmat, dicots, litter, and foliage) are about twice that of wood debris, the greater mass associated with the high proportion of woody components (i.e., stem diameters classes of 0-0.64 cm, 0.65-2.54 cm, 2.55-7.6 cm, 7.6-20.5 cm, > 20.5 cm, and palms) in forest and pasture sites in Rondônia result in 56-78% of the N pool represented in the wood debris [*Guild et al.*, 1998; *Hughes et al.*, in press; *Kauffman et al.*, 1995; 1998]. Since the mass of N is highest in woody components and forest sites have higher N pools than pastures, forest site burns contribute to greater N losses than pastures (Figure 4.1, Table 4.1). This underscores the importance of burning of cut forest in the contribution of N losses. N has a low temperature of volatilization, resulting in most of the N being lost in combustion processes and only low concentrations of N remain in the ash [*Kauffman et al.*, 1992]. In Rondônia, the postfire N pool in ash has been reported to be 9-22% [*Guild et al.*, 1998; *Hughes et al.*, submitted; *Kauffman et al.*, 1995]. N is more likely to be lost via erosion than C since a higher proportion of the postfire pools are found in the ash.

S is found in higher concentrations of the fine fuels including rootmat, dicots, litter, and foliage, however, the mass of non-wood fuels is dominated by wood fuels in the forest and pasture sites. As with N, a substantial portion (52-77%) of the S pool is sequestered in the coarse woody components of the sites studied in Rondônia [*Guild et al.*, 1998; *Hughes et al.*, submitted; *Kauffman et al.*, 1995]. Since the S pool is higher in wood debris and higher forest sites than in pasture sites, the contribution of S losses is less in pasture sites than forest sites. Ash typically represents 14-57% of the S pool following fire [*Guild et al.*, 1998; *Hughes et al.*, submitted; *Kauffman et al.*, 1998]. S has a higher temperature of volatilization compared to C and N, and is found in higher proportions in the postfire ash [*Kauffman et al.*, 1995; 1998]. It is more likely that S be susceptible to erosion than C and N.

Quantifying the terrestrial C, N, and S losses associated with increasing land use intensity during this period of early colonization demonstrated the variation in losses and the importance of delineating forest clearing and land conversion rates through time on landscapes experiencing rapid change. Other research in Rondônia has primarily estimated C emissions losses from satellite derived deforestation estimates at local and regional scales, but few estimates of C pools and losses at these scales have been derived from scaling up site measurements [*Fearnside*, 1997; *Fujisaka et al.*, 1998; *Moraes et al.*, 1998]. The estimates of C (and nutrient) loss for our study are associated only with combustion of aboveground biomass and do not include all possible paths of loss.

Moraes et al. [1998] used one date of TM data (1991) to map pastures of different ages and integrated published data on vegetation and soil C stocks to estimate C release during deforestation and different pasture stages (0-3 years, 3-5 years, and 5-20 years). C pools for aboveground and belowground forest and pasture biomass and soil (0-30 cm) were estimated from published data. Losses from burning and decay during the pasture stages was also estimated from published data. It was reported that C flux remained as a source of CO_2 to the atmosphere until the reaching five years under pasture. Following five years under pasture, it was found that most of the unburned woody debris was consumed and there was an increase of C sequestration in pasture grass and accumulation of C in biomass belowground. Therfore, the CO_2 emitted from pastures older than five years function as a sink. These methods identify the shortcomings of our research in that we did not delineate pasture C losses associated with age since deforestation or losses associated with decay. Additionally, we did not include belowground pools and losses in our estimates. Multi-date TM data may have improved the classification results of land cover in *Moraes et al.* [1998] and additionally may have allowed identification of regrowth areas for estimating the contribution of these areas to the C balance.

There are additional processes that decrease C pools in deforested areas other than from burning. Decomposition by microbes and termites of uncombusted biomass was utilized by *Fearnside* [1997] to estimate Amazon wide deforestation emissions of CO_2. In addition to aboveground C losses, there are C losses associated with soil and decay of belowground biomass [*Fearnside*, 1997; *Fujisaka et al.*, 1998]. Fearnside's [1997] approach to estimating emissions for the Legal Amazon is based on the C stock per hectare and the partitioning of C released as the unit area is transformed from the forest to the resultant land use of pasture over a period of 10 years. Sources of C emissions used by *Fearnside* [1997] include: initial burning of forest, pasture reburning, decomposition (aboveground and belowground), soil C, and cattle.

