Gradient analysis of biomass in Costa Rica and a first estimate of country wide emissions of greenhouse gases from biomass burning

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ACKNOWLEDGEMENTS

INTRODUCTION

The United Nations Framework Convention on Climate Change, signed at the UN Conference on Environment and Development in Rio de Janeiro during June, 1992, came into effect on 21 March 1994, when more than 50 nations ratified the agreement. The objective of the agreement was to stabilize atmospheric concentrations of greenhouse gases (GHGs) and thereby prevent anthropogenic interference with the earth's climate. The carbon balances of the earth's forests are important to sustaining the earth's climate, among other things. Two of the agreement's major provisions are especially relevant to this analysis: "preparing national reports on how to reduce emissions and/or expand carbon sinks;" and "providing financial and technical assistance to developing countries for inventories of greenhouse gas emissions" (Hecht and Tirpak 1995).

An important component of sustainability issues is therefore the degree to which a region or nation is balance with respect to atmospheric gases. Clearing and burning of tropical forests for expansion of pasture and agriculture or for other reasons, most associated with economic expansion, release GHGs such as CO_2, CH_4 and N_2O to the atmosphere. Other gases, such as CO and NO_x, also are released, which may cause respectively elevated ozone levels or acid precipitation. Both deforestation and associated emissions of GHGs have the potential to impact biogeochemical cycles (Crutzen and Andreae 1990, Houghton 1995), climate and hydrology at local and global scales (e.g. Salati and Vose 1984, Fleming 1986, Luvall and Uhl 1990), and the availability and quality of land and water resources (Young 1994). Many of these processes, such as global climate change, cross political boundaries.

Although each hectare of forest that is cleared and burned releases only small amounts of gas, the cumulative impact can be very large. The global total of C emissions from fossil-fuel burning and cement manufacture in the decade of the 1980s was about 5500 Tg yr^{-1} (Schimel et al. 1995; 1 Tg $= 10^6$ Mg $= 10^{12}$ g). Net C emissions from changes in land use during the same period have been estimated at 490 to 1600 Tg yr^{-1}, primarily from the tropics (Detwiler and Hall 1988; Hall and Uhlig 1991; Houghton et al. 1995). Biomass burning, occurring mainly as fires for clearing of tropical forests and savannas and fuel wood burning, also releases trace GHGs. Such burning contributes about 40 Tg C as the trace gas methane annually, which is about 8% of global CH_4 emissions (Prather et al. 1995); methane molecules have about 20 times the warming potential of CO_2 molecules. Thus about 50% (40 Tg CH_4-C x 20/1600 Tg CO_2-C) of the warming potential from biomass burning derives from CH_4 emissions. While biomass burning contributes only about 3% of global emissions of the trace gas N_2O (Prather et al. 1995), its warming potential per molecule is about 200 times that of CO_2 (Lashof and Ahuja 1990).

Estimates of net emissions of GHGs from biomass burning are uncertain for many reasons. Key uncertainties are the quantity of biomass fuel originally on the areas burned (Hao et al. 1991), the portion of that fuel consumed and emitted as various gases in a

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burning event, and the area impacted (Detwiler and Hall 1988). Depending on forest type and human disturbances, biomass and thus fuel can vary from less than 40 Mg ha^{-1} for a dry, open canopy forest, to greater than 500 Mg ha^{-1} for a well-developed primary humid forest (Brown et al. 1993; Brown 1996). Thus GHG emission estimates from biomass burning are better when higher resolution data are used in the analysis, such as country specific data subdivided along environmental gradients such as life zones (*sensu* Holdridge 1967). Burn combustion characteristics, which influence emissions from forest burning, also vary by forest type. Generally in more humid forests combustion is both less complete and less efficient. The result is that trace gas and particulate emissions are higher in humid as compared to dry forests (Kauffman et al. 1992). Thus information specific to climatic forest formation is also important to estimate emissions of trace GHGs, such as CH_4 and N_2O.

However, past efforts to estimate GHG emissions from biomass burning generally have relied on global or regional data bases (e.g. Hao et al. 1991). Although the Central American country of Costa Rica is small in area (50,060 km^2), it has several forest formations for which data on forest structure and areas deforested are available. Between 1950 and 1984, the rate of deforestation in Costa Rica was one of the highest in the world at about 3.9% per year (Harrison 1991; see Chapter 16).

In addition, as ecologists and geographers (Tosi and Voertman 1964, Sader and Joyce 1988, Janzen 1988) have noted, humans inhabit and clear drier forests preferentially to wet ones. Lower elevation forests in Costa Rica were also cleared preferentially. Therefore the country provides an excellent "microcosm" for an analysis of GHG emissions by forest type.

Costa Rica is located in Central America between 8° and 11° N and 82° to 85° W. It is bordered by Nicaragua to the north and Panama to the southeast, and by the Caribbean and Pacific Oceans. Down its center from northwest to southeast lies a series of mountain ranges along the continental divide that grow progressively larger toward Panama. Valleys have formed between the continental divide and smaller mountain ranges along parts of the Pacific slope. Seasonal variations in the strength of the trade winds interact with the mountain chains to create rain shadows, vortices and eddies. Thus, while both the Pacific and Caribbean slopes have both a dry and wet season, the Caribbean tends to have a longer rainy season, and both slopes are influenced by various ocean winds (Stiles and Skutch 1989). Under these diverse climatic and topographic conditions, many forest types have formed in areas of different temperature and rainfall regimes and elevation.

With the steep environmental gradients and consequent range of forest formations in Costa Rica, our first objective in this chapter was to estimate biomass of Costa Rican forest stands undisturbed by recent human activity (as far as we can tell) as a function of environmental gradients.

Our second objective was to estimate the release of GHGs to the atmosphere from forests developed under differing climatic conditions. We estimate emissions simply by assuming that 100% of aboveground biomass is burned eventually. This assumption simplifies the the calculations for GHG emissions from burning. This simple method also is

recommended in the Intergovernmental Panel on Climate Change and Organization for Economic Cooperation and Development (IPCC/OECD) methodology for inventory of GHG emissions (Houghton et al. 1995). Therefore, we feel that comparing the results from this simple approach, with the more sophisticated models used in Chapter 16, can provide information on differences between the two estimations.

Our final objective was to integrate the biophysical variability in emissions from biomass burning with the spatial and temporal pattern of how humans clear forests. Thus we evaluate the net result of the fact that environmental gradients affect both the rates and pattern of deforestation as well as the GHG emissions per hectare of forest cleared.

METHODS

We derived biomass estimates from a series of previously sampled plots of tree diameters and densities, and analyzed the results as a function of environmental gradients and by Holdridge Life Zones (Holdridge et al. 1971; the life zone approach is a particular gradient analysis using rain, evapotranspiration and temperature). From these estimates we derived national inventories of biomass using estimates of the areal extent of forest cover, by life zone, for 1940 and 1983 (Sader and Joyce 1988). The change in biomass associated with forest removal between the two dates was used to estimate transfer to the atmosphere of trace gases, including carbon as CO_2, using emission factors based on biomass burned, from previously published information (Ward et al. 1992, Hao et al. 1991).

Aboveground biomass density

Costa Rica is a well-studied country from the perspective of its biota. Yet in order to determine whether its greenhouse gas emissions meet with the goals of the agreement reached during the UN Conference in Rio de Janiero, data are needed on the past, current and potential biomass of its varied forest types. While the maximum biomass a forest can attain depends on its climate and soils (Brown and Lugo 1982, Chapter 7), the biomass present in a given stand is also dependent on its age and disturbance history. We wanted to estimate how biomass might have been distributed spatially across Costa Rica's climatically varied landscape when much of its forest was relatively undisturbed by human activities and therefore near a maximum potential biomass. Such data indicate the probable spatial distribution of forest biomass in the past and the biomass distribution possible under future scenarios.

We calculated aboveground forest biomass (AGB), in Mg dry-weight biomass ha^{-1}, from forest stand data collected in 1964-1966 (Holdridge et al. 1971). The stands were described as "relatively undisturbed," and areas described in the text as having experienced obvious human disturbance are not included here in mean biomass values. Riparian and swamp sites were not included in the analysis because information on changes in their extent was not available. Detailed data also were not available on the structure or deforestation extent of wind-clipped, short-statured wet montane forest known as "elfin" forest (Lawton

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and Dryer 1980), or other subalpine formations. Kappelle (1991) notes that these forests do not usually exceed 12 m in height, and have an extent of about 3,750 ha in Chirripo National Park. In each life zone, Holdridge et al. (1971) established one to 10 plots of 0.1 to 1.2 ha. Most of the 33 plots were 0.3-0.5 ha (19); 7 were 0.1-0.2 ha; 5 plots were 0.6-0.8 ha; and two plots were 1 ha or larger. Within each of these plots diameter (dbh) and height for all trees with a dbh $\geq$ 2.5 cm were measured; species also were identified. The number of plots per life zone varied and are listed in Table 1.

The biomass values we calculated could over or underestimate actual stand biomass because of the small plot sizes used. Plot sizes of 1 ha or more are generally recommended for estimation of biomass for an ecosystem in an ecological study, and for regional biomass estimates, more extensive forest inventories are desirable (Brown et al. 1989). Only two plots were 1 ha or more in size. However, this data set was the only set available that surveyed in a consistent fashion forest stands representative of the various climatic forest formations in Costa Rica.

This series of previously sampled plots of tree diameters were distributed geographically along climatic gradients to represent the range of forest environments in Costa Rica (Holdridge et al. 1971). We used techniques developed in Brown et al. (1989), that use regression equations, stratified by life zone, to calculate average mass per tree (in kg) from the tree diameters. The biomass regression equations for broadleaf forests were developed from a large data base that includes trees of many species harvested from forests from all three tropical regions of the world. It includes a total of 371 trees with a diameter range of 5 to 148 cm from ten different literature sources, and it has been updated recently (Brown 1997).

The data we used were in the form of stand tables listed in the vegetation data summary for each site (Holdridge et al. 1971). These tables give the number of trees, as tree density in number/ha, by the following diameter classes: 2.5-7.5, 7.5-15, 15-30, 30-60 cm, and >60 cm. Tree by tree data have since been destroyed, so we used the reported stand tables (number of trees per ha by diameter class). Estimation of biomass from the stand tables and regression equations consisted of calculating the biomass of a tree with dbh set at the midpoint of a given diameter class, multiplying that biomass by the number of stems in that class, repeating that calculation for each diameter class, and summing the biomass over all diameter classes. The quadratic mean diameter of each diameter class may be more representative of each class diameter distribution than the class midpoint (Brown 1997). However, basal area was not reported by diameter class in the stand tables. Thus we were not able to use quadratic mean diameter for calculation of average biomass per tree.

Potential problems with the regression approach that we used to estimate biomass include: (1) the choice of the correct biomass regression equation to apply to a given site, (2) the small number of large diameter trees used to develop the regression equations (e.g., for the moist tropical equation, the largest dbh was 148 cm, with only 5 trees >100 cm diameter, therefore diameters should be less than about 148 cm when using this equation),

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(3) the open-ended nature of the >60 cm diameter class in stand tables, (4) the wide and uneven-width diameter classes in stand tables, (5) the difficulty with selecting an appropriate average diameter to represent a diameter class, and (6) the occasional missing diameter classes (i.e., incomplete stand tables to minimum diameter of 10 cm, which was not a problem with the Holdridge data set). We describe how we addressed these shortcomings in the paragraphs which follow.

The biomass regression equations were stratified into four climate types or life zones: very dry, dry, moist and wet. In lowland forests, rainfall can indicate the appropriate regression equation to use. In lowland dry, moist, and wet forests, respectively, rainfall is usually about <1500 mm, 1500 to 4000 mm, and >4000 mm annually (Holdridge 1967), and these ranges can indicate which regression equation to use. In addition, Brown (1997) lists two possible dry forest equations: one for sites with annual rainfall <900 mm (very dry forest), developed for a Mexican forest (original data source Martinez-Yrizar et al. 1992), and one from a dry forest in India where rainfall exceeded 1200 mm.

In drier forest formations, forest stature is indicative of rainfall and dry season length, and can help determine which of the dry forest regression equations is most appropriate for estimating biomass. In addition, because the biomass regression equations do not specifically include height, we assumed that using regression equations derived from forests with heights similar to the stand of interest will yield the best biomass estimates. Therefore, we used forest stature, rather than rainfall, to indicate which of the two dry forest equations would be the most appropriate for biomass estimation in the dry zone Costa Rican stands.

The Costa Rican dry forest sites had very different structures, the closest rain gauge to them, at *Cañas*, recorded a relatively wet 1665 mm of precipitation annually, despite their classification as dry forests. In addition, the differences in these forests stature, from an average of 5 m in height to an average of 22 m, indicated that the one rain gauge did not represent precipitation for these varied forest sites adequately. We assumed that the small-statured, savanna-like stands probably received much less rainfall than 1665 mm annually, while the well-developed stands probably received closer to that level.
Consequently, those short-statured forest sites, which averaged 5 m in height, were assumed to be very dry forests, and we used that equation to estimate their biomass. Their height was close to the average height of 7 m for forest which Brown (1997) used to develop the very dry forest regression equation. The well-developed dry sites averaged 13 and 22 m tall, closer to the average height of 12 m for the forest from which the other dry forest regression equation was developed, where rainfall is >1200 mm. We also used the biomass regression equation for dry forest with rainfall >1200 mm to estimate biomass of the premontane moist forest sites due to their short stature of 15 to 22 m.

