

Land-Use Change and Carbon Flux Between 1970s and 1990s in Central Highlands of Chiapas, Mexico

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ABSTRACT / We present results of a study in an intensively impacted and highly fragmented landscape in which we apply field-measured carbon (C) density values to land-use/land-cover (LU/LC) statistics to estimate the flux of C between terrestrial ecosystems and the atmosphere from the 1970s and 1990s. Carbon densities were assigned to common LU/LC classes on vegetation maps produced by Mexican governmental organizations and, by differencing areas and C pools, net C flux was calculated from the central highlands of Chiapas, Mexico, during a 16-year period. The total area of closed forests was reduced by half while degraded and fragmented forests expanded 56% and cultivated land and pasture areas increased by 8% and 30%, respectively. Total mean C densities ranged from a high of 504 tons C/ha in the oak and evergreen cloud forests class to a low of 147 tons C/ha in the pasture class. The differences in total C densities among the various LU/LC classes were due to changes in biomass while soil organic matter C remained similar. We estimate that a total of 19.99×10^6 tons C were released to the atmosphere during the period of time covered by our study, equal to approximately 34% of the 1975 vegetation C pool. The Chiapas highlands, while comprising just 0.3% of Mexico's surface area, contributed 3% of the net national C emissions.

Forest ecosystems contain large amounts of carbon (C), both in living and dead biomass and in soil (Post and others 1982). Recent estimates place the current global terrestrial biomass C pool at 550×10^9 tons and the soil C pool at 1500×10^9 tons (Siegenthaler and Sarmiento 1993). Comparable pre-industrial figures are 610×10^9 and 1560×10^9 tons (Siegenthaler and Sarmiento 1993). Forests account for the largest fractions of these terrestrial pools.

In tropical latitudes high-biomass forest ecosystems are converted to systems of lower biomass such as secondary forests, cultivated lands, and pastures (FAO 1993). The type and successional stage of ecosystems

occurring within a landscape, resulting from these conversions, have profound implications for regional biochemical flux and atmosphere-biosphere interactions. However, surprisingly little is known about: (1) the precise extent of different tropical forest types, (2) the relative distribution of primary versus secondary forests, (3) the amount of C stored by each type, (4) the rate at which each type is lost or degraded, and (5) the fraction of stored C lost during the process (Detwiler and Hall 1988; Hall and Uhlig 1991; Houghton and others 1991). Thus estimates of the total CO₂ release from biomass reductions and land clearing vary between 8% and 44% of total anthropogenic CO₂ emissions (Cook and others 1990; Schimel 1995). Additionally, little is known about the complex dynamics of social, economic, and political forces, which together promote deforestation and land-use change processes, especially in developing countries (Lambin 1994).

KEY WORDS: Land use; Land cover; Carbon flux; Forests; Chiapas highlands; Mexico

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The highlands of Chiapas display land-use patterns that can be recognized in many montane settings in Latin America, such as Guatemala, Peru, Colombia, and Ecuador, as well as other areas of Mexico. These areas contain established, traditionally managed agricultural systems but are subject to processes of deforestation and land degradation due to increasing population pressure without economic alternatives to increase sustainable production (Bremer and others 1984). Estimated deforestation rates in the Chiapas highlands ranged from 3.23% (1974–1984) to 3.58% (1984–1990), for only closed forests, and from 1.58% (1974–1984) to 2.13% (1984–1990) for open and closed forests combined (Ochoa-Gaona and Gonzalez-Espinosa 1998). The main causes of forest conversions are agropastoral land creation, tree harvesting, and wildfires (Masera and others 1992, De Jong and Montoya-Gómez 1994). Driving forces behind the process have been governmental incentives for agricultural development, population growth, shifts from subsistence to commercial production systems, infrastructure development, and insecure land and tree tenure systems (De Jong and Montoya-Gómez 1994).

Lack of accurate data on deforestation and forest degradation, and the associated biomass reduction, hinders the development of C models for region-specific land-use patterns. Research on C emissions from land-use change and forestry in Mexico has thus been identified as a high priority in order to better understand the role of biomass and land-cover types in the overall national C balance (Masera and others 1992).

In this paper we present results of a case study in a highly impacted and fragmented landscape in which we apply field-measured C density values to land-use/land-cover (LU/LC) statistics. Our approach for estimating the flux of C between terrestrial ecosystems and the atmosphere was to assign C densities to LU/LC classes and to thus calculate net C flux due to changes in LU/LC areas derived from land-use and forest inventories conducted by Mexican governmental organizations.

Methods

Study Area

The highlands of Chiapas (Figure 1), located at 1500–2900 m elevation in southeast Mexico encompass all or parts of 34 *municipios* (local governmental units akin to US counties). The area contains various forest formations in a complex landscape with high biodiversity resulting from interactions among geological,

edaphic, climatological, and anthropogenic factors (Breedlove 1981). The most important ecosystems include pine, pine–oak, oak, and evergreen cloud forests (Breedlove 1981; González-Espinosa and others 1995b). These forests are broadly representative of the highlands of southern Mexico, Guatemala, Honduras, and northern Nicaragua. The regional climate is subtropical to temperate subhumid, with summer rains and winter droughts. Mean annual rainfall varies between 1100 and 2000 mm. Life zones in the study area include tropical lower montane and premontane moist and subtropical lower montane and montane wet zones (Holdridge 1967). The soils are predominantly dark brown, clayey loam, derived from calcareous rocks (Parra-Vázquez and others 1989). Soil pH varies between 4.6 and 7.3.

Recent changes in land-use patterns have been both extensive and intensive. Only a few decades ago large areas were still covered by old-growth forest, while the current land cover represents a highly disturbed landscape that includes plots of 0.5–2 ha of cultivated land, secondary forests (or tree and shrub fallow), temporary and permanent grazing lands, and disturbed forests (Parra-Vázquez and others 1989, González-Espinosa and others 1991). The structure and composition of the remaining forests have been altered due to selective harvest of pine trees for local timber production and of oak trees for fuelwood and charcoal (De Jong and Montoya-Gómez 1994, González-Espinosa and others 1995a,b) and extensive sheep grazing (De Jong and Montoya-Gómez 1994). Currently, only a few small patches of old-growth forest remain.

