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Household Land Management and Biodiversity: Secondary Succession in a Forest-Agriculture Mosaic in Southern Mexico

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ABSTRACT. This study evaluates anthropogenic and ecological dimensions of secondary forest succession in Mexico's southern Yucatán peninsular region, a hotspot of biodiversity and tropical deforestation. Secondary succession in particular constitutes an ecologically and economically important process, driven by and strongly influencing land management and local ecosystem structure and dynamics. As agents of local land management, smallholding farmers in communal, i.e., *ejido* lands affect rates of forest change, biodiversity, and sustainability within and beyond their land parcels. This research uses household surveys and land parcel mapping in two *ejidos* located along the buffer of the Calakmul Biosphere Reserve to analyze how household socioeconomics and policy institutions drive allocations to successional forests in traditional crop fallows and in enriched fallows. Results indicate that household tenancy, livestock holdings, labor-consumer ratios, and receipts of agricultural subsidies are the strongest determinants of traditional fallow areas. Whereas the latter two factors also influence enriched successions, local agroforestry and reforestation programs were the strongest drivers of fallow enrichment. Additionally, the study conducts field vegetation sampling in a nested design within traditional and enriched fallow sites to comparatively assess biodiversity consequences of fallow management. Although enriched fallows display greater species richness in 10x10 m plots and 2x2 m quadrats, plot-scale data reveal no significant differences in Shannon-Wiener or Simpson's diversity indices. Traditional fallows display greater species heterogeneity at the quadrat scale, however, indicating a complex relationship of diversity to fallow management over time. The article discusses the implications of the social and ecological analyses for land change research and conservation policies.

Key Words: *biodiversity; household decision making; land allocation models; Mexico; secondary forest.*

INTRODUCTION

Tropical forests face accelerated rates of transformation worldwide, reflecting a leading source of land use/cover change (Achard et al. 1998, FAO 1999, Cincotta and Engelman 2000). This trend is particularly disturbing given the numerous ecological services they provide, including the maintenance of biodiversity, carbon sequestration, net primary production, hydrologic cycles, and global biogeochemistry (Wilson 1988, Shukla et al. 1990, Skole and Tucker 1993, Myers 1994, Vitousek et al. 1997, DeFries 1999, Daily et al. 2000, Houghton et al. 2000). In addition, forests are a fundamental resource for local cultures. The extending reach of national and global economies into previously remote forest frontiers has led to rising concerns about land degradation, loss of

habitat, ecosystem services and diversity, invasion by exotic species, and increasing vulnerability of forest-dependent cultures (Barrow 1991, Kummer 1992, Kasperson et al. 1995, Angelsen and Kaimowitz 1999, Mooney and Hobbs 2001, Wright and Muller-Landau 2006). Multiple efforts to monitor and analyze the problem include the identification of deforestation "hotspots" (Achard 1998), and the use of remote sensing technologies to monitor them (Fuller 2006, Roy Chowdhury 2006*a*).

A number of studies are also beginning to focus on regenerating tropical forest, widespread in Latin America and other parts of the world (Brown and Lugo, 1990, Corlett 1995, FAO 1996, Moran and Brondizio 1998, Smith et al. 1999, de Jong et al. 2001, Southworth and Tucker 2001, Moran et al.

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2002, Abizaid and Coomes 2004). Secondary succession following anthropogenic land use, e.g., after land abandonment or as part of agricultural fallow cycles, comprised a third of the world's closed-canopy tropical forests in 1980 (Lugo and Brown 1990). It is estimated that a third or more of the area deforested between 1970 and 1990 is now returning to forest (Rudel et al. 2002, Hecht 2004). Several studies indicate that although secondary forests are distinct from mature forest ecosystems in structure, composition, function, and ecological services, they nonetheless play significant roles in local land dynamics, contributing to soil stabilization and the restoration of soil nutrient stocks and flows, carbon sequestration, vegetative cover, and habitat for various forest-dwelling species, and accreting biodiversity (Unruh 1988, Brown and Lugo 1990, Lugo and Brown 1992). Forest fallows also provide economic value by restoring soil fertility for future agricultural land uses, and provide timber and nontimber resources for extractive communities (Brown and Lugo 1990, Anderson et al. 1991, Southworth and Tucker 2001). As sites of intensive human use, secondary forests may release some extractive pressure on mature forests, thereby aiding their conservation (Brown and Lugo 1990). National and international regimes of environmental conservation, including state and nongovernmental organizations, recognize the ecological and economic potential of secondary forests, and frequently promote fallow conservation, agroforestry, and community-based forest management (Snook and Zapata 1998, Roy Chowdhury 1999, Abizaid and Coomes 2004). Antinori and Bray (2005) highlight the fusion of traditional and enterprise-based institutions in Mexico's *ejido*-based community forestry sector, detailing their success in reconciling local conservation and economic development goals.

The processes that drive forest change are complex and multiscalar, and include local decision-making agents, as well as national and international policies. In a cross-national analysis, Kauppi et al. (2006) linked transitions in forest area, growing stock, biomass, and sequestered carbon to complex national trends in conservation, urbanization, infrastructure development, migration, and agricultural transformations. In addition to regional and cross-national approaches, household studies form an important component of forest transition analyses. Secondary forests are critical to farming household economