# CHAPTER 5

# Slash-and-Burn Effects on Carbon Stocks in the Humid Tropics

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## I. Introduction

The relative importance of tropical forests in the global carbon cycle has been debated over the past 20 years with several estimates of their contribution to the increase in atmospheric carbon dioxide (Woodwell et al., 1978; Houghton et al., 1987; Detwiler and Hall, 1988; Hall, 1989; Post et al., 1990). Currently there is general agreement, based on land use change data and atmospheric data, that the tropics are a net source of C to the atmosphere, in the range of 1.1 to 2.1 Pg C $y^{-1}$ (Houghton, 1997). The primary cause of this net source is deforestation in the tropical zone, with Asia and Latin America accounting for over 80% of the flux (Houghton, 1995). Sanchez et al. (1994) estimate total tropical deforestation to be 627 million ha, or approximately 40% of the potential humid forest zone, with 120 million ha of these lands subject to shifting cultivation or slash-and-burn agriculture.

Tropical forests are cleared for a variety of reasons that include logging, establishment of plantations and pastures, and slash-and-burn agriculture. The primary cause of deforestation differs by country and even regions within countries (Tomich and van Noordwijk, 1996) but is usually associated with some form of slash-and-burn agriculture, either as the primary driving force or as a consequence of increased access to forests by logging operations and road construction. Farmers practicing slash-and-burn agriculture are clearing forests to produce food and seek improvements in their families' standards of living. In most cases, they are marginalized from society and government support programs and live in relative poverty. Efforts to reduce deforestation and greenhouse gas emissions resulting from deforestation must address these root causes.

In 1991 a global program, Alternatives to Slash-and-Burn Program (ASB), was initiated to address the agronomic, environmental, social and political implications of slash-and-burn (Brady, 1996). The overall goal of the program was to compare the impact of current land use systems in the tropics and to identify alternatives that were sound from an environmental, agronomic and economic perspective. In addition, policies that currently inhibit the adoption of these alternatives were considered. Teams of national and international scientists were established in key locations, referred to as benchmark areas, around the world representing the range in biophysical and socioeconomic environments in which slash-and-burn is practiced.

The environmental impact of slash-and-burn in terms of net $CO_2$ flux as a result of land use depends on the rates of land use change, the biomass of the vegetation that is cleared and the fate of the carbon within that biomass, the potential for reaccumulation of carbon within subsequent land use systems and the regrowth rates of vegetation. Much of the uncertainty in the values of $CO_2$ flux from the tropics is a result of inadequate estimates for these parameters (Houghton, 1997). One activity of the ASB project was to characterize the patterns of land clearing and subsequent land use at the different sites and to quantify the changes in carbon stocks associated with land clearing and establishment of different land use systems. Standardized methods were established to measure carbon stocks in the forests, the various land use systems established following slash-and-burn clearing, and promising "best-bet" alternatives at the different sites. These data can be used to calculate the immediate and longer term loss of carbon with slash-and-burn clearing and to take carbon stocks into account in multiple goal evaluation of existing and proposed land use systems. In this chapter we present summary data on carbon stocks in forests and slash-and-burn systems from nine of the ASB sites located in Brazil, Cameroon, Indonesia, and Peru.

## II. Estimation of Carbon Stocks in Slash-and-Burn and Alternative Land Uses

Early in ASB activities, a carbon stocks working group was formed among program collaborators and given the responsibility of measuring (1) C stocks in forests undergoing slash-and-burn; (2) dynamics as these forests are converted to current land uses and (3) the potential to sequester C in alternative "best-bet" land uses. An approach was adopted that nests forests and current land uses within chronosequential transects, where short distances in space substitute for relatively great differences in time (Sanchez, 1987).

### A. Institutional Participation and Benchmark Area Selection

Benchmark areas were selected by four national committees in the ICRAF-coordinated global Alternatives to Slash-and-Burn Program (Brady, 1996). The nine benchmark areas reported in this study belong to three floristic zones, the Amazonian, Dipterocarp (S.E. Asia) and Guineo-Congolian (West and Central Africa) Forests (Table 1). The national committees of Brazil, Cameroon, Indonesia and Peru identified a range of benchmark areas that were considered to be typical of current forest conversion by slash-and-burn or deforested lands requiring rehabilitation resulting from past slash-and-burn agriculture (Table 1). Within national research institutions, carbon study research teams were organized and assisted through the development of standardized methods by scientists from ICRAF and the Tropical Soil Biology and Fertility Programme (see Murdiyarso et al., 1994).

### B. Transect Position and Land Use Selection

The exact position of the land use chronosequential transects and the selection of land uses within them was based upon the local knowledge and community contacts of national team members. In principle, land use transformation by slash-and-burn occurs in stages beginning with *primary*, managed or mature secondary forests that are felled and burned, *cultivated* and then alternatively placed into a longer-term land use (e.g., pasture or tree plantation) or *abandoned* to natural succession. Team members would discuss these stages at informal meetings with local chiefs or community leaders and in turn be introduced to individual slash-and-burn farmers willing to host the study. The various land uses, their ages and original forest conditions were discussed with farmers during site visits to candidate transects. Similar texture of the topsoil was used as a criteria to assure that all land-use types within a chronosequence were of the same soil type. Care was taken to exclude transects with nonrepresentative soil conditions or abrupt changes in terrain. During the selection process, several