Map Data

The data used in this study were derived from two 1:250,000-scale map series bounding the period between the mid-1970s and the early 1990s, a time of dynamic land-use change. The first series is designated Carta de Uso del Suelo y Vegetación (Land Use and Vegetation Maps). For the Chiapas highlands coverage we used three individual LU/LC maps generated from aerial photography covering the years 1972–1982 at scales ranging from 1:50,000 to 1:80,000. Classifications were verified in the field between 1980 and 1985. Individual maps were published by the Mexican federal government agency Instituto Nacional de Estadística, Geografía e Informática (INEGI 1984, 1987, 1988). The maps were digitized into Arc/Info format from paper sources using standard digitizing techniques, maintaining an RMS error of less than 0.004 in the map registration. The individual maps were joined into a single layer in a Transverse Mercator projection following edge-matching procedures. We assigned a nominal date of 1975, corresponding to the mean of the air

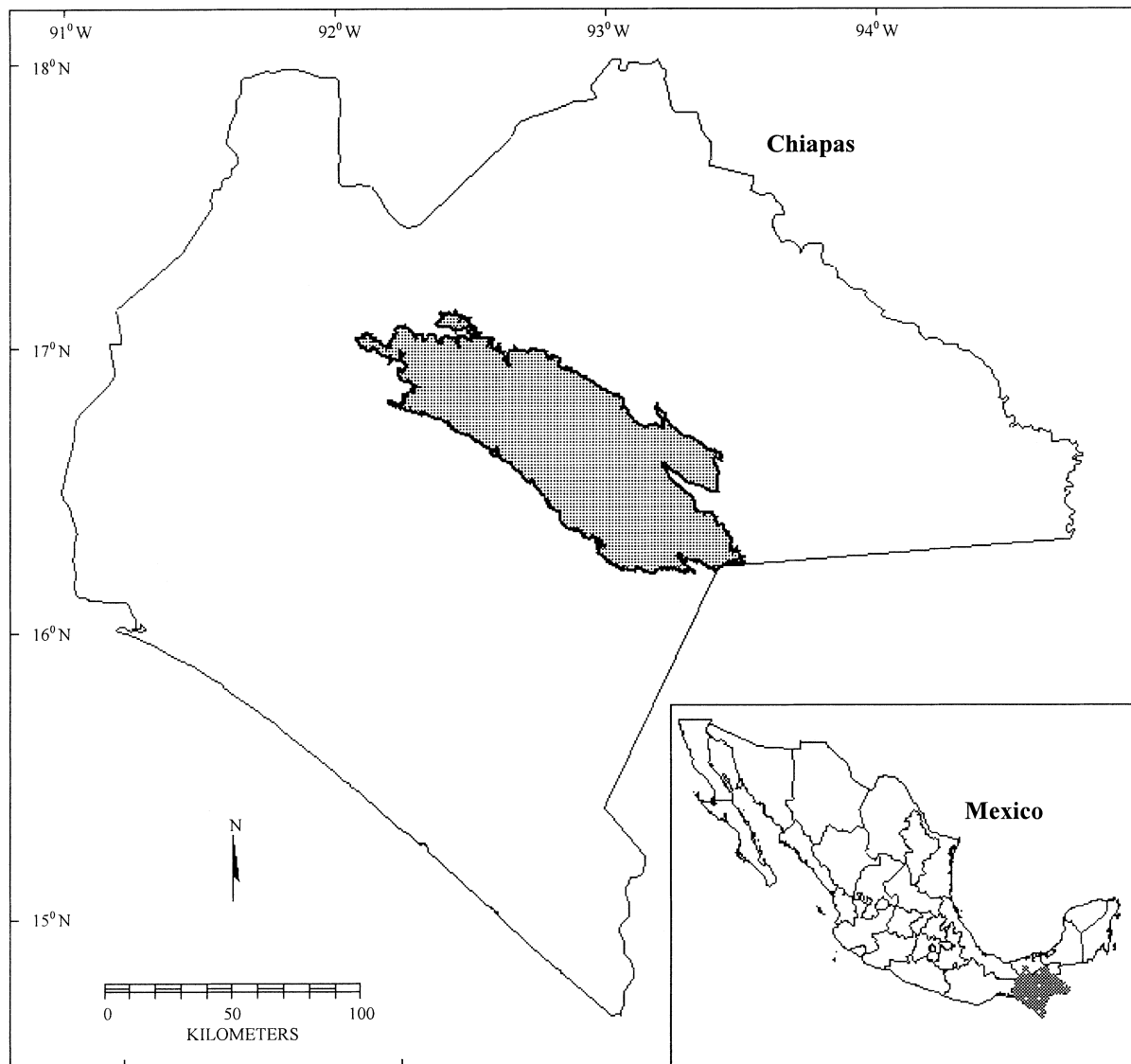


Figure 1. Chiapas highlands study area.

photo dates, to represent the former LU/LC in the Chiapas highlands.

The second map series was published by the Secretaría de Agricultura y Recursos Hidráulicos (SARH 1994) as the *Inventario Nacional Forestal Periódico* (Periodic National Forest Inventory) (Sorani and others 1993) and is described as a hybrid map for forest inventory assessment (Sorani and Alvarez 1996). It was derived by updating existing INEGI land-use classifications with 30-m resolution Landsat TM imagery and improved geographic control. Forested areas were visually classified from 1:250,000 mosaic color composites georeferenced with a 3 arc-second digital elevation model (DEM) by a team of photo interpreters experi-

enced with former forest inventories (Sorani and Alvarez 1996). Classification in nonforested areas were of less importance in this inventory. Because this map was based on cartographic units of the previous map (*Carta de Uso de Suelo y Vegetación*), it had the same accuracy and precision for cartographic purposes and varied only in its areal coverage of the various LU/LC classes (Sorani and Alvarez 1996). A digital copy of this map series was obtained from the Instituto de Geografía, Universidad Nacional Autónoma de México. The individual maps were placed into a common projection (Lambert), edge-matched, and joined into a single layer. Based on an average of the dates of the TM imagery employed, we assigned a nominal date of 1991

to represent the current LU/LC in the Chiapas highlands. Our temporal scale was therefore the 16-year period between 1975 and 1991.