We adjusted for the wide diameter class of 30-60 cm, as reported in the stand tables in Holdridge et al. (1971), by assuming the stands had a j-shaped diameter distribution. A j-shaped diameter distribution, in which tree number decreases asymptotically with diameter class, is characteristic of mature stands (Gillespie et al. 1992). Gillespie et al. (1992)

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showed that this diameter distribution curve can be approximated by a straight line from one diameter class to the next. Using that approximation, we estimated tree density for two 15-cm subsets of the wide diameter class. In other words, we expected that using the midpoint of 45 cm for the 30-60 cm diameter class would overestimate biomass in that class if the majority of trees had diameters smaller than 45 cm, which is usually the case. Thus we developed a procedure, based on one presented in Gillespie et al. (1992), for dividing the number of trees in the 30-60 cm class into two smaller classes: 15-30 cm and 30-45 cm. The adjustment consisted of calculating the ratio of tree densities in two smaller diameter classes and assuming that ratio was equal to the ratio between two larger, 15-cm wide classes.

Another shortcoming is that Holdridge et al (1971) did not specify the individual diameters of the largest trees in each plot. Rather, they reported number of trees in an open-ended diameter class of ≥ 60 cm. To use the biomass equations, we needed a midpoint diameter for that size class. We estimated that midpoint diameter by the following sequence of calculations: (1) calculate the basal areas for all but the largest size class as the basal area for a tree with diameter equal to the class midpoint, multiplied by the number of trees in that class, (2) calculate the basal area for all trees in the largest size class as the difference between the sum of the basal areas calculated in the first step from the total stand basal area reported in Holdridge et al (1971), which did not report basal area by diameter class, (3) calculate the average diameter of the largest size class as the class basal area, calculated in step 2, divided by the number of trees in that size class.

This procedure did not work for nine of the thirty-three plots because the sum of the basal areas of the smaller size classes, those below 60 cm dbh, equaled or exceeded the total stand basal area reported by Holdridge et al. (1971). The over-estimation of basal area occurred even when trees were reported to be present in the > 60-cm size class. Our estimation of basal area in the smaller diameter classes, from the midpoint of wide diameter classes, probably overestimated the basal area in each size class. In these cases the majority of trees in the various diameter classes probably had a smaller diameter than the class midpoints (Gillespie et al. 1992). Therefore, because we were unable to calculate average diameter in the largest size class, we had to assume an arbitrary average diameter of 70 cm for the largest class in these stands. Then, we adjusted the final stand biomass estimate downwards by the ratio of total reported basal area to the sum of the class basal areas calculated from diameter class midpoints, including basal area from the trees with the assumed 70-cm average diameter.

Fine and coarse litter biomass and belowground biomass

Belowground biomass (BGB) was estimated from aboveground live biomass using a regression equation developed from 160 data points from published studies of biomass of tropical, temperate and boreal forests located worldwide (Cairns et al. 1997). The equation for estimating root biomass, including stumps is: BGB = exp{-1.0850 + 0.9256*AGB}, r^2 = 0.83.

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Fine and coarse litter was also estimated from AGB, by life zone, using data in Delaney et al. (1997), Saldarriaga et al. (1988), and Kauffman et al. (1988) for forests of Venezuela. From these studies, we calculated ratios of fine litter to AGB and dead wood to AGB by life zone (Table 1).

Fuel biomass and consumption

Fuel biomass in the Costa Rican forests was assumed equal to AGB plus fine and coarse litter biomass (necromass). We did not correct fuel biomass with combustion factors because we assumed that with repeated burnings the total biomass would eventually be burned. This calculation simplifies the process by which burning consumes cleared forest biomass. The first time a forest is burned, a portion of coarse components such as tree boles remain on-site (Kauffman et al. 1993), while most finer components, such as leaves, herbs, fine litter, and small branches, are largely consumed by fire (Kauffman et al. 1992, Kauffman et al. 1988). Different trace gas emission factors might apply to the remaining dead wood which is consumed in later cycles of pasture burning. In addition, for any given forest type, the combustion factor will change with fuel moisture content and fire conditions (e.g., wind speed, topographic influences, etc.; Kauffman et al. 1992). Although, repeated burning of regrowth would add to the emissions estimated here, shifting cultivation is now limited to relatively small regions in Costa Rica. Thus there is no need for complicated decay and regrowth dynamics such as in carbon dynamics model used by Detwiler and Hall (1988).

Trace greenhouse gas emissions

We derived the greenhouse warming potential resulting from forest destruction by estimating the release of those GHGs and correcting for the relative greenhouse potential of each. Ward et al. (1992) measured the combustion characteristics for several compounds and two to three phases of burning intensity during experimental burns in Brazilian tropical forests and savannas. We used Ward's estimates for trace gas emissions to estimate emissions of GHGs.

Ward et al. (1992) used the mass ratio CO/CO_2 in emissions from deforestation fires to calculate combustion efficiency values that varied from 0.84 to 0.95 in Amazonian forests (Figure 1). Combustion efficiencies varied from 0.92 to 0.95 during the flaming phases of drier savanna (*Cerrado*) fires, which consumed about 97% of the carbon released from burning. Ward et al. (1992) also found that about 52% of all carbon released from fires in primary moist forest occurred during the flaming phase of the burn, in which combustion efficiency was 0.88, and that most fine fuels were consumed by fire. Combustion efficiency during the ensuing smoldering phases was lower, about 0.84-0.85.

Thus Ward et al. (1992) found that the proportion of biomass converted into products of incomplete combustion varied by forest type. The emission factor (EF) for a combustion product is the mass of an emission product released per unit mass of fuel consumed (on a

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carbon or other basis). For particulate matter (PM) in smoke, and gases such as CO, NO_x, CH_4 and H_2, EFs increase with decreasing combustion efficiency (Ward et al. 1992). In addition, combustion efficiency decreases and therefore EFs increase as a forest burn proceeds from the initial flaming through smoldering phases (Ward et al. 1992).

We used the average of the EFs that Ward et al. (1992) published for the flaming phase and the two smoldering phases of moist primary forest for most of our estimates of burning emissions (Table 2). For the very dry forests, we used the average of two EFs published for two flaming phases of Amazonian savanna (*Cerrado sensu stricto*) because the 4-8 m stand height for this formation is similar to the stand height of about 5 m for the dry, savanna-like forest site sampled by Holdridge et al. (1971). A source of error in using these EFs from Amazonian savanna is that they were measured from burning of a natural fuel bed rather than a slash fire (Ward et al. 1992). However, comparable EF data from burning a slashed savanna were not available. We felt that these savanna EFs would be more representative than EFs from burning slash in a moist forest.

To estimate emissions of N_2O, we assumed that about 0.7% of the nitrogen in biomass is oxidized to N_2O based on an average N/C ratio (mole/mole) of 1.4% in tropical forest vegetation (Hao et al. 1991), and an average biomass C content of 50%. Molecules of CH_4 and N_2O are assumed to have 20 and 200 times the atmospheric warming potential of each molecule of CO_2 (Lashof and Ahuja 1990). To compare the total warming potential of these trace gases with CO_2 emissions, we multiplied the number of moles of each of those compounds emitted by their warming potential multipliers (20 and 200) and converted moles to g C.