**Table 1.** Carbon stocks estimated for different land uses within the Alternatives to Slash-and-Burn benchmark areas

| Benchmark area/ coordinates | Location/ lead agency | Comments (No. of land uses, chronosequences) |
|---|---|---|
| Pedro Peixoto 61.7°W, 10.0°S | Acre, Brazil EMBRAPA[a] | Logged semideciduous Amazonian forest occupied by government-sponsored colonist ranchers and farmers since 1973 (7,1) |
| Theobroma 62.1°W, 10.1°S | Rondônia, Brazil EMBRAPA | Logged-over semideciduous Amazonian forest occupied by government-sponsored colonist ranchers and farmers since 1979 (7,2) |
| Ebolowa 11.1°E, 2.5°N | Cameroon IRAD[b] | Unlogged evergreen Guineo-Congolian rain forest occupied by indigenous tribes practicing long-term fallow (6,2) |
| Mbalmayo 11.7°E, 3.5°N | Cameroon IRAD | Logged moist semideciduous Guineo-Congolian forest occupied by indigenous tribes with large areas of young tree fallow (6,2) |
| Yaounde 11.4°E, 4.1°N | Cameroon IRAD | Logged-over, drier, peripheral semi-deciduous Guineo-Congolian forest occupied by indigenous tribes and spontaneous migrants practicing bush fallows and continuous cultivation (6,2) |
| Jambi 102.2°E, .5°S | Sumatra, Indonesia AARD[c] | Logged Dipterocarp evergreen forest concession occupied by nonagricultural indigenous tribes and migrants practicing mixed cultivation and rubber agroforests (6,2) |
| Lampung 108.4°E, 4.5°S | Sumatra, Indonesia AARD | Logged-over Dipterocarp forest occupied by spontaneous migrants cultivating food and market crops under continuous cultivation and agroforests, extensive *Imperata* grasslands (6,2) |
| Pucallpa 74.5°W, 8.2°S | Ucayali, Peru INIA[d] | Logged-over evergreen Amazonian forest settled by migrant ranchers and farmers with close proximity to city markets (7,1) |
| Yurimaguas 76.1°W, 5.8°S | Loreto, Peru INIA | Logged Amazonian rain forest settled by farmers with large areas of tree fallow and poor proximity to markets (7,1) |

[a]Empresa Brasileira de Pesquisa Agropecuaria, Acre and Rondônia; [b]Institut de la Recherche Agronomique pour Development, Nkolbison; [c]Agency for Agricultural Research and Development; and [d]Instituto Nacional de Investigation Agraria.

slash-and-burn land use types were identified, including original forest (slight human impact), managed forest (selectively logged), recently cleared croplands, bush fallow (less than 5 years following clearing), open-canopy tree fallow (5 to 12 years), secondary forest (18 to 25 years), pasture, *Imperata* grassland, young agroforest or improved fallow (often experimental) and mature agroforest or tree plantation. Whenever possible, "best-bets" were included within the land-use chronosequences, but in many cases it was necessary to collect data from experimental stations, development projects, or other farms. Knowledge of original forest condition, establishment date, and management history was prerequisite for inclusion of "best-bets" within this study.

## C. Carbon Pool Measurement

Aboveground carbon was measured for trees, understorey, and surface litter (necromass). Tree diameter measurements were used for estimating tree biomass. Diameter at breast height (DBH) was measured by callipers or diameter tapes and recorded for all trees with diameters greater than 2.5 cm within five quadrates of 4 m x 25 m (Figure 1 A). The positions of the quadrates were assigned by entering well within the individual land use and randomly selecting a direction of the longitudinal axis of the quadrat, and then randomly selecting a new direction in which to place the next quadrat (Figure 1 B). Quadrates were not allowed to "cross-over" one another or to fall outside their intended land use. In Indonesia, part of the data were collected in conjunction with an integrated survey of carbon stocks, biodiversity and greenhouse gas emissions with a sample area of 40 x 5 $m^2$. Tree buttressing was corrected by measuring the diameter above the buttress. For trees branching below breast height, the diameter of all branches was measured separately. Only trees with more than one half of their diameter falling within the quadrates were recorded. Tree biomass was estimated with the allometric equation based on tree diameter of Brown et al. (1989) for moist tropical forests: tree biomass (kg $tree^{-1}$) = 38.4908 - 11.7883*D + 1.1926 * $D^2$ (adj $R^2$=0.78). The biomass of fallen dead logs was measured within the quadrates based on volume (length x cross-sectional area) while assuming a density of 0.4 g $cm^2$. Tree biomass was converted to C by a factor of 0.45.

Understorey biomass, excluding trees with DBH > 2.5 cm, was collected from two 1 m x 1 m subquadrates positioned randomly along the central axis of each 100 $m^2$ quadrat (Figure 1 C). All vegetation occurring within the borders of the quadrat was cut at ground level and collected. Surface litter, including rotting logs and charcoal, was collected within a 50 cm x 50 cm frame centrally placed within each subquadrat (Figure 1 C). Samples were weighed, subsampled, oven dried at 65° C to constant weight and corrected for moisture content. Live vegetation was assumed to contain 45% C on a dry weight basis and surface litter was ground and analyzed for total organic carbon (Anderson and Ingram, 1993).

A soil and root sample was recovered from an area 20 cm x 20 cm within each subquadrat. The original guidelines recommended excavation to a depth of 40 cm, at 20 cm intervals, but some cooperators chose shallower (15 cm in Indonesia) or greater depths (50 cm in Brazil). Care was taken to recover as much roots as possible during the excavation except in Indonesia where root biomass was not measured. Also during excavation, bulk density measurements were taken at 10 and 30 cm by use of 100 $cm^3$ rings. All soil and roots from the hole were placed in bags and transported to the laboratory. A subsample was taken for total C analysis. The remaining sample was dispersed in water and passed through a 2 mm sieve; roots were collected from the sieve and washed in water without distinguishing live and dead roots. Roots were oven dried at 65° C to constant weight, weighed, ground and ashed. Ash-corrected dry weight was assumed to contain 0.45% C.