In both map layers, discontinuity of polygons was evident at map edges. This was observed on paper *Uso del Suelo y Vegetación* maps and was also present in the digital *Inventario Nacional Forestal Periódico* maps. These were examined in the individual map tiles and were not altered in the final joined map layers. The map layers were minimally adjusted after reprojection to a common projection (UTM zone 15) and prior to clipping to the study area of the highlands region.

The perimeter of the Chiapas highlands region was demarcated by a 30-m DEM file (EPA 1993), generated by digitizing 1:250,000 scale INEGI topographic maps (UTM zone 15 projection). The 1500-m elevation level was defined as the lower limit of the study area. Overlays of the two digital maps clipped to the highlands region were made and areal summations of the LU/LC classes represented on the two map series were calculated.

In order to quantify LU/LC changes between 1975 and 1991, we devised a composite classification system for purposes of comparison. The INEGI maps used more than 60 classes, some of which were equivalent to one of the 39 UNAM classes (Sorani and others 1993). However, some differences had to be reconciled. For example, the first map reported secondary forests and shrub lands within each forest type, four tropical medium-stature forest classes, 14 scrub vegetation classes, seven pasture classes, and six agriculture classes, but did not differentiate between open and closed forests or report fragmented forests. By contrast, the second map reported fewer forest, scrub, and agricultural types, grouped all high- and medium-stature tropical forests, and did not show secondary forest or scrub, but did separate open and closed forest and reported fragmented forests and perturbed areas (*áreas perturbadas*, defined as forested in the INEGI maps, but no longer so in 1991; SARH 1994). Rates of LU/LC change were calculated by comparing the areal coverage of the various equivalent classes between the two maps.

Based on a 16.3:83.7 ratio derived from the relative distribution of pasture and cultivated land in the INEGI (1984, 1987, 1988) maps of the Chiapas highlands, land use in the perturbed area of the second map was assigned to either pasture or cultivated land based on adjacent polygons until 16.3% of the area was assigned as pasture. In cases where there were no adjacent pasture or cultivated land polygons, perturbed sites in lands that had previously been forested were assigned to pasture. The remaining area of perturbed polygons (83.7%) was then assigned to the cultivated land class.

After carefully reviewing the coding systems and

definitions of terms in documentation for both maps, the following composite classes were adopted as a common basis for comparison: (1) Oak and evergreen cloud forests are forests in which oak trees constitute >80% of the basal area, and remaining small patches of mixed evergreen cloud forests. (2) Pine–oak forests are forests in which oak and pine trees together constitute >80% of the basal area. (3) Pine forests are forests in which pine trees constitute >80% of the basal area. Total canopy cover of the first three classes is >40%. (4) Degraded and fragmented forests include open forests (10%–40% canopy cover), shrubland, and fragmented forest from SARH (1994), and secondary forest, secondary shrubland, tree plantations, and nomadic agriculture from INEGI (1984, 1987, 1988). (5) Cultivated land includes seasonal and permanent agricultural systems. (6) Pasture includes induced and introduced pasture and grasslands. (7) Waterbodies includes lakes and streams, these are assumed to contain minimal biomass and are excluded from C flux analyses.

Carbon Densities

Carbon density data for the major LU/LC classes and the main pools in each class were collected in 39 field plots of 60 × 90-m (0.54 ha) each. The number of plots of each class and their locations were selected to be generally representative of a combination of the distribution of LU/LC classes in the study area and the variability in C densities (based on González-Espinosa and others 1995a), ranging from three plots for the low C-density land uses (pasture and cultivated land) to 10–11 for the pine–oak and degraded and fragmented forest classes (the most common land-use classes and those expected to exhibit high variability in C density). Each plot was divided into six 30 × 30-m subplots to account for intrasite variability.

The average C storage was calculated per unit area (C tons per hectare) for the various pools of each composite LU/LC class, with a transformation of data to account for slope. Diameter at breast height (dbh, 1.3 m) and total height were measured and species determined for all trees >5 cm dbh. Locally developed allometric equations of biomass in relation to dbh and height were used to calculate C densities of the tree component in each plot. All saplings, herbaceous plants, and litter were sampled in four 1-m² plots, dried, and weighed in the laboratory, and the C content was determined on a subsample for each plot. Soil samples were collected in four 1-m deep pits and separated into strata 1–10, 11–20, 21–30, 31–50, and 51–100 cm deep. Soil organic matter (SOM) C content was determined for each sample in the laboratory on three replicates (Walkley and Black 1934). The volume of fallen dead

twigs, branches, and trees was measured using a line intersect method with corrections for vertical angles (Brown and Roussopoulos 1974). Samples of branches and twigs were taken to the laboratory to determine wood density and C content. Small roots (<0.5 cm) were collected in four 0.25-m², 1-m deep pits. The samples were oven dried and weighed (Saldarriaga and others 1988). Carbon content was determined on subsamples of the roots from each pit. Carbon densities of large roots (>0.5 cm) were estimated from published allometric relationships between root and aboveground biomass (Pinard and Putz 1997, Cairns and others 1997).

A one-way analysis of variance (ANOVA) was performed to test for differences among the LU/LC classes for overall, aboveground, root, and SOM C densities (SAS Institute Inc. 1988). Where significant differences were detected by the ANOVA, the Scheffe multiple comparison procedure was used to identify specific pairwise differences among the various classes. The level of significance was $\alpha = 0.05$.

The C density of the oak and evergreen cloud forest class was calculated as an area-weighted average of the C densities of the sampled oak ($N = 5$) and evergreen cloud forest ($N = 2$) plots. The C density of the degraded and fragmented forest class was calculated as an area-weighted average of the C densities of the sampled tree ($N = 6$) and shrub ($N = 4$) fallows. In both cases, weighting was done according to the relative 1975 areas of these land covers.