RESULTS AND DISCUSSION

Biomass

The average estimated AGB ranged from 14.2 dry Mg (or Tons) ha^{-1} in the very dry forest to a high of 518 Mg ha^{-1} in lowland moist forest (Table 1). The trends in biomass by life zone were consistent with our understanding of the influences that environmental gradients have on forest biomass accumulation (Figure 2). Biomass tends be highest in moist tropical forests and lower where dry, very wet or colder conditions limit potential accumulation (Brown and Lugo 1982).

The very dry, savanna-like plots had very low biomass values, averaging 16 Mg ha^{-1}. Productivity in these sites is limited by hot, dry conditions during dry months, and soil saturation during the rainy season. The dry forest sites, where moisture deficits are a major limitation to productivity, contained the next lowest biomass, averaging 188 Mg ha^{-1} AGB. Published values for AGB for other dry forest sites ranged between the very dry and dry Costa Rican forest plots. The Guanica forest in Puerto Rico contains about 44.9 Mg ha^{-1} AGB. In second growth Brazilian dry forest, known as *Caatinga*, AGB was reported 74 Mg

ha^{-1} (Kauffman et al. 1993). Venezuelan very dry forest measured by Delaney et al. (1997) contained 140 Mg ha^{-1}.

Biomass as a function of temperature and moisture peaks in tropical moist forest. The Costa Rican lowland moist forest AGB, at 518 Mg ha^{-1}, is comparable to the 513 Mg ha^{-1} reported for the Ivory Coast by Huttel (1975). Moist forest biomass in Costa Rica appears to be much higher than moist forest reported in other areas. Saldarriaga et al. (1988) reported 255 Mg ha^{-1} for lowland moist forest in Venezuela. Fittkau and Klinge (1973) report 406 Mg ha^{-1} for Brazilian moist forest. Golley et al (1975) reported 264 and 378 Mg ha^{-1} for two moist, transitional to dry, sites in Panama.

Lowland wet forest biomass in Costa Rica is intermediate between moist and dry forest biomass values, and averaged 365 Mg ha^{-1}, which is greater than the 322 Mg ha^{-1} reported for Cambodian wet forest by Hozumi et al (1969).

Moisture limitation was apparent in the two premontane moist sites, which averaged 208 Mg ha^{-1} AGB. Their biomass is similar to values reported in Brown and Lugo (1982), which averaged 241 Mg ha^{-1}. The Costa Rican lower montane moist site contained 319 Mg ha^{-1} AGB, which is similar to the 346 Mg ha^{-1} reported in a comparable Venezuelan forest (Delaney et al. 1997).

All 10 higher elevation wet sites averaged 324 Mg ha^{-1}, which included all plots in premontane, lower montane and montane wet and rain forest. These values were similar to the 310 Mg/ha reported for lower montane rain forest in New Guinea by Edwards and Grubb (1977). They were lower than what Brun (1976) reported for montane wet forest in Venezuela, which contained 347 Mg ha^{-1}. The average for the Costa Rican plots was somewhat higher than the 279 Mg ha^{-1} reported for lower montane rain forest in Jamaica (Tanner 1980) or 198 to 223 Mg ha^{-1} reported for subtropical Puerto Rican lower montane rain forest (Frangi and Lugo 1985, Ovington and Olson 1970). For the premontane wet forest at La Selva, Costa Rica, our estimate of 389 Mg ha^{-1} was nearly identical to a previous estimate of 382 Mg ha^{-1} (Jordan 1985).

Also, we calculated biomass for a Costa Rican montane rain forest site based on a stand table presented in Jiménez et al. (1988). The calculation yielded an estimated AGB of 388 Mg ha^{-1}. That value is somewhat higher than our estimate of 309 Mg ha^{-1} for AGB of a comparable forest analyzed in this study, and higher than the 314 Mg ha^{-1} reported in Delaney et al. (1997) in a comparable Venezuelan forest.

Greenhouse gas emissions

In the case of Costa Rica, our estimates of carbon emissions from deforestation over the period of study were higher than previous estimates. The discrepancy is due partly to the fact that our estimates of biomass density in extensively cleared forest types, when based on country-specific data and stratified by environmental gradients, were higher than the globally-

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based biomass estimates used previously. This finding shows the importance of refining spatial analyses when undertaking analyses of greenhouse gas emissions.

The spatial pattern of GHG emissions reflects the general historical pattern of forest clearing and land development in Costa Rica, where the mesic and drier regions were developed first and then later wetter areas (Chapter 6). Thus trace gas emissions from deforestation are not proportional to the area of a given forest type. Sixty percent of trace gas emissions from deforestation and burning of the total fuel biomass between 1940 and 1983 (298 and 281 Tg, respectively) was from burning of lowland moist and wet forests, (Table 3 and Figure 2). The burning of lowland dry and moist forests, and premontane moist zones, contributed respectively 0.5%, 31% and 2% of emissions. The first two were disproportionate to their areas, which was already only 0.6%, 15% and 2.5% of forest area in 1940 due to earlier deforestation that had started in those regions (Chapter 13). Lower elevation, wet forests (T-wf, P-wf, P-rf) comprised 66% of total forest area in 1940, and together contributed 60% of emissions. In contrast, higher elevation wet forests (LM-wf, LM-rf, M-rf) comprised 14% of forest area in 1940, but together contributed only 7% of emissions.

By 1983, no original forest biomass remained in dry and moist life zones. In addition, while higher elevation wet forests zones comprised 12% of potential fuel biomass in 1940, they comprised 29% of potential fuel biomass in 1983. Most of the remaining biomass was in lower elevation, wet forest zones (69%). Almost no biomass remains in those forest zones which were settled by humans first: the dry and moist zones. While productivity in dry forest is similar to that in the higher elevation wet forests, productivity peaks in moist forest. Therefore, the moist forest regions, which have been settled for a long time, are also where most efficient carbon sequestration via secondary regrowth or plantation planting occurs.

We estimated annual release of carbon-containing compounds, over the 43-year period of 1940-83, as 8.9 Tg CO_2-C, 1.2 Tg CO-C and 0.14 Tg CH_4-C (Table 3). We also estimated an annual N_2O-N release of about 1.28 x 10^{-3} Tg. We multiplied these molar emissions of CH_4 and N_2O by their respective warming potential multipliers and converted to mass of carbon to obtain the following annual "CO_2 greenhouse forcing equivalent" emissions: 2.8 Tg C from CH_4, and 0.11 Tg C from N_2O. These amounts are almost half of the 8.9 Tg C from CO_2. Therefore, in Costa Rica, emissions of CH_4 from incomplete combustion during biomass burning has about 30 percent of the warming potential of CO_2 in terms of warming potential. Industrial emissions of CO_2 in Costa Rica in 1991 were roughly 8.87 Tg CO_2-C (WRI 1994).