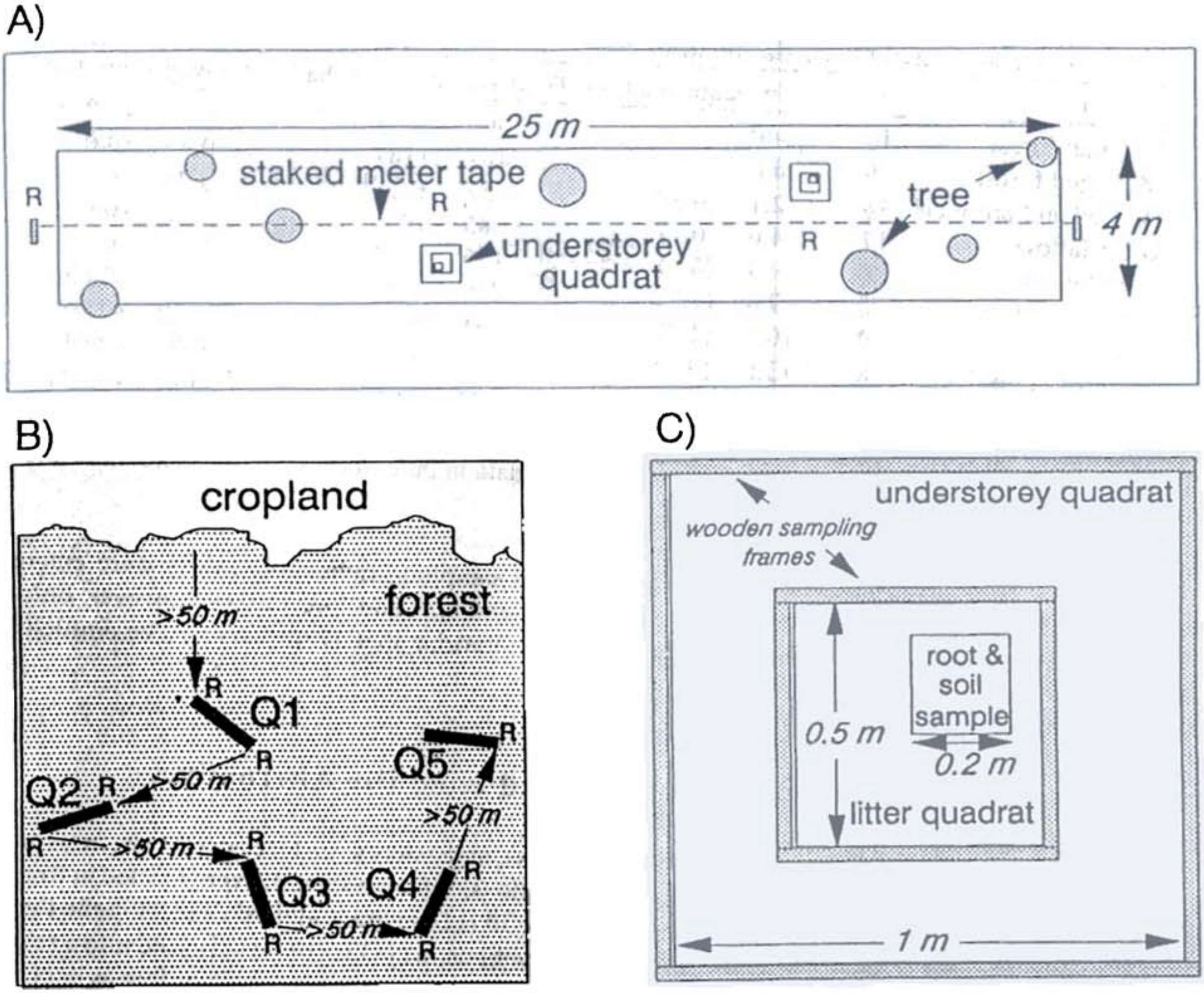


**Figure 1.** Carbon stock estimation of woody biomass based on five 100 $m^2$ quadrates (A) randomly positioned within land uses (B) with understorey, litter and soil measurements collected within sub-quadrates (C).

### D. Data Compilation

Additional information on each land use included its chronosequence, benchmark area, geographic coordinates, duration, interval since forest clearing (chronosequential age), soil sampling depth and bulk density. The carbon stock measurements were tree, understorey, surface litter (necromass), root and soil C as t C $ha^{-1}$. Combined variables were generated including total system C, total above- and below-ground C, relative C stocks with respect to initial forest and proportion of aboveground and soil C. The data were imported into a computer software program, sorted by land use and summary statistics generated. Chronosequences were sorted into types (omitting initial forest C) and carbon sequestration calculated by linear regression. In compiling total system carbon from individual C

**Table 2.** Chronosequential age, total system C and proportion of aboveground carbon in tropical forests and lands converted by slash-and-burn

| Land use | n | Age in sequence (yr) | | Total system C (t C $ha^{-1}$) | | Aboveground:total C | |
|---|---|---|---|---|---|---|---|
| Original forest | 10 | n/a[a] | | 305 | (23) | 0.72 | (0.04) |
| Managed forest | 9 | n/a | | 181 | (18) | 0.73 | (0.05) |
| Burned and cropped | 18 | 2.1 | (0.3)[b] | 52 | (7) | 0.23 | (0.07) |
| Bush fallow | 17 | 4.6 | (0.2) | 85 | (9) | 0.22 | (0.06) |
| Tree fallow | 8 | 9.4 | (0.3) | 136 | (16) | 0.48 | (0.08) |
| Secondary forest | 8 | 19.4 | (2.2) | 219 | (18) | 0.61 | (0.03) |
| Pasture | 9 | 10.0 | (1.2) | 48 | (11) | 0.20 | (0.06) |
| *Imperata* grassland | 8 | 13.0 | (2.0) | 47 | (6) | 0.05 | (0.01) |
| Young agroforest | 10 | 5.0 | (0.7) | 65 | (10) | 0.28 | (0.04) |
| Mature agroforest | 19 | 23.1 | (1.6) | 130 | (11) | 0.58 | (0.04) |

[a]Not applicable, sequences begin at forest clearing; [b]data in parentheses denote standard errors.

pools, no distinction was given to the depth of soil sampling, rather soil values were entered as provided. When soil data were compared between land uses and forest types, however, only data reported for 0 to 15 and 0 to 20 cm were considered requiring that 31 of 116 cases be omitted from analysis.