To calculate the C pools and flux resulting from land-use changes between the 1970s and 1990s, we separated vegetation pools, including live biomass, necromass, and roots, from the SOM pool. For each LU/LC class we then multiplied the total area present in both 1975 and 1991 by the average C densities of the vegetation and SOM pools. For demonstration purposes, but not for analytical use, we presented C flux in the study area for the period of time represented by our study.

To examine possible errors in estimates associated with field-derived C densities, we calculated 95% confidence intervals (CI) for vegetation and SOM C pools (SAS Institute Inc. 1988) and reported the resulting ranges in C flux.

Results

Changes in Land Use and Land Cover

During the 16-year period between the two inventories, the area of oak and evergreen cloud forests decreased by 50%, while both pine-oak and pine forests decreased by 51% (Table 1). Thus the total area of

Table 1. Chiapas highlands land-use/land-cover (LU/LC) change from 1970s to 1990s

LU/LC class	Area (ha)		Change (%)
	1975 ^a	1991 ^b	
Oak and evergreen cloud forests	65,312	32,699	-50
Pine-oak forests	119,139	58,249	-51
Pine forests	72,109	35,001	-51
Total of closed forests	256,560	125,949	-51
Degraded and fragmented forests ^c	203,236	316,414	56
Cultivated land	136,182	146,563 ^d	8
Pasture	23,603	30,685 ^d	30
Water bodies	581	551	-5
Totals	620,162	620,162	

^aNominal year of INEGI (1984, 1987, 1988) land-use and vegetation inventory.

^bNominal year of SARH (1994) national forest inventory.

^cIncludes open forests and fragmented forests from SARH (1994) and secondary forest, secondary shrub, tree plantations, and nomadic agriculture from INEGI (1984, 1987, 1988).

^dIncludes áreas perturbadas of SARH (1994) (61,507 ha) parceled to cultivated lands (83.7%) and pasture (16.3%).

closed forests was reduced by about half, from 256,560 to 125,949 ha. Meanwhile, the degraded and fragmented forests increased by 113,178 ha, cultivated lands grew by 10,381 ha, and pastures increased by 7082 ha. The net LU/LC change during this time interval was thus a trend away from closed forest ecosystems toward open, fragmented, and secondary forests and, to a lesser extent, toward various agricultural uses of the land (Figures 2 and 3).

While the net trend was from closed forests toward disturbed forests and agricultural land uses, there is also evidence of abandonment and forest regeneration for some of the land in agropastoral uses between 1975 and 1991. The 1991 LU/LC areas include 61,507 ha classified as perturbed areas in the SARH (1994) forest inventory parceled 83.7% to cultivated land (51,481 ha) and 16.3% to pasture (10,026 ha). When these areas are subtracted, the result is that the agropastoral land use decreased between 1975 and 1991. Thus abandonment of 41,100 ha (30%) of the cultivated land and 2944 ha (12%) of the pasture, and regrowth as forest, may be deduced. The distribution of forest classes to which these lands regenerated is unknown.

Carbon Densities and Flux

Mean total C densities in the LU/LC classes varied more than threefold, from 147 to 504 tons/ha in pastures and oak and evergreen cloud forest, respectively (Table 2). Only cultivated land and pasture had statistically lower total mean C densities than did the

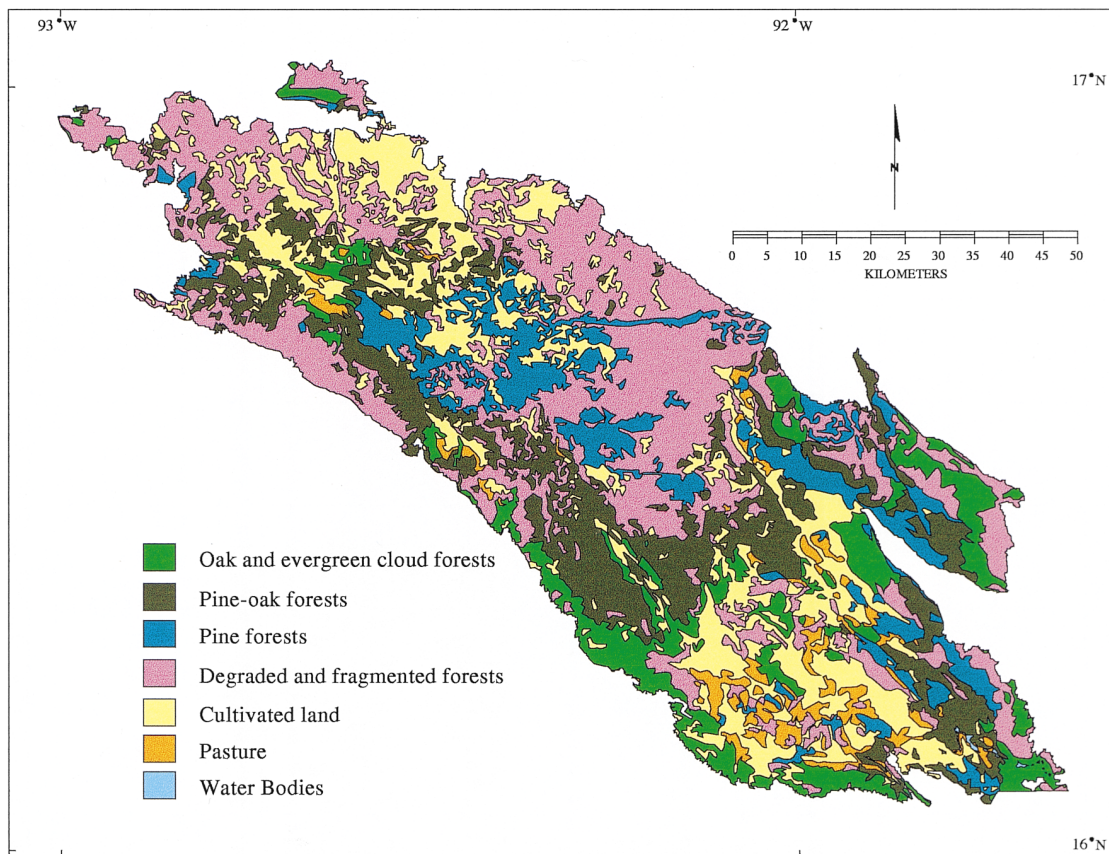


Figure 2. Land-use/land-cover in the Chiapas highlands, Mexico, circa 1975.