Tree plantations in tropical regions range in productivity from about 3 to 30 Mg C ha yr^{-1} (Brown et al. 1986) in their first 30 years of growth. Costa Rica would need to plant between about 1000 and 10,000 km^2 of plantations, which amounts to between 2 and 20% of the country's total area, to offset annual deforestation emissions of the trace greenhouse gases N_2O and CH_4 alone. One estimate of tree plantation area in Costa Rica is 400 km^2 in 1990,

and about 37 km² are planted annually (FAO 1993). Thus, in Costa Rica, plantations are only half as extensive as the minimum area needed to offset annual emissions of N_2O and CH_4 from deforestation alone, and additional area would be needed to offset CO_2 emissions.

Alternatively, Costa Rica could focus its efforts on ending deforestation. If that effort were successful, Costa Rica would "save" enough carbon to offset all of its industrial emissions.

Hall et al. (1985) estimated CO_2 emissions from deforestation in Costa Rica from 1950-1980 using a computer program that corrected for the dynamics of decomposition and vegetation regrowth (where that took place). Their biomass estimates were derived from those published in Brown and Lugo (1982), which we mentioned previously, and were stratified into six life zones (5 of which were present in Costa Rica). They did not estimate other trace gas emissions. They estimated release of 4.6 ± 3 Tg C yr⁻¹ due to land use changes in Costa Rica between 1950 and 1980. Later, Hall and Uhlig (1991) revised estimates of biomass and estimated C release using the same dynamic C model. Biomass estimates were also stratified by life zone and were global estimates. Their estimate of C release in Costa Rica from land use change, mainly from deforestation, was 4.67 Tg C yr⁻¹. They also narrowed the range of the previous estimate such that their low estimate was 4.2 Tg C yr⁻¹ and their high estimate was 5.7 Tg C ⁻¹. The sum of all emission products containing carbon, from our study results, produced an estimate of 10 Tg C yr⁻¹. Our estimate of 8.9 Tg C yr⁻¹ from CO_2 alone,
is somewhat larger than these previous estimates.

According to Hall et al. (1985), roughly 20% of the difference between their high and low estimates is due to delays in carbon release from decomposition, plus the carbon absorbed in regrowth, for their more dynamic model. So our estimates, if corrected for this factor, are not very different from Hall et al's (1985) earlier estimates, but appear somewhat higher than his later estimates. Much of the difference is also probably due to the greater geographical specificity of our analysis. Rather than using global estimates of biomass density, we estimated biomass from surveys of Costa Rican forests. As discussed previously, our estimates of Costa Rican lowland wet and moist forest biomass density are about 25% and 60% higher, respectively, than the global averages for equivalent forest types that were reported in Brown and Lugo (1982). Those were the life zones in which the rate of deforestation and carbon release in Costa Rica was highest on average over the time periods investigated. All in all the two estimates are remarkably similar given the difficulty in such analyses. This issue is dealt with using a more sophisticated model in the next chapter.

Errors in estimating greenhouse gas emissions

Because we used only one EF for each forest type, our analysis may either under- or over-estimate trace gas emissions. A more accurate model might use EFs that are time-weighted according to the changing combustion efficiency of an entire burn (Figure 1). In

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addition, combustion efficiency increases as fuel becomes drier. Therefore, the EFs measured during a given tropical forest burn will tend to overestimate emissions from drier forest conditions and underestimate those occurring under wetter forest conditions.

Ideally, a model of emissions from biomass burning would incorporate several additional facets of the process under study, all of which vary spatially. For example, whereas the fuel loads shown in Table 3 are for total aboveground biomass, they are generally higher than the mass that would burn during one cutting and burn event. In reality not all biomass burns during one deforestation burning, and the portion that burns will vary by forest formation as well as land use practices within a given region. For example, Kauffman et al. (1993) and Sampaio et al. (1993) estimated the biomass of slashed Brazilian tropical dry forest known as *Caatinga* to be about 74 Mg ha^{-1}. After the first slash fire, 16.4 Mg ha^{-1} of wood debris remained on site. By contrast, for two primary evergreen forest sites in the Brazilian Amazon, total aboveground fuel biomass ranged from 292 to 435 Mg ha^{-1} (Kauffman et al. 1995). Fires resulted in the consumption of a much smaller portion, 42-57% of aboveground biomass, including >99% of litter and rootmat, but <50% of the coarse wood debris. This delayed oxidation of carbon is one reason that the analysis of Hall et al. (1985) gives a lower estimate than our estimate. The remaining biomass is, however, likely consumed by subsequent burns.

Socioeconomic processes also influence how burning proceeds within a region. The Rio Los Santos Forest Reserve, which is located in the southwestern portion of the Talamancan mountain range, provides an example. In that region, partially burned tree boles which remain in pastures are gradually being converted to charcoal and subsequently burned for cooking (Kappelle and Juarez 1995). Greater emission factors for products of incomplete combustion would be applicable to that biomass.

In addition, the combustion factor and combustion efficiency for a given fire within a particular ecosystem will vary according to wind and moisture conditions. Combustion factors of 78 to 95% were measured by Kauffman et al. (1993), for Brazilian *Caatinga*, and they depended on moisture conditions. Also, combustion efficiency decreases, and thus quantities of trace gas emissions other than CO_2 increase, as a burn event proceeds from flaming to smoldering phases. A model which incorporates that change in combustion efficiency throughout a burn might be more accurate.

Emission factors were not available to estimate trace gas emissions from tropical dry, wet or rain forest life zones, therefore we had to use emission factors measured in a savanna and a moist forest. This aspect adds to uncertainty regarding incomplete combustion product emissions in the wet zones.

Given all the sources of uncertainty that we have described, a more accurate biomass burning and trace gas emissions model would incorporate the following: (1) the successively decreasing amount of original forest biomass which is burned during repeated slash-and-burn events (Kauffman and Stocks 1996), and the spatially-variable burning fate of that biomass,

for example, in Costa Rica stumps and boles still remain in some pastures. These boles are often made into charcoal in high elevation regions (>2000 m, Kappelle and Juarez 1995); (2) pulses of biomass re-accumulation during fallow periods for those regions where slash and burn agriculture still occurs, such as Indigenous Reserves in the eastern portion of the Costa Rican Atlantic slope; (3) a gradual increase in combustion efficiency and fuel consumption as a disturbed site becomes less biomass-dense, and drier; (4) knowledge of trace gas emission factors in very wet and dry forest formations; and (5) knowledge of the portion of biomass first made into charcoal and then burned for fuel; (6) knowledge of the extent of field and pasture abandonment and subsequent secondary forest development; and (7) additional stratification of biomass and deforestation data along soil drainage and fertility gradients.