## III. Carbon Stocks in Slash-and-Burn and Land Use Alternatives

Estimates of carbon stocks residing in woody biomass, understorey, surface necromass, roots and soil were collected for 116 sites within 9 benchmark locations. Of these sites, 30 were located in Rondônia and Acre, Brazil, 35 in the forest zone of Cameroon, 30 in Sumatra, Indonesia and 21 in the Peruvian Amazon. Current land uses accounted for 85 observations, with 33% of these placed into a sequence of natural fallow succession, 9% agroforests, 6% pastures and the remainder either forests or cropland (Table 2). "Best-bet" candidates accounted for 31 observations with 68% classified as agroforests, 16% improved fallows, 13% improved pastures and 3% improved managed forests. Because some "best-bet" alternatives in one benchmark site closely resemble current practices at others, no distinction was drawn between them when total system carbon estimates are compared between land uses across benchmark sites.

Carbon stock estimates were grouped by land use and the duration since forest clearance, total system carbon and the proportion of carbon residing aboveground calculated (Table 2). No age was calculated for original or managed (disturbed) forests because the sequence was assumed to begin with forest clearing. When compared by pairwise Tukey *t*-tests, five classes of carbon stocks emerged. Carbon stocks were significantly greater in original than in secondary ($p = 0.003$) and managed forests ($p < 0.001$). Managed forests, mature agroforests and tree fallows did not significantly differ, nor did tree fallows and bush fallows, or bush fallows from young agroforests, croplands, pastures and *Imperata* grasslands. Carbon dynamics may be inferred from the proportion of carbon residing aboveground (Table 2). Forests and mature agroforests contain greater than 50% of C stocks aboveground, while croplands, grasslands, recovering fallows and establishing agroforests contain less than half.

The forest system C of the original forests (Table 3) contained approximately 200 t aboveground C $ha^{-1}$ (data not presented) in Cameroon and the Amazon. This estimate appears large compared to

Table 3. Total system carbon in original and managed forest systems

| Forest zone | Total system C Original | Managed | Comments on management |
|---|---|---|---|
| | t C $ha^{-1}$ | | |
| Amazonian | 256 (22)[a] | 197 (9) | Selectively logged by settlers |
| Dipterocarp | 433 (n/a)[b] | 143 (17) | Logging concessions and agroforests |
| Guineo-Congolian | 308 (26) | 179 (14) | Mature cacao agroforests[c] |
| Overall | 305 (23) | 166 (11) | |

[a]Standard errors in parentheses; [b]based upon single observation; [c]trees are selectively felled and cacao understorey established.

140 t C $ha^{-1}$ (310 t DM $ha^{-1}$) reported by FAO (1997) for the same areas estimated by forest inventory methods. However, when the estimates obtained by forest inventory are corrected for trees less than 10 cm DBH (an additional 5%, data not presented), understorey (an additional 3%), and litter layer (an additional 10%, see Figure 2), the value is adjusted to 165 t C $ha^{-1}$. The original quadratic allometric equation of Brown et al. (1989) for estimating tree biomass that was employed in this study also gives relatively higher values than the more recently derived power function of FAO (1997). Use of the new equation with data from our study gives tree biomass estimates 70 to 95% that of the former equation, bringing the aboveground estimates within the range of those reported above by forest inventory methods. The single estimate for aboveground C in primary Dipterocarp forests of Indonesia in our study, however, is extremely high compared to the 126 to 182 t C $ha^{-1}$ reported by FAO (1997) for Malaysia. Given the relatively good agreement between aboveground C estimates using our fairly rapid methodology with the estimates from forest inventories, we recommend this method for measuring the C stocks of the vegetation in slash-and-burn areas. It must be noted that the specific allometric equation employed to convert tree dimensions to biomass affects results depending upon the size distribution of trees.

The impact of forest management, other than slash-and-burn, is presented in Table 3. The Amazonian forests were selectively logged by settlers, often with land title (Brazil). The forests in Sumatra were either in the process of logging by large concession (Jambi) or were extensively logged in the past (Lampung). Forest management was heterogeneous in Cameroon with active forest concessions in Mbalmayo and extending toward Ebolowa. Another feature of forest management in Cameroon is the cacao (*Theobroma cacao* L.) forests, where trees are selectively felled and cacao planted as an understorey. This land use is considered as mature agroforests in other tables and figures. Overall, forest management reduced original forest carbon stocks by 46% with greatest losses observed from mechanized logging operations.

It is from these disturbed or logged forests that most slash-and-burn clearing is occuring, the logging practice itself reducing the carbon stocks by more than half. Cropping or pastures further reduce C stocks to 18% and 15%, respectively, that of the orginal forest, or 29% of the disturbed forest.

Soil organic carbon stocks in the 0–15 and 0–20 cm soil layers in different land use categories and forest zones are presented in Table 4. The average of 43 t C $ha^{-1}$ in the top 15–20 cm of soil in the forest ecosystems (Table 4) is lower than the range of 46 to 69 t C $ha^{-1}$ reported by Detwiler (1986),