Fujisaka et al. [1998] estimated C emissions based on aboveground (trees, understory, and charcoal) and belowground (roots and soil) C stock losses associated with deforestation in the area of Theobroma, Rondônia. These C emissions are calculated as the C stock of the original forest area subtracted by the C stock of the area estimates of replacement land use vegetation (i.e., pasture, fallow, and annual crops). In the 216,500 ha study area, 93,211 ha (43% of the study area) was deforested by 1993 with C emission estimate of 14 Tg over a 20-year period (1973-1993). On an annual basis for the Theobroma study area, C emissions associated with deforestation and land use were approximately 3 Mg C ha^{-1} yr^{-1}. For our 94,372 ha study area, of which only 14% was deforested by 1992, our total C pool loss was 12 Mg ha^{-1} or approximately 1.5 Mg C ha^{-1} yr^{-1} and this estimate included reburning of pasture areas. The forest and fallow aboveground C stocks used in the Theobroma study were within the range for our primary and regenerating forest measurements, but the aboveground pasture C stocks for our estimates were five-fold greater than those used in the Theobroma study. This could be associated with pasture sites in the Theobroma study being much older than our sites and perhaps remnant wood following forest slash fires was mechanically removed. The Theobroma study is situated in an area subjected to wide-spread pasture management practices for a longer period than that experienced in our study area. The differences between the Theobroma study and ours is likely due to the proportion of area deforested in the study area and not from the estimates of C stocks.

We estimated that cumulative C and nutrient losses for the study area between 1984 and 1992 were 11,650 kg C ha^{-1}, 170 kg N ha^{-1}, and 14 kg S ha^{-1} (Tables 4.3, 4.4). On an annual basis, losses were 1,456 kg C ha^{-1} yr^{-1}, 21 kg N ha^{-1} yr^{-1}, and 1,750 g S ha^{-1} yr^{-1}. These estimates include forest burning and pastures burned (including forests converted to pasture) on a two-year frequency. We can compare the site C pool losses estimated here with the C emissions we estimated in Chapter 3, however, N and S emissions were not estimated and will not be compared. In both methods of estimating C loss, we scaled up site measurements to the landscape level of our study area. The correspondence of the estimates in both approaches may serve as a validation of the different methods in effectively scaling up site measurements to the landscape level. Testing these methods in other regions of Rondônia experiencing different levels of deforestation and land use intensity would be useful in evaluating if these methods would be appropriate for scaling site measurements to the regional scale. These methods could be a means to improve regional estimates of C and elemental pool losses in areas undergoing varying rates of change and hence may provide better estimates for global approximations of C and elemental pool dynamics.

Total C losses based on burning emissions estimates for the study area between 1984 and 1992 were 1,486 kg C ha^{-1} yr^{-1} (Figures 3.5, 3.6). This value was derived by summing the mass of C emitted in each of the emissions that we quantified. These C emissions include CO, CO_2, CH_4, C_2H_2 (ethyne), C_2H_4 (ethene), C_2H_6 (ethane), C_3H_4 (propyne), C_3H_6 (propene), and C_3H_8 (propane). The emissions C loss estimate is only 2% higher than the C loss calculation based on our approach at measuring loss through combustion of C pools. Since C emissions of CO and CO_2 from biomass combustion represent 95 to 99% of the C released from biomass burning [*Ward and Hardy*, 1991], we are confident that we are not underestimating the mass of C emitted based on the suite of emissions we have selected to quantify.

Conclusion

We quantified terrestrial C, N, and S losses associated with biomass burning in an area of increasing land use intensity during a period of early colonization 1984-1992 in Rondônia. The results demonstrated the variation in losses by land use/land cover type and the importance of delineating forest clearing and land conversion rates through time on landscapes experiencing rapid change. As forested areas were slashed and burned during the study period, our estimates show that these sites continued to contribute to additional elemental losses as new pasture areas were maintained by a two to three year burning frequency. By 1992, elemental losses from pasture burning were seven times greater than in 1984 due to increasing area in pasture from forest conversion. We found that emissions in sites initially are relatively low, but increase as colonization and deforestation rates increase.

During the 1984-1992 study period, losses of 7%, 9%, and 7% of the 1984 C, N, and S pools respectively were reported in our study area. The difference in 1984 elemental pool levels from pre-disturbance pools along with the 1984-1992 estimated elemental losses resulted in a reduction of 11% C, 12% N, and 11% S from pre-disturbance elemental pool levels.