Interaction between patterns of deforestation and greenhouse gas emissions

By the end of the period of observation for this study, insignificant amounts of the dry forest types and lowland moist forests remained in Costa Rica. In contrast, montane and lower montane rain forests still have more original forest area remaining than has been cleared (Sader and Joyce 1988). As lower elevation regions are saturated, deforestation in humid montane regions became increasingly common in Costa Rica--as elsewhere in tropical America (e.g. Southgate and Basterrechea 1992, Young 1994).

Given the observation that most recent clearing and burning is in wetter tropical forests, the average emissions of incomplete combustion products per area of land deforested may increase globally. The cause for this increase would be the lower combustion efficiency with which wetter forests burn as compared to dry forests, and consequent increase in products of incomplete combustion, such as CH_4. Thus, if tropical deforestation continues there could be an increase in trace GHGs released per km^2 cleared.

CONCLUSIONS

A prominent feature of these results holds implications for future management of greenhouse gas emissions in Costa Rica and other tropical areas. If clearing of the dry and moist forest formations throughout the world has or will shift to clearing of wetter forests, much greater quantities of trace GHGs will be released (Figure 2). Thus the average warming potential from tropical forest burning could increase.

Information gathering should include a thorough evaluation of carbon stores in dead plant materials (litter, organic soil horizons, dead wood). Such information could provide additional insight into an analysis such as this one. The high elevation forests may, in some cases, have thick organic soil horizons and large amounts of downed wood. Therefore the biomass density remaining there might even be greater relative to other forest types than currently appears to be the case. These forests also recover their biomass more slowly than at lower (warmer) elevations (Kappelle et al. 1996).

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Table 2. Emission factors (EF) of greenhouse gases (in kg/kg) for different forest types and for different size classes of wood (From Ward et al. 1992).

Forest Type:	Compound	....EF (Kg/Kg)....	
		$Moist_1$	Dry_2
	$PM-2.5_3$	0.0085	0.0045
	CO_2	1.582	1.713
	CO	0.127	0.063
	NO-N	0.0007	0.0005
	CH_4	0.00805	0.0016
	H_2	0.0036	0.0012

[1] Moist primary forest
[2] very dry, savanna-like (*Cerrado sensu stricto*)

Table 1. Biomass estimation by life zone.

Forest formation	Symbol	Number of sites	Total live aboveground biomass (non-weighted average) [1]	Belowground biomass [2]	Fine litter fraction [3]	Dead wood fraction [3]	Fine litter biomass	Dead wood biomass	Total biomass [4]
			(Mg ha-1)	(Mg ha-1)			(Mg ha-1)	(Mg ha-1)	(Mg ha-1)
Dry, low	T-vdf	2	14	5	0.074	0.034	1.1	0.5	21
Dry, well-developed	T-df	2	188	64	0.018	0.019	3.4	3.5	258
Lowland moist	T-mf	7	518	175	0.013	0.093	6.9	48.3	748
Lowland wet	T-wf	8	365	123	0.011	0.103	3.9	37.5	529
Premontane moist	P-mf	2	208	70	0.018	0.123	3.7	25.5	307
Premontane wet	P-wf	4	306	103	0.011	0.103	3.3	31.5	445
Premontane rain	P-rf	2	318	107	0.011	0.103	5.7	39.0	470
Lower Montane moist	LM-mf	1	319	108	0.018	0.123	5.7	39.1	472
Lower Montane wet	LM-wf	1	421	142	0.017	0.110	7.2	46.1	617
Lower Montane rain	LM-rf	3	324	110	0.017	0.110	5.6	35.5	475
Montane rain	M-rf	1	309	104	0.017	0.110	5.3	33.9	453

[1] Calculated from stand tables in Holdridge et al. (1971) with methodology in Brown et al (1996).

[2] Calculated using equation in Cairns et al. (in review).

[3] Based on data in Delaney et al. (in review), Saldarriaga et al. (1988) and Kauffman et al. (1988).

[4] Aboveground plus belowground plus fine and coarse litter.

Table 2. Trace gas emissions by biomass burning in Costa Rica by life zone.

Forest formation[1]	Total aboveground biomass (live + dead) = total fuel biomass	Forest area cleared 1940-83	Forest area remaining in 1983	Biomass consumption (by clearing/burning of forest 1940-83)	Biomass remaining	PM2.5	CO2	CO	CH4	NO-N	H2
	(Mg ha-1)	(x 10 km²)	(x 10 km²)	Tg	(% of original)	Tg	Tg	Tg	Tg	Tg	Tg
T-vdf	16	21	0	0.34	0	0.002	0.6	0.02	0.001	0.0002	0.0004
T-df	195	21	0	4.2	0	0.033	6.6	0.53	0.033	0.003	0.015
T-mf	573	520	1	298	0	2.3	471	38	2.4	0.21	1.1
T-wf	406	691	282	281	29	2.2	444	36	2.3	0.20	1.010
P-mf	237	85	0	20	0	0.16	32	2.6	0.16	0.014	0.072
P-wf	341	656	224	224	26	1.8	354	28	1.8	0.16	0.805
P-rf	363	195	216	71	53	0.55	112	9	0.57	0.049	0.254
LM-mf	364	0.10	0	0.04	0	0.0003	0.06	0.005	0.0003	0.0000	0.0001
LM-wf	474	39	13	18	25	0.14	29	2.3	0.15	0.013	0.066
LM-rf	365	111	234	41	68	0.32	64	5.2	0.33	0.028	0.15
M-rf	348	35	79	12	70	0.10	19	1.5	0.10	0.008	0.044
Total		2373	1048	968	29	7.6	1531	123	7.8	0.7	3.5
Annual Emissions						0.18	36	2.9	0.18	0.02	0.08
Annual C Emissions							8.9	1.2	0.14		

[1]As in Table 1.
[2]Based on total emissions divided by number of years between 1940 and 1983.

Figure 1. Variation of emission factor for methane by burning stage and moisture of forest formation. Mainly only wet forest formations remained in Costa Rica by 1983 (data from Ward et al. 1992). Combustion efficiency (CE): fraction of biomass carbon emitted as CO_2. Burning stage: F1, F2 - first and second flaming phases, respectively; S1, S2 - first and second smoldering phases, respectively.