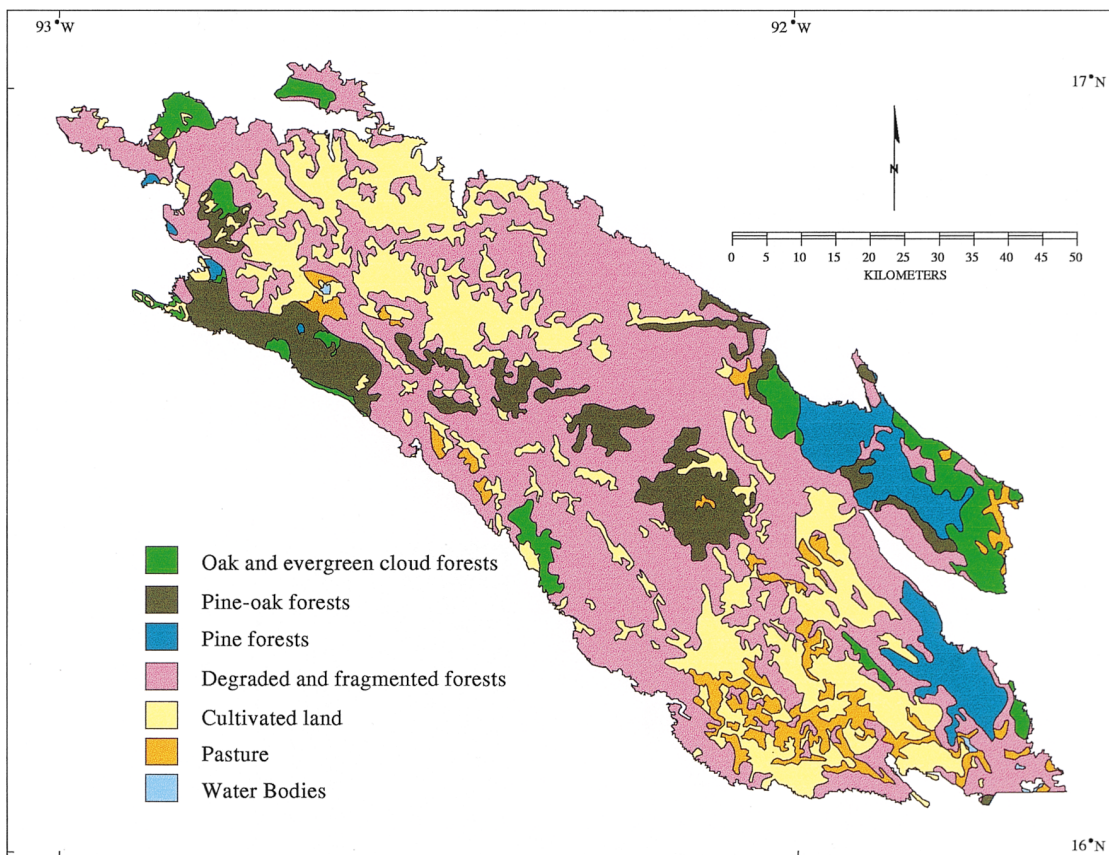


Figure 3. Land-use/land-cover in the Chiapas highlands, Mexico, circa 1991.

Table 2. Estimated carbon (C) densities for Chiapas highlands land-use/land-cover (LU/LC) classes^a

LU/LC class	N	C Density (Mg C / ha)			Total
		Aboveground	Belowground		
			Roots	Soil	
Oak and evergreen cloud forests	7				
Mean		189.0 ^A	36.0 ^A	242.8 ^A	503.7 ^A
Range		108.4–302.0	20.7–55.1	101.5–460.8	232.3–818.0
CV (%)		14	16	16	14
Pine–oak forests	11				
Mean		135.4 ^A	30.9 ^A	174.4 ^A	340.7 ^A
Range		77.6–168.6	19.8–40.5	85.4–412.0	229.2–595.6
CV (%)		21	24	53	32
Pine forests	5				
Mean		120.0 ^{AB}	25.7 ^{AB}	172.6 ^A	318.3 ^A
Range		79.7–244.0	16.7–46.1	129.7–275.9	243.1–566.0
CV (%)		58	46	34	44
Degraded and fragmented forests	10				
Mean		29.1 ^B	8.9 ^{BC}	184.2 ^A	222.2 ^A
Range		9.6–106.2	1.3–26.8	144.5–315.5	168.2–401.7
CV (%)		32	36	46	44
Cultivated land	3				
Mean		6.0 ^B	0.6 ^C	153.3 ^A	159.9 ^B
Range		4.0–8.1	0.2–0.8	135.2–177.8	143.6–184.4
CV (%)		34	53	14	14
Pasture	3				
Mean		18.1 ^B	3.8 ^{BC}	124.8 ^A	146.7 ^B
Range		4.1–37.1	0.8–8.7	88.2–183.5	117.7–188.4
CV (%)		94	114	41	25

^aAboveground includes tree and herbaceous phytomass, coarse woody debris, and litter. Belowground includes roots and soil organic carbon. Means with the same superscript are not statistically different ($\alpha = 0.05$) from other means in the same column.

forest classes. These modified LU/LC types had lower C densities due largely to smaller aboveground and root biomass pools. The belowground C densities showed a decrease from approximately 279 tons/ha in oak and evergreen cloud forests to 129 and 154 Mg/ha in pasture and cultivated land, respectively. The average CV for total C density was 29%, with higher values associated with pasture aboveground (94%) and root biomass (114%), pine forest aboveground biomass (58%), and pine–oak forest SOM (53%) (Table 2). Lower coefficients of variation were observed in oak and evergreen cloud forests (14%) and cultivated land (14%).