Repeated burning of sites following conversion of forest to pasture effectively consumes the C, N, and S pools of the original forested sites. Under frequent pasture burning practices, we found that established pasture elemental pools were consumed within eight years under pasture management. For the cumulative area of primary forest cut and burned followed by conversion to pasture, 58% of the C pool, 69% of the N pool, and 55% of the S pool was depleted during the eight-year study period (1984-1992).

This approach to quantify C, N, and S pool losses during the 1984-1992 study period provided an important validation to the approach of estimating C losses by quantifying biomass combustion C emissions for the study area (Chapter 3). There was only a 2% difference in the results of the two approaches.

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Chapter 5

Summary

Investigation of deforestation and land use in the vicinity of Jamari, Rondônia between 1984 and 1992 during a period of early colonization and agricultural expansion yielded areal estimates of forest slash and pasture burning. In addition, based on the areal estimates of land cover and change, the quantity of emissions arising from biomass burning and elemental pool losses following burning were estimated. The significant results and conclusions of this investigation include:

- Spatial and temporal extent of forest clearing (primary and regenerating forests) and pastures were quantified utilizing multi-date Landsat Thematic Mapper (TM) data between 1984 and 1992. The Tasseled Cap (TC) spectral transformation enhanced the contrast between land cover types of forest, regrowth, and cleared areas. Classification of the multi-date TC composite produced a land cover and change map for the 94,372 ha study area with high accuracy (79% overall accuracy and Kappa Statistic of 78%) for 18 classes. We found that area estimates of deforestation and land clearing are better interpreted in a temporal series of satellite data at a frequency to adequately capture change rather than from single-date analysis.
- Integration of knowledge of land use dynamics in the Amazon along with the TM derived land cover and change map for the period of 1984-1992 allowed modeling of cumulative area of pasture burned between 1986 and 1992.
- Between 1984 and 1992, 8,250 ha of primary and 828 ha of regenerating forest were cleared. This represented 18% and 1% of the study area, respectively. Areas in a state of sustained clearing (not returning to regrowth) increased from 1% (1,231 ha) in 1984 to 3% (2,692 ha) of the

study area by 1986 and remained cleared by the end of the study period in 1992. This area was assumed to be pasture during the study period. The cumulative area of pasture burned during the study period comprised 19,008 ha.

- In an emissions model, local scale emissions of CO, CO_2, CH_4, and hydrocarbon trace gases were computed by scaling up: 1) ground-based estimates of prefire and postfire aboveground biomass and biomass combustion factors delineated by fuel categories and land cover type; and 2) tower derived estimates of emission factors of flaming and smoldering combustion by land cover type. Refinement of emissions estimates were made by delineating flaming and smoldering combustion emissions. The emissions model was modified to utilize data of biomass, combustion factor, and fraction of biomass burned delineated by fuel categories of specific land cover types.
- Based over the entire study area, a mean of 138 kg C ha^{-1} yr^{-1} in CO, 920 kg C ha^{-1} yr^{-1} in CO_2, and 16 kg C ha^{-1} yr^{-1} in CH_4 were generated from primary forest slash burns. Regenerating forest slash fires contributed 4 kg C ha^{-1} yr^{-1} in CO, 31 kg C ha^{-1} yr^{-1} in CO_2, and <1 kg C ha^{-1} yr^{-1} in CH_4. The cumulative area of pasture burned produced 46 kg C ha^{-1} yr^{-1} in CO, 316 kg C ha^{-1} yr^{-1} in CO_2, and 4 kg C ha^{-1} yr^{-1} in CH_4.
- With increasing area in pasture as well as frequent reburning of pastures, the mounting importance of pastures in the contribution of emissions is revealed in our results. Hence, the conversion of forests to pastures represents a potential long-term source of emissions, but has received little attention. In other areas of Rondônia that are further along in the process of agriculture expansion, pasture burning emissions could potentially rival those from deforestation. Consideration of these anthropogenic sources of emissions is essential for reporting regional and Amazon emissions estimates.