Emission Factors for Methane (g CH4/Kg biomass burned)

Forest Type, Biomass and Deforestation

Wet, Intermediate Biomass, Forest Remaining

Moist, High Biomass, > 99% Deforested by 1983

Secondary Forest

Dry, Low Biomass, Deforested by 1961

Burning Stage and Combustion Efficiency
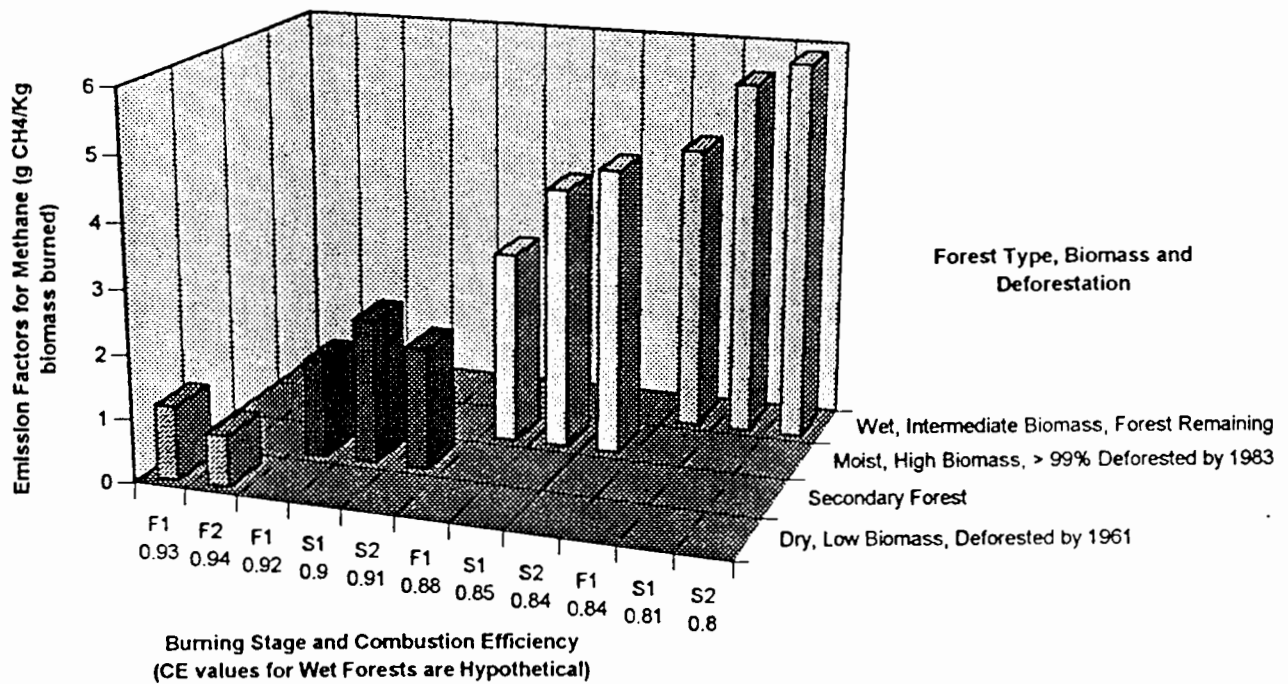(CE values for Wet Forests are Hypothetical)

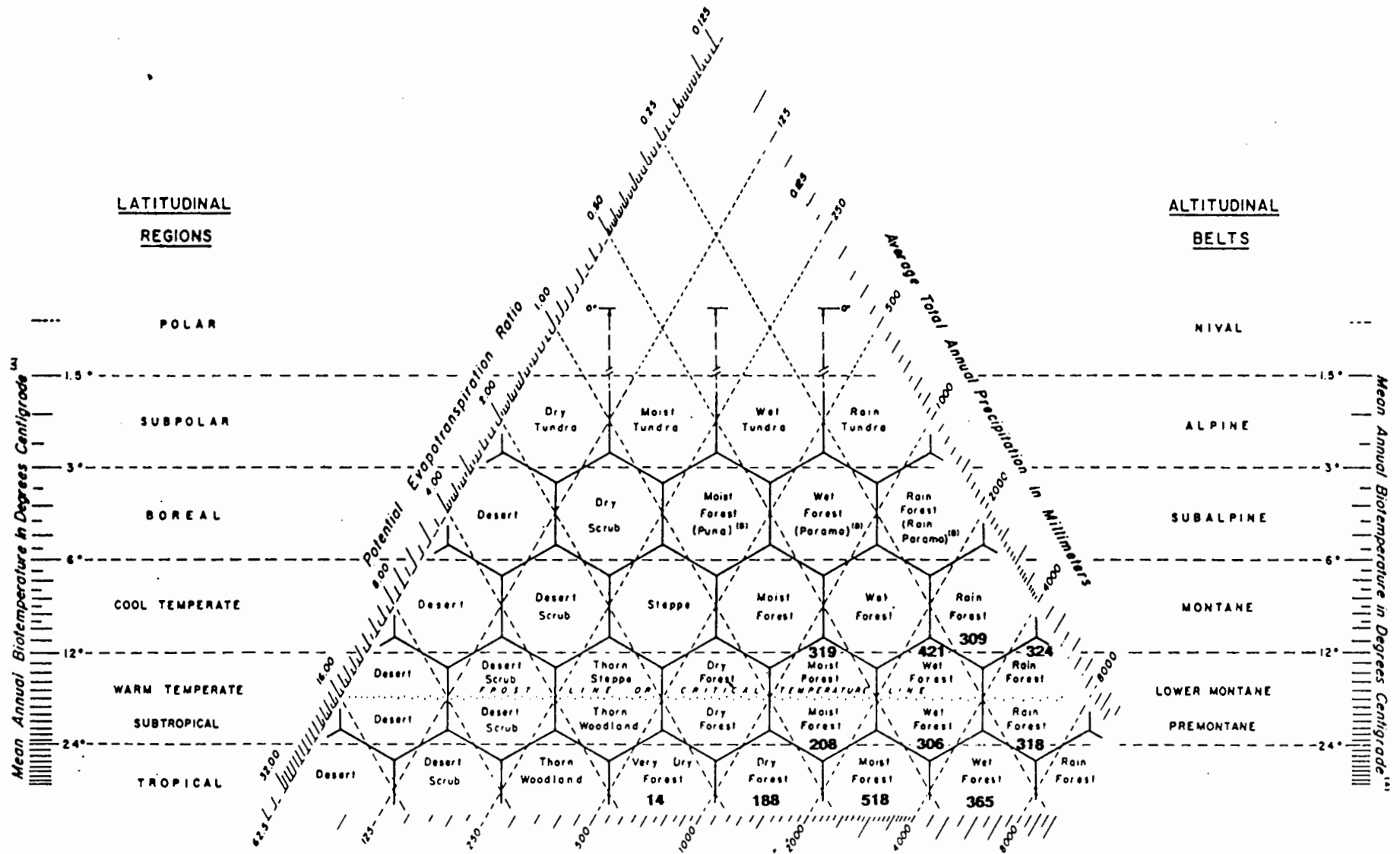Figure 2. Biomass (Mg ha⁻¹ dry weight) plotted along Holdridge (1967) life zone gradients.