A close examination (Table 2) reveals that the lower belowground C densities are a result of lower root biomass C densities, which range from <1 to 36 tons/ha, while SOM C densities were not significantly different, ranging from 125 to 243 tons/ha. The proportion of roots in the total belowground C pools decreased from a range of 13%–15% in the intact forest types to a range of 0.4%–5% in the modified LU/LC types (Table 2). When only the forest LU/LC classes were compared (agropastoral land uses excluded), only the degraded and fragmented forests had significantly lower aboveground and root C densities.

From differences in C storage between 1975 and 1991, we estimate that a net total of 19.99×10^6 tons C were released to the atmosphere during the study (Table 3). This is equal to approximately 12% of the total 1975 biospheric C pool. Approximately 34% of the 1975 vegetation C pool was lost, while the SOM C pool remained nearly stable. Total C flux in the study area is shown for demonstration purposes only (Figure 4). The primary flux of terrestrial C to the atmosphere may be attributed to the loss of above and belowground biomass from the three closed forest ecosystems in the study area.

A large degree of uncertainty exists in our flux estimates, even regarding the sign, i.e., whether the region is a net source or sink (Table 3). If extreme estimates were correct, e.g., maximum 1975 pools compared to minimum 1991 pools, or vice versa, then based only on field-derived C density estimates, the highlands study area could have been either a highly aggrading or degrading landscape.

Discussion

Although the study area has been intensively managed by indigenous Mayan populations for centuries

Table 3. Carbon (C) pool and flux estimates resulting from land-use change between 1970s and 1990s in Chiapas highlands^a

Land-use / land-cover class	C Pools 1975 ($\times 10^6$ tons)		C Pools 1991 ($\times 10^6$ tons)		C Flux 1975–1991 ($\times 10^6$ tons)		
	Vegetation	Soils	Vegetation	Soils	Vegetation	Soils	Total
Oak and evergreen cloud forests	14.70 (1.58) ^b	15.86 (1.86)	7.36 (0.79)	7.94 (0.93)	7.34 ^c (4.97, 9.71)	7.92 (5.13, 10.71)	15.26 (10.10, 20.42)
Pine–oak forests	19.82 (2.50)	20.78 (6.50)	9.69 (1.22)	10.16 (3.18)	10.13 (6.40, 13.85)	10.62 (0.95, 20.30)	20.75 (7.35, 34.15)
Pine forests	10.51 (5.12)	12.45 (3.71)	5.10 (2.49)	6.04 (1.80)	5.41 (–2.21, 13.02)	6.41 (0.89, 11.92)	11.82 (–1.32, 24.94)
Degraded and fragmented forests	7.71 (1.59)	37.44 (10.76)	12.00 (2.48)	58.29 (16.76)	–4.29 (–8.36, –0.23)	–20.85 (–48.37, 6.67)	–25.14 (–56.73, 6.44)
Cultivated land	0.90 (0.27)	20.88 (3.39)	0.97 (0.29)	22.47 (3.65)	–0.07 (–0.63, 0.49)	–1.59 (–8.63, 5.44)	–1.66 (–9.26, 5.93)
Pasture	0.52 (0.56)	2.95 (1.37)	0.68 (0.73)	3.83 (1.78)	–0.16 (–1.46, 1.15)	–0.88 (–4.04, 2.27)	–1.04 (–5.50, 3.42)
Total C pools and flux	54.16 (11.62)	110.36 (27.59)	35.80 (8.00)	108.73 (28.10)	18.36 (–1.29, 37.99)	1.63 (–54.07, 57.31)	19.99 (–55.36, 95.30)
As % of total 1975 C pools					34%	1%	12%

^aPositive values indicate flux from the terrestrial biosphere; negative values indicate flux from the atmosphere.

^bMean with 95% confidence interval shown in parentheses.

^cMean (range) is based on error associated with estimating C flux from differencing 1975 and 1991 pools only. Upper range = (maximum pool for 1975—minimum pool for 1991), and lower range = (minimum pool for 1975—maximum pool for 1991). Negative pools were assumed equal to zero for flux calculations.

with slash-and-burn (*milpa*) management systems, these systems have substantially changed in recent decades to more permanent agriculture and extensive cattle and sheep ranching. Land use under *milpa* has been intensified from long fallow and short crop rotations to systems with short fallow and long crop rotations, more resembling permanent agriculture. Furthermore, the fallow land is often intensively grazed by sheep or cattle, thus limiting regeneration of shrub or tree biomass (Parra-Vázquez and others 1989).

Changes in forest cover can be largely attributed to opening of closed forests. Pine trees in both pine and pine–oak forests are cut for timber production, while oak trees are used for fuelwood and making charcoal. Logged pine forests are later used for extensive grazing of cattle and sheep, while pine–oak and oak forests are continuously logged for timber, fuelwood, and charcoal production or are cultivated or converted to pasture. Resulting landscapes include small patches of lightly perturbed old-growth forests surrounded by large areas of young secondary shrub and woody vegetation, heavily degraded forests, induced pastures, and cultivated lands (González-Espinosa and others 1991, De Jong and Montoya-Gómez 1994).

Soil C densities measured in oak and evergreen cloud forests, pine–oak forests, and pine forests were somewhat higher than the range of values reported

from similar forest types (Table 4). Oak and evergreen cloud forests had a particularly high soil C density, 243 tons/ha. Soil C densities were generally high across all forest classes because the soils are calcareous and the active calcium carbonate and, to a lesser degree, the exchangeable calcium in saturated environments are effective stabilizers, protecting organic matter against microbial biodegradation (Duchaufour 1982). Other factors affecting soil C storage include rainfall and soil fertility (Lugo and Brown 1993). Detwiler (1986) notes no reduction in soil C density due to harvesting. This observation is in agreement with our results. Thus the soil C density of the degraded and fragmented forests (184 tons/ha) was not lower than the three closed forest classes (Table 2). In contrast, Houghton and others (1983) report a 35% decrease in soil C following selective logging in tropical forests.