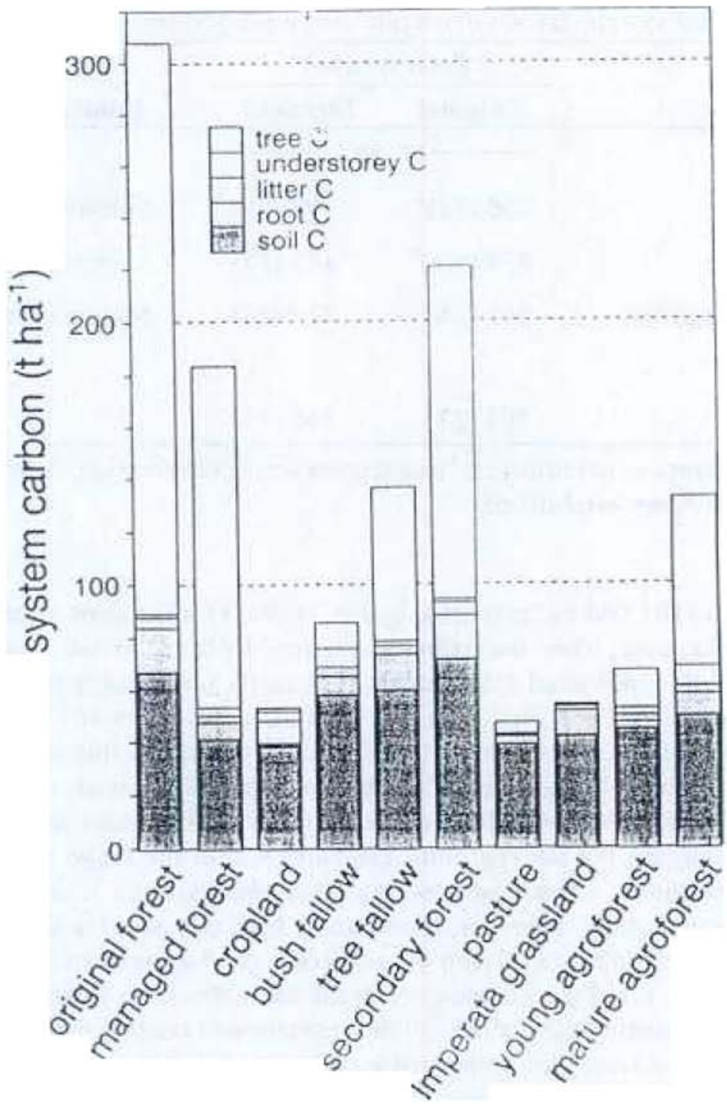


**Figure 2.** Carbon stocks in 10 slash-and-burn and alternative land uses.

assuming that 45% of the carbon in a 1 m profile is found in the top 20 cm (Moraes et al., 1995). The value is within the range found by Moraes et al. (1995) for undisturbed forests in the Amazon Basin of Brazil. The extremely low values of 31 t C ha$^{-1}$ for the Amazon forests of our study cannot be explained, especially since the forest values are lower than the other land use practices.

Overall, soil C stocks were greatest in forests and agroforests and less in crops, bush fallows and grasslands. Significant differences in soil C with land use was not observed in Cameroon (Kotto-Same et al., 1997), probably reflecting the less intensive use of land in this benchmark area. The topsoil lost 13 to 39% of its carbon during the cropping phase in Cameroon and Indonesia, the larger losses in Indonesia perhaps reflecting the more intensive land use in the colonization areas compared to the traditional slash-and-burn systems in Cameroon. Detwiler (1986) reported averages losses of 40% of the soil carbon in the top 40 cm with cropping. Grasslands, including pastures, lost 21% of the carbon in the topsoil, similar to the 20% loss in pastures reported by Detwiler (1986).

**Table 4.** Soil organic carbon in different slash-and-burn land uses and forest zones (0–15 or –20 cm[a])

| Land use | Forest zone: Amazonian | Dipterocarp | Guineo-Congolian | Mean | Tukey *t*-test (P) |
|---|---|---|---|---|---|
| | t soil C ha$^{-1}$ | | | | |
| Forests[b] | 30.9 | 48.1 | 42.8 | 42.7 | 0.03 |
| Crops and bush[c] | 38.8 | 29.5 | 37.2 | 35.3 | n.s. |
| Agroforests[d] | 22.5 | 46.7 | 43.2 | 39.8 | 0.02 |
| Grassland[e] | 29.6 | 36.8 | – | 33.2 | n.s. |
| Mean | 31.2 | 40.5 | 41.1 | 38.4 | 0.01 |
| Tukey *t*-test (P)[f] | 0.01 | 0.04 | n.s. | 0.07 | |

[a]Soil sampling depths are consistent within forest zone; [b]forests include original and secondary forests and tree fallows; [c]crops and bush include all burned and cropped lands and bush fallows; [d]agroforests include all young and mature agroforests; [e]grasslands include all pastures and *Imperata* grasslands; [f]probabilities assigned through Tukey Highest Significant Difference *t*-test.

The allocation of carbon between woody biomass, understorey, litter, roots and soil is presented in Figure 2. Soil C represents 13% of the forest system carbon and increases to 68% in the cropping systems. A large proportion of system carbon occurs within woody biomass in forests, tree fallows and agroforests and is nearly absent in croplands, pastures and *Imperata* grasslands. Croplands contain higher amounts of litter than do other land uses, which may be largely attributed to fallen and partly combusted woody residues. *Imperata* grasslands contain a large proportion of root biomass. Tree roots are likely to be seriously underestimated by our methods, particularly deeper, structural roots. The shoot:root ratios we obtained were much higher than those reported by Sanford and Cuevas (1996) in fewer but more intensively studied sites.

## IV. Carbon Dynamics within Tropical Forest Land Uses

The most rapid loss of system carbon results from felling and burning original and managed forests, with an average loss of 120 t C ha$^{-1}$ until the end of cropping or 58.3 t C ha$^{-1}$ yr$^{-1}$ for a period of 2.1 years (Table 2, Figure 3). Difficulties were encountered in quantifying carbon loss from slash-and-burn during a single year beginning with forest disturbance. On one hand, farmers do not always clear forests during a given year, on the other, burning continues over several years as remnant trees are felled and burned along with remaining woody litter. Pasture establishment in the Brazilian Amazon is often based upon a two stage burning strategy where forests are cut and burned, seeded while land returns to fallow for 2 to 4 years, then cut and burned again resulting in near-complete stands of fire resistant grasses. In general, 80% of the system carbon is lost during the clearing and cropping phase.