- The emissions modeling research is based upon aboveground biomass burning, however, there are additional sources of emissions that arise from the land use/land cover change areas besides those directly from combustion not estimated by our techniques. These additional sources of emissions include decomposition (aboveground and belowground), soils, and flooding of areas due to hydroelectric dams.
- The methods presented in the emissions research could be applied to other states in Amazonia to scale-up local ground-based measurements to estimate state biomass burning emissions and ultimately improve Amazon Basin biomass burning emissions associated with land use. To accomplish this, appropriate ground-based data on emission factors, TAGB, and combustion factors would need to be integrated with regional deforestation and land clearing estimates.
- Site estimates of C, N, and S pools and resultant losses associated with burning of primary and regenerating forest slash and pastures were applied to the TM derived land cover change map and the modeled estimate of cumulative area of pasture burned. Integration of this data allowed scaling up of site pools and losses of C, N, and S to the landscape level corresponding to the study area.
- In 1984, study area elemental pools were 163,297 kg C ha^{-1}, 1,971 kg N ha^{-1}, and 199 kg S ha^{-1}. Between 1984 and 1992, 5%, 7%, and 5% of the C, N, and S pools, respectively were lost from primary forest slash fires. Also, cumulative pasture burning contributed to losses of 1%, 2%, and 1% of the C, N, and S pools, respectively. As forested areas were slashed and burned during the study period, these sites continued to contribute to additional elemental losses as new pasture areas were maintained by frequent burning. By 1992, elemental losses from pasture burning were seven times greater than in 1984 due to increasing area in pasture from forest conversion.

- We estimated that cumulative C and nutrient losses for the study area between 1984 and 1992 were 11,650 kg C ha^{-1}, 170 kg N ha^{-1}, and 14 kg S ha^{-1}. On an annual basis, losses were 1,456 kg C ha^{-1} yr^{-1}, 21 kg N ha^{-1} yr^{-1}, and 1,750 g S ha^{-1} yr^{-1}.
- In the absence of deforestation (prior to 1984), the C, N, and S pools of the 94,372 ha study area were estimated to be approximately 16 Tg C (170,004 kg C ha^{-1}), 2 x 10^{-1} Tg N (2,051 kg N ha^{-1}), and 2 x 10^{-2} Tg S (208 kg S ha^{-1}) in primary forest. We estimated that elemental pools for the study area at the beginning of the time frame for this study in 1984, were 161,369 kg C ha^{-1} in primary forest, 1,368 kg C ha^{-1} in regenerating forest, and 561 kg C ha^{-1} in pasture. The 1984 N pools for the study area were estimated as 1,947 kg ha^{-1} in primary forests, 19 kg ha^{-1} in regenerating forests, and 6 kg ha^{-1} in pastures. In 1984, S pools for the study area were 197 kg ha^{-1} in primary forests, 2 kg ha^{-1} in regenerating forests, and <1 kg ha^{-1} in pasture. By 1984, 4% of the pre-disturbance C, N, and S pools were lost, however, the losses associated with the cumulative area of reburnings (shifting cultivation and pasture) that occurred in deforested areas prior to 1984 (beginning of our study period) was not estimated and not included in the total C, N, and S loss estimates. Due to the relatively low proportion of primary forest cleared by 1984 and the likelihood of areas in pasture were of relatively recent origin, we assume that pasture reburns per site were low and, therefore, the contribution of these pasture emissions were low compared to primary forest slash emissions generated.
- By 1992, the cumulative C, N, and S losses in the study area since pre-disturbance were 2 Tg C (18,357 kg C ha^{-1}), 2 x 10^{-2} Tg N (250 kg N ha^{-1}), and 2 x 10^{-3} Tg S (22 kg S ha^{-1}). By 1992, the difference in elemental pool levels from pre-disturbance pools resulted in a reduction of 11% C, 12% N, and 11% S from pre-disturbance elemental pool levels.

- Quantifying the terrestrial C, N, and S losses associated with increasing land use intensity, during this period of early colonization, demonstrated the variability in losses and the importance of delineating forest clearing and land conversion rates through time on landscapes experiencing rapid change. We found that emissions in sites initially (1984) were relatively low, but increased as colonization and deforestation increased during the study period.
- We estimated that cumulative C losses for the study area between 1984 and 1992 were 1,099,385 Mg C (11,650 kg C ha^{-1}). On an annual basis, C losses were 1,456 kg ha^{-1} yr^{-1}. These estimates include forest burning and pastures burned (including forests converted to pasture) on a two-year frequency. We compared the site C pool losses with the C emissions. Total C losses based on burning emissions estimates for the study area between 1984 and 1992 were 1,121,726 Mg C (1,486 kg C ha^{-1} yr^{-1}). This value was derived by summing the mass of C emitted in each of the emissions that we quantified. The emissions C loss estimate is only 2% higher than the C loss calculation based on our approach at measuring loss through combustion of C pools. In both methods of estimating C loss, we scaled up site measurements to the landscape level of our study area. The correspondence of the estimates in both approaches may serve as a validation of the different methods in effectively scaling up site measurements to the landscape level.

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