In this study soil C was reduced 22% in cultivated land and 36% in pasture, relative to the average values for the intact forests. These results may be compared with those reported by Detwiler (1986) that the cultivation of tropical soils reduces their C content by 40% five years after clearing and that use of these lands for pasture reduces their soil C content by about 20% within one year. Similarly, Houghton and others (1983) report that a conversion of tropical forests to permanent agriculture decreases soil C by 50% and conver-

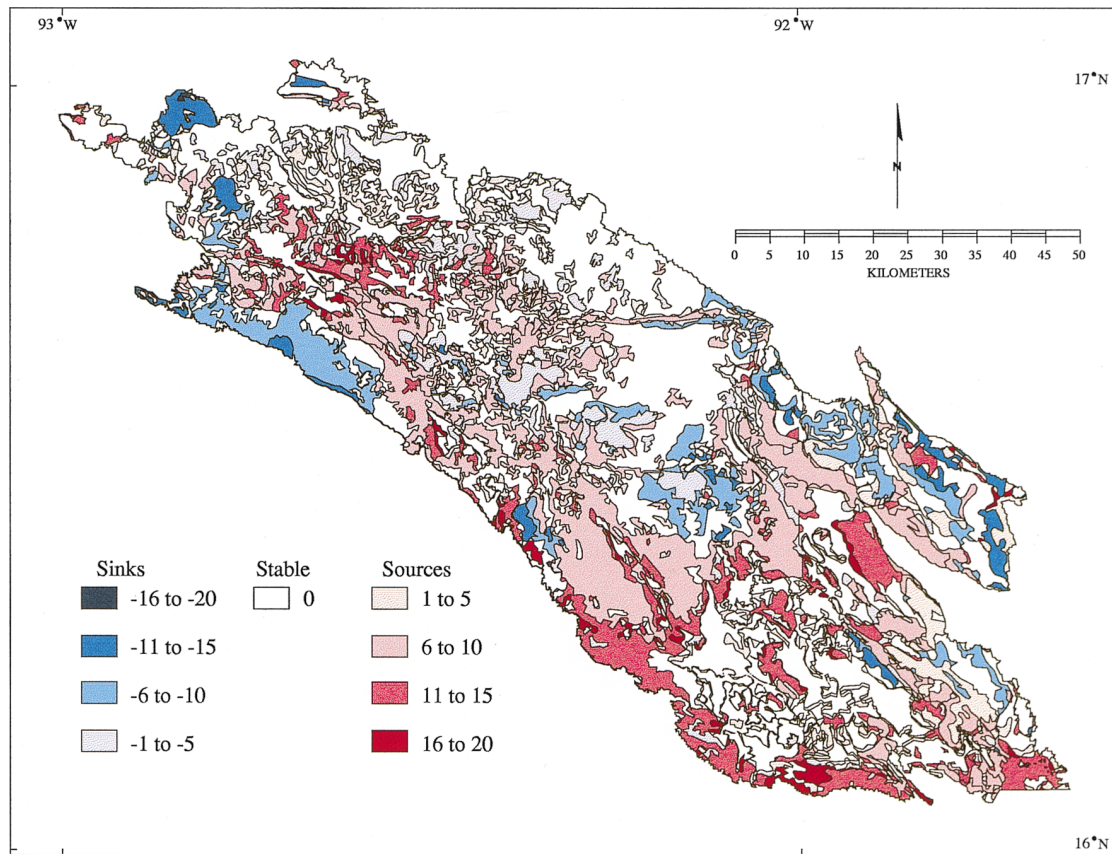


Figure 4. Estimated mean carbon flux (tons C/ha/yr) between ca. 1975 and ca. 1991. This map is for demonstration purposes and was not used for carbon pool and flux analyses. Positive values indicate fluxes from the terrestrial biosphere; negative values indicate fluxes from the atmosphere.

sion to pasture decreases soil C by 25%. A possible explanation for the difference is that the cultivated plots in our study may have been created from forests more recently and the pastures were more intensively impacted and/or were older than those observed by Detwiler (1986) and Houghton and others (1983).

Because these land-use changes not only cause changes in land cover, but may also result in changes in C density within each LU/LC, we suspect that the average C density for each land cover class may have been higher in 1975 than in 1991, the latter being close to when C densities were measured in our study. For example, activities such as grazing, or pole and firewood harvest, can reduce forest biomass without affecting crown cover when observed by remote sensing. Thus the C flux that occurred between 1975 and 1991 might be higher than we have estimated. This could not be verified because of the lack of previous studies of soil and biomass C in Chiapas highlands forests. However, such biomass degradation has been reported for other tropical forests (Flint and Richards 1991, Brown and Lugo 1992).

Potential sources of error in our analysis associated with cartographic data included the wide range of dates encompassed in the *Uso del Suelo y Vegetación* maps, the multiple organizations participating in the generation of both map layers, and the different classification systems employed by the original map creators. Because the aerial photography covered a decade, it is impossible to define precisely the time frame over which the modeled LU/LC occurred, thus we used a nominal date. While the SARH (1994) map is estimated to have an accuracy rate of >85% (Sorani and others 1993), the accuracy of the INEGI (1984, 1987, 1988) maps is unknown, although the aerial photography was field-verified at a minimum of 50 points per map, indicating a relatively high degree of accuracy at the time of compilation of the maps. A complete independent assessment of the accuracy of the registration of the two map layers was not possible due to a shortage of relocatable points within the layers; however, best efforts were made to find and align identifiable locations.

In a project in Mexico's Yucatán Peninsula employing a comparison of the same cartographic data sources

Table 4. Reported above plus belowground biomass and soil organic carbon (C) densities to 100 cm depth compared to three closed forest land-use/land-cover classes—oak and evergreen cloud, pine–oak, and pine—represented in Chiapas highlands