Land use following the cropping phase can result in increased carbon losses or carbon sequestration. Following crop abandonement, carbon sequestration in the vegetation and soils of natural fallow succession was 7.9 t C ha$^{-1}$ yr$^{-1}$ (Figure 4) and was least in Amazonian bush fallow and greatest in the tree fallows and secondary forests in Cameroon (Table 5). Carbon sequestration rates of various fallow types could not be calculated for all forest zones, owing to the scarcity of older

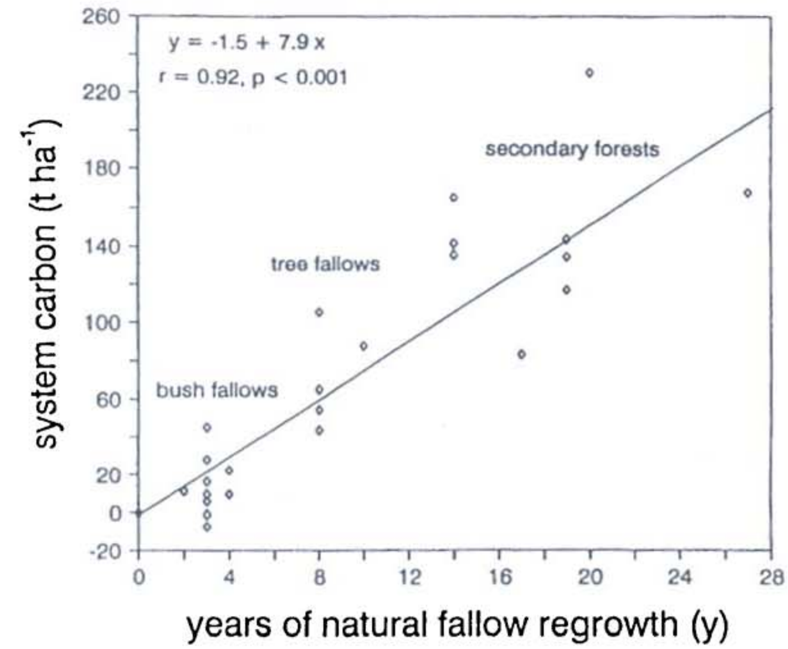


**Figure 3.** Carbon loss due to forest conversion and recovery in natural fallows and agroforestry systems: all observations plotted.

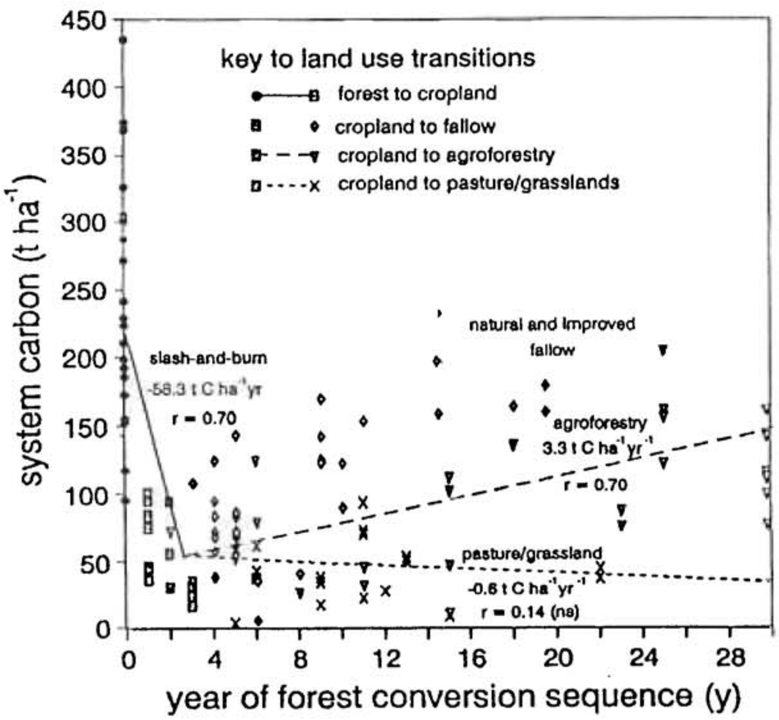


**Figure 4.** Total system carbon measured along slash-and-burn land use chronosequences in Brazil, Cameroon, Indonesia, and Peru.

**Table 5.** Carbon sequestration in natural fallows and secondary forests

| Forest zone | | | |
|---|---|---|---|
| | | t C ha$^{-1}$ y | |
| Amazonian | 3.9 .0)[a] | | 6.2 (1.3) |
| Dipterocarp | | | 6.2 (n/a)[b] |
| Guineo-Congolian | 2.6) | 8.5 3) | 9.3 (0.9) |
| Overall | 4.6 (1.6) | 8.5 (1.3) | 8.3 (0.8) |

[a]Standard errors in parentheses; [b]based upon single observation

fallows in many locations. These carbon sequestration rates are less in burn fallows and fall within the range of 2 to 9 t C ha$^{-1}$ y$^{-1}$ in vegetation and litter reported by Szott et al. (1994). Houghton (1997) reported lower values of 2 to 5 t C ha$^{-1}$ y$^{-1}$. Our study found the recovery of carbon in the soil to be 0.2 t C ha$^{-1}$ y$^{-1}$ at a maximim. The relatively high C recovery rates measured in the benchmark sites may help substantiate the claim that regrowth rates in the tropics may be higher than previously estimated (Houghton, 1995) or could be simply be a function of using allometric equations for estimating biomass that were developed for mature forests rather than young secondary forests. Planted tree fallows do not seem to increase carbon sequestration rates above that of the natural fallow (Szott et al., 1994) but might do so in cases where seed banks of trees have been depleted, such as the case of pastures (Uhl et al., 1988). Agroforests sequestered carbon at a lower rate than do natural fallows (3.3 t C ha$^{-1}$ yr$^{-1}$, $r = 0.70$). Small amounts of carbon loss continued during the cropland to pasture or *Imperata* grassland sequence (Figure 3).