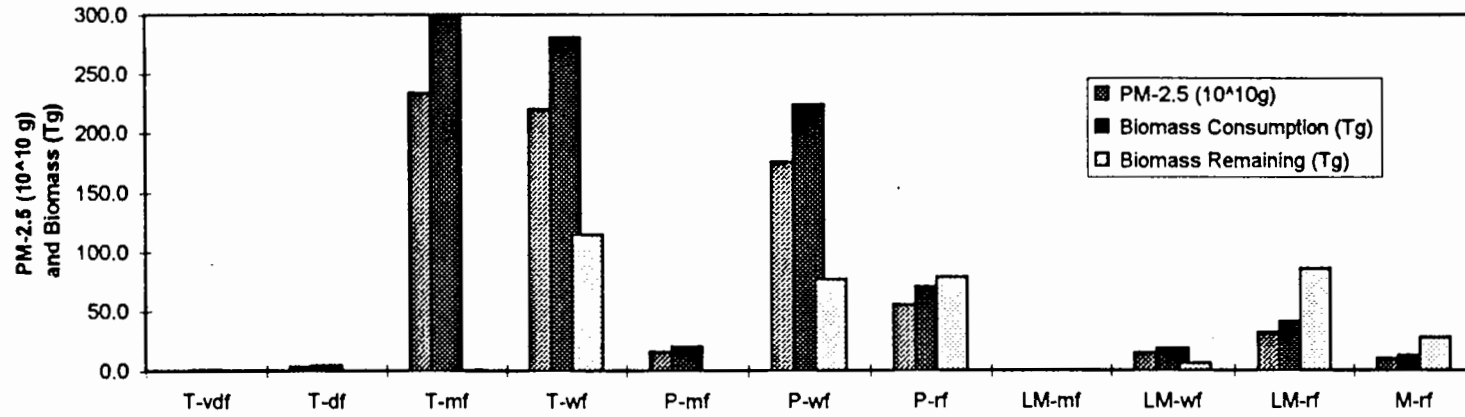Figure 3. Particulate emissions in smoke, biomass burned, and biomass remaining by forest formation.
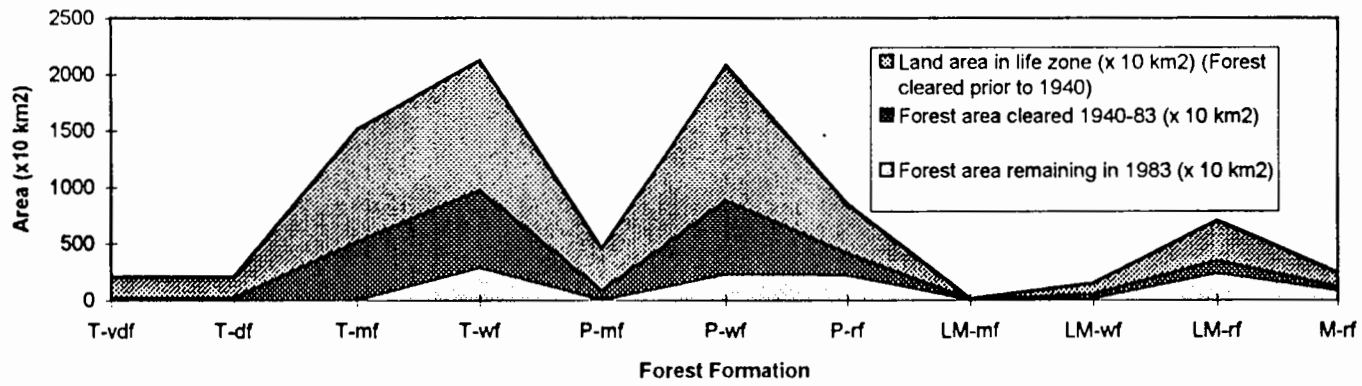
Figure 4. Land area, forest area cleared 1940-83, and forest area remaining, by life zone.

TECHNICAL REPORT DATA
(Please read instructions on the reverse before complet

1. REPORT NO. **EPA/600/A-97/088**	2.	₵

4. TITLE AND SUBTITLE Gradient analysis of biomass in costa rica and an estimate of total emissions of greenhouse gases from biomass burning	5. REPORT DATE
	6. PERFORMING ORGANIZATION CODE

7. AUTHOR(S) E.H. Helmer*, Sandra Brown**	8. PERFORMING ORGANIZATION REPORT NO.

9. PERFORMING ORGANIZATION NAME AND ADDRESS *Department of Forest Science **US. Environmental Protection Agency Oregon State University NHEERL Corvallis, Oregon 97331 200 S.W. 35th Street Corvallis, Oregon 97333	10. PROGRAM ELEMENT NO.
	11. CONTRACT/GRANT NO.

12. SPONSORING AGENCY NAME AND ADDRESS US EPA ENVIRONMENTAL RESEARCH LABORATORY 200 SW 35th Street Corvallis, OR 97333	13. TYPE OF REPORT AND PERIOD COVERED
	14. SPONSORING AGENCY CODE EPA/600/02

15. SUPPLEMENTARY NOTES:

16. Abstract:
One important component of sustainable development for a nation is the degree to which it can balance greenhouse gas (GHG) exchange with the atmosphere. Clearing and burning of tropical forests for conversion to pasture and agriculture, most associated with economic expansion in many tropical countries, release GHGs such as CO_2, CH_4, N_2O, CO, and No_x. Scientists at NHEERL-WED recently estimated the release of such GHGs from the conversion of a range of forest types in Costa Rica between 1940-1983. They also evaluated the influence of environmental gradients that affect the rates and patterns of deforestation and the carbon pools of the forest cleared on GHG emissions. Biomass estimates were derived from a series of previously sampled plots of tree diameters and densities. From these estimates, national inventories of biomass were derived using estimates of the areal extent of forest cover, by life zone, for 1940 and 1983. The change in biomass associated with forest removal between the two dates was used to estimate transfer to the atmosphere of GHGs, using emission factors based on biomass burned, from previously published information. The estimated annual release of carbon-containing compounds over the 43-year period was 8.9 Tg Co_2-C, 1.2 Tg CO-C and 0.14 Tg CH_4-C. The estimated annual N_2O-N release was 1.28×10^{-3} Tg. Based on the relative warming potentials of CH_4 and N_2O, emissions of these gases during biomass burning have contributed about 30 percent of the warming potential of the CO_2 released in Costa Rica. The spatial pattern of GHG emissions reflected the historical pattern of forest clearing in which lowland and dry or mesic forests were developed before wet or high elevation forests. Sixty percent of trace gas emissions from deforestation and burning of the total fuel biomass between 1940 and 1983 was from burning of lowland moist and wet forests. The burning of lowland dry and moist forests contributed respectively 0.5%,31% of emissions. However, these forests comprised only 0.6% and 15% of forest area in 1940 due to earlier deforestation that had started in those regions. In contrast, higher elevation wet forests comprised 14% of forest area in 1940, but together contributed only 7% of emissions. Thus trace gas emissions from deforestation were not proportional to the area of a given forest type. It was concluded that if clearing of the dry and moist forest for nations throughout the world has or will shift to clearing of wetter forests, the average warming potential from tropical forest burning would increase.

17.	KEY WORDS AND DOCUMENT ANALYSIS	
a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group