Forest type	Density (tons/ha)		Source	Comments
	Biomass C	Soil C		
Tropical montane	145	130	Adams 1997	preanthropogenic (= potential)
Temperate coniferous	56	109.1	Masera and others 1992	nomenclature used in Mexico
Temperature broadleaf	39	29.5	Masera and others 1992	nomenclature used in Mexico
Montane	60		Olson and others 1983	low values
Temperate coniferous	50	102	Cairns and others 1995	nomenclature used in Mexico
Temperature broadleaf	60	102	Cairns and others 1995	nomenclature used in Mexico
Tropical premontane wet	142		Golley and others 1975	cited in Brown and Lugo 1982
Tropical montane wet	214		Brun 1976	cited in Brown and Lugo 1982
Tropical premontane moist	90		Bandhu and others 1973	cited in Brown and Lugo 1982
Tropical premontane moist	181	44	Greenland and Kowal 1960	cited in Brown and Lugo 1982
Tropical premontane moist	228		Huttel 1975	cited in Brown and Lugo 1982
Subtropical lower montane wet	172		Tanner 1980	cited in Brown and Lugo 1982
Tropical lower montane wet		193	Holdridge and others 1971	cited in Brown and Lugo 1982
Tropical lower montane moist		170	Holdridge and others 1971	cited in Brown and Lugo 1982
Subtropical/tropical wet & rain		175	Detwiler 1986	adapted from Brown and Lugo 1982
Subtropical/tropical moist		98	Detwiler 1986	adapted from Brown and Lugo 1982
Tropical montane moist		120	Detwiler 1986	interpolated from Post and others 1982
Tropical montane wet		160	Detwiler 1986	interpolated from Post and others 1982
Tropical closed		123	Detwiler 1986	area weighted all tropical life zones
Reported averages	120	120		
Reported ranges	39–228	30–193		
This study averages	179	197		
This study range of averages	146–225	174–243		

we used in our present work, Carranza-Sánchez and others (1996) concluded that, because of the differences between the two sources in terms of their use of aerial photography vs satellite imagery, the relative degree of ground truthing, and the different terminology used to label LU/LC classes, these maps are unsuitable for estimating land-use change and deforestation. While we agree that the ideal situation for comparing past and future LU/LC inventories would use similar data sources, others have recently reported land-use change analysis based on comparisons of different map layers and overlay techniques (Schreier and other 1994, Kress and others 1996; Thomlinson and others 1996). Because the 1990s map (SARH 1994) uses as its basis the same cartographic data as the 1970s map (INEGI 1984, 1987, 1988), the primary factor making a comparison difficult is the difference in labeling LU/LC classes. We feel that for C flux purposes, unlike for strictly land-use change or deforestation purposes, comparisons based on our assumptions regarding equivalent LU/LC classes are clearly warranted. Our analysis used two digital maps of the

Chiapas highlands produced by Mexican government agencies for these time periods that are the best available sources.

Forest biomass C densities are commonly estimated in one of two ways. One method uses inventory data, either volume estimates or stand tables, to calculate total aboveground biomass (Brown 1997). Advantages of this approach are that inventory data are usually more abundant and gathered from large sampling areas using a planned sampling method designed to represent the population of interest. Common problems with the inventory approach are that such inventories are usually focused on commercial forests and species, only large trees are measured, many of the inventories are very old, critical data and regions are omitted, only the aboveground tree biomass is estimated, and the maximum diameter class is open-ended. The second approach is based on ecological studies that have sampled the various biomass pools in plots of various sizes and using different methods (Brown and Lugo 1982, Olson and others 1983). The main advantages are that local conditions are accounted for and all the ecosystem

components can be measured. Disadvantages are that study sites are frequently not representative of the population of interest, results cannot be extrapolated to larger areas, and sites are often selected in forests with many large diameter trees, thus tending to overestimate biomass density (Brown and others 1989, Brown and Lugo 1992).

Our approach incorporates some strengths of both approaches. All LU/LC class biomass and soil C pools were accounted for, locally developed allometric equations were used to calculate tree biomass C densities, and all species and sizes of plants were measured. Even though our plots were not strictly randomly selected and the total area sampled was $<0.01\%$ of the study area, the locations and numbers of plots were selected to be representative of both the distribution and variability of the LU/LC types present in the Chiapas highlands.

A measure of statistical confidence in our mean C density estimates provides a unique opportunity to demonstrate the uncertainty of estimating either regional C flux, as in this study, or national greenhouse gas inventories, as required by the UN Framework Convention on Climate Change (United Nations 1992). Because cartographic errors were not quantified, the errors indicated for C pool and flux estimates are only those associated with experimental plot sampling (Table 3). The resulting degree of uncertainty, expressed as ranges, in our flux estimates is large. While some of the confidence intervals are larger than the mean C pool values, we may rightfully assume these values are unrealistic, because this would infer negative pools (i.e., pasture vegetation). We therefore assumed them to be equal to zero when calculating C flux. Because zero pools of known LU/LC ecosystems are unrealistic, ranges in our reported C flux values may be considered to reflect worst-case errors. For instance, because we know that the area of pine forests decreased in our study area between the 1970s and 1990s, it is reasonable to assume that the vegetation and total net fluxes could not have been from the atmosphere (negative values in Table 3). Similarly the areas of both cultivated land and pasture increased, thus net fluxes could not have been from the biosphere (positive values in Table 3). While the large range in SOM flux estimates indicates a high degree of heterogeneity in soil C concentrations, the net result is assumed to be approximately no change in the SOM pool. These constraints on the error associated with our C flux estimates due to field sampling make them more realistic and strengthen our confidence in the values.

Means of reducing errors in C flux estimates would include using larger sample sizes and conducting sequen-

tial LU/LC inventories in an identical manner so that spatial matching of cartographic areas and C densities would be possible. These means would also increase the costs of producing greenhouse gas emissions estimates. Standard guidelines for reporting greenhouse gas emissions encourage the quantification of uncertainty (UNEP/OECD/IEA/IPCC 1995), but the default values that are commonly employed make such quantification difficult.

The latest Mexican greenhouse gas inventory update reports that land-use change and forestry result in a national annual net emission of 37.1 Tg C as CO_2 , composed of a source of 59.4 Tg C, partially offset by a sink in managed forests and abandoned lands of 22.3×10^6 tons C (Masera and others 1995). Therefore, based on our findings, the Chiapas highlands, representing only 0.3% of Mexico's territory, contributed 3% of the nation's annual LU/LC C emissions $[(17.5 \times 10^6 \text{ tons}/16 \text{ yr})/(37.1 \times 10^6 \text{ tons}, \text{ Table } 3)]$. This indicates that the Chiapas highlands have experienced a rapid process of deforestation and land use change during these 16 years compared to the rest of Mexico.

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