## V. "Bad Bets" and "Best Bets"

Traditional slash-and-burn agriculture as practiced by sparse indigenous populations in large forests and nonmarket settings does not result in large-scale or long-term environmental damage but rather may be viewed as another source of patch dynamics (Kotto-Same et al., 1997). But this form of slash-and-burn was not encountered during our investigations, its closest resemblance being the fallow regeneration practiced by indigenous tribes with poor access to markets in Mbalmayo and Ebolowa, Cameroon. Yet even these farmers have created a rapidly retreating forest margin and reduced fallow intervals. Forest destruction is massive when settlers with poor knowledge of forest resource utilization (Fujisaka et al., 1996) migrate to humid forests and prepare lands for permanent utilization. From the environmental standpoint, attempts to mine system nutrients for annual food crop production or pasture establishment without regard to the need for longer-term input management are poor land use alternatives. The invasion of *Imperata* species into transmigration areas of Indonesia and degraded pastures of Brazil may be regarded as symptomatic expression of poor land management. Another example of poor management of humid forest resources is exhaustive logging where standards for acceptable extraction are progressively lowered until only ill-formed or very small trees remain, as was noted in Jambi and Pucallpa. But the documentation of carbon loss from deforestation "horror stories" was not the intent of our studies, rather we sought to identify opportunities to reduce the deleterious environmental impacts within the tropical forests undergoing rapid transition.

### A. Current Forestry and Agroforestry

Owing to the large proportion of system C residing in trees of the humid forests (Figure 2), the obvious opportunities to conserve or sequester carbon involve the protection or re-establishment of trees. In many cases, these involve current management practices, as with the jungle rubber (*Hevea brasiliensis* Muell.-Arg.) agroforestry in Indonesia which contains up to 147 C $ha^{-1}$ after 30 years or forest cacao management in Cameroon with 179 t C $ha^{-1}$, these systems have remained over many decades. The economic viability of those pursuits dictate the future of carbon storage within those land uses. In the same manner, tropical forest plantations have potential for significant and renewable carbon storage. Although an *Acacia mangium* plantation in Sumatra was projected to contain 200 t C $ha^{-1}$, over half that of the original forest 20 years later, these production systems cannot meet the short-term needs of growing and migrating populations of smallholders under current policy environments (Tomich et al., 1997). The economies of many lesser developed nations are dependent upon forest resource utilization, and expectations by more developed nations that tropical forests be set aside for future ecotourists are not realistic, nor consistent with the historical utilization of forests in developed countries.

The potential of more simple agroforests, as compared to complex agroforests such as jungle rubber, to sequester carbon is less well known because of the relatively young age of many of these agroforestry systems. The young (less than 10 y) simple agroforests measured in this study had an average of 65 t C $ha^{-1}$, accumlating about 1.5 t C $ha^{-1}$ $y^{-1}$. Dixon (1995) reports carbon stocks of 12 to 228 t C $ha^{-1}$ for a variety of agroforestry, including silvopastoral, systems in the tropics, the large range depending on the complexity and age of the systems.

### B. Deflection from Deforestation

Sanchez (1990) described the socio-economic effects likely to result from widespread tropical deforestation where landless poor with inadequate land management skills and inputs migrate to tropical forests, and speculated that each hectare placed into permanent, profitable agriculture has potential to offset deforestation of 5 to 10 additional hectares by shifting cultivators. He also noted the important role of national policies in deforestation, particularly through government-sponsored settlement programs. The concept of deflection from deforestation through agricultural intensification remains unproven and a provocative issue because of the likelihood that landless poor would "stream" toward new land and agricultural opportunities (Tomich and van Noordwijk, 1996). Harwood (1996) describes the need for capital and technical inputs required within agricultural intensification, suggesting that these remain a barrier to agricultural change unless production problems within current practices arise. Agroforestry systems offer potential for carbon sequestration (Figures 2 and 3) but do these systems deflect from deforestation? In Cameroon, farmers adopted cacao agroforestry but continue to clear forest margins to cultivate annual crops for food and market (Kotto-Same, 1997). Farmers in Theobroma who participated in the carbon dynamics studies were converting degraded pastures to fruit orchards but continuing to clear additional forest for pasture. It may be naive to consider that a new land management will completely substitute for another, rather it will be adapted within the slash-and-burn farming system, or that new agricultural opportunities are sufficient to eliminate the perceived need to clear forests. Another approach to deflection is the requirement that settlers in Theobroma and Pedro Peixoto (Brazil) clear only 50% of their land and manage the remaining forest. This principle is seldom effective because migrant farmers often lack forest management skills, land derives value through clearance and the 50% clearance requirement is seldom enforced (Fujisaka et al., 1996). An important criterion in comparing the environmental benefits of various "best bets" is their potential for deflection; however, criteria for determining deflection must be better defined.

## VI. Extrapolation of System C Measurements

The carbon estimates of different land uses in sites of active deforestation have value within themselves in comparing the environmental benefits of different land use alternatives, but may also be used in geographic applications and to validate output of simulation models. Kotto-Same et al. (1997) describe an analysis using the Cameroon data set in which system carbon estimates were substituted into mapping units which had changed from tropical forest to other land uses between 1973 and 1988. This approach indicated that 202 Mt C were lost from 20,000 km$^{-2}$ deforested during 15 years. Similar data extrapolation is being undertaken with data sets from the other benchmark areas.

The carbon estimates may also validate environmental simulations generated by the CENTURY Model (see Parton et al., 1987, 1994). Sitompul et al. (1996) simulated the effects of contrasting land management following forest disturbance in an Ultisol at Lampung including natural fallow regrowth, conversion to sugarcane (*Saccharum* cv. L.) and current rice cultivation practices. Natural fallow rapidly reestablished soil organic C at 50 t C ha$^{-1}$, but all crop management strategies resulted in continued SOC loss over 25 years, with most loss occurring from the lighter fractions (SOM1C and SOM2C). Sugarcane management resulted in 11% less loss in SOC than did current farmer practices.

Output from CENTURY version 4.0 (Metherell et al., 1993) is presented in Figure 5 where two alternative land managements are compared to an expected scenario of slash-and-burn followed by bush fallow and continuous maize (*Zea mays* L.) cultivation. When basic conservation measures, such as protection of economically important trees, felling along contours to reduce soil erosion and organic inputs to soils are practiced (Kotto-Same et al., 1997), total system C losses are reduced by 24% over 53 years. In contrast, establishment of rubber agroforestry increases the 53 year average total system C by 99 t C ha$^{-1}$ over current practices. The agroforestry scenario may be overestimated as many rubber plantations must be replanted in less than 53 years.

## VII. Carbon Dynamics and National Agendas

One of the benefits of this research is to raise the profile of carbon dynamics studies within national research structures. From its inception, the Alternatives to Slash-and-Burn Programme was based upon partnership between international and national research institutes (Brady, 1996) with critical decisions on study locations, land use selection and "best-bet" alternatives as the responsibilities of the lead national agency at each benchmark area (Table 1). In Brazil and Cameroon, lead agencies had little past experience in carbon dynamics measurement and were assisted by national universities and international partners, particularly the Tropical Soil Biology and Fertility Programme and the International Centre for Research in Agroforestry. In Peru, national partners included teams with strong background in forestry, but required assistance in selecting representative land uses within the slash-and-burn chronosequence. Indonesia initiated its carbon studies by sponsoring a workshop attended by representatives from several national institutions and universities where experiences were shared and research plans formalized (Murdiyarso et al., 1994).

Mid-way through these investigations, all national teams were conducting autonomous studies at sites and within land uses of their choice. With this autonomy, some sacrifices in cross-site comparability were experienced as some teams chose to modify soil sampling methods. This sacrifice seems small by comparison to the strengthening of global change agendas within the national structures of countries most affected by tropical deforestation.

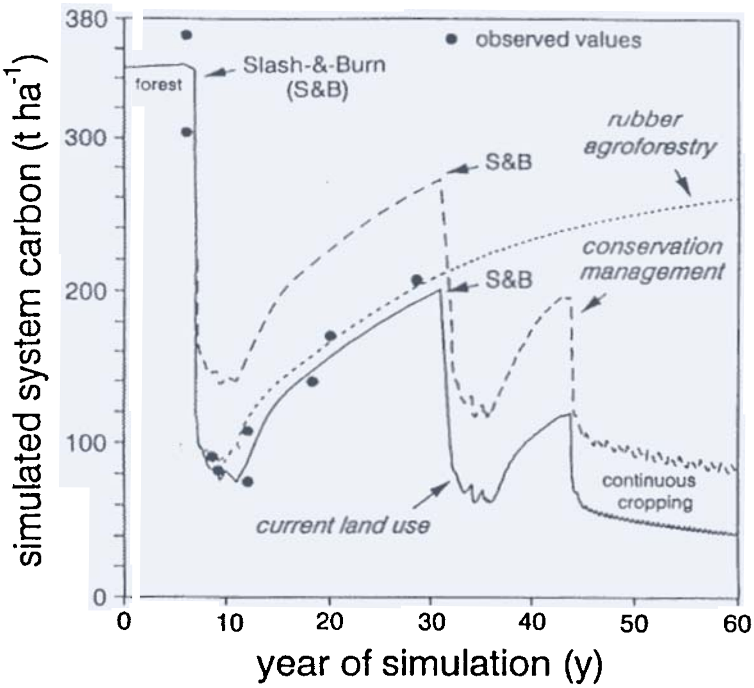


**Figure 5.** A CENTURY Model simulation of current and alternative land management practices based upon data from Ebolowa, Cameroon.

## VIII. Assessment and Complementarity of Rapid Carbon Estimation Procedures

One criticism open to the relatively rapid carbon estimation methods developed for this study is the relative importance of tree biomass (Figure 2) compared to the area and number of trees measured (Figure 1). The allometric equations used for estimating tree biomass were developed from mature forests while many of the systems studied were young secondary forests and agroforests. There is perhaps a need to develop equations for young secondary forests and specific equations for many of the common agroforestry tree species. FAO (1997) recommend power equations for dry, moist and wet forests that may be better suited for calculating the biomass of smaller diameter trees.

No predetermined method of main quadrat alignment (randomization) was found acceptable as this would often lead the team outside of its intended land use or into poorly representative terrain. This led to our selection of field randomization of quadrat direction by "blind-spin-and-toss". In practicing this form of randomization, one must be careful not to attempt targeting larger trees as these often account for a large proportion of woody forest biomass, particularly after previous tosses have established quadrates that contain few trees. In some cases, teams were subject to additional pressures by onlookers who criticized them for failing to measure the largest trees in their forests. In one case, a local chieftain in Ebolowa, Cameroon, insisted that the largest trees in his forest be measured by the research team, and this was done but without entering these data into further analysis. Another source of field confusion was disagreement between local experts and/or farmers in distinguishing primary, managed and mature forests. In general, land managers with clear land tenure (Brazil and Indonesia) appeared to have more concrete opinions on land history, occasionally consulting farm records. In areas with traditional land tenure (Cameroon) or where land operations were previously interrupted by civil unrest (Peru), land histories tended to be more controversial and greater importance was placed upon the insight of the research team.

The carbon dynamics studies reported in this chapter are but one component of the international research agenda of the Alternatives to Slash-and-Burn Programme. Other activities include monitoring greenhouse gases (GHG), and the assessment of above- and below-ground biodiversity. In many cases, carbon dynamics (CD) studies preceded these other investigations and many of the chronosequences and land uses identified by the carbon dynamics team were later characterized in terms of GHG and biodiversity. Some difficulties were encountered in complete interchange of sites between research themes. Aboveground biodiversity (AGB) was assessed by combining plant species with indices of plant functional attributes (Kenyatta, 1997). The minimum sample area deemed necessary for this approach was larger than that for C estimation and many transitional land uses could not be measured. Below-ground biodiversity (BGB) teams focused their attention on the population sizes and diversity with five functional groups of soil organisms; macrofauna, nematodes, rhizobia, mycorrhizae and decomposer communities. These procedures were too time and material intensive to measure all of the sites characterized by the carbon team. Nonetheless, plans are underway to combine the data obtained by the four research groups (CD, GHG, AGB and BGB) into the fuller context of environmental impacts of current slash-and-burn and alternative land uses, and to compare these impacts to the potential of various land uses to alleviate poverty and to deflect from future deforestation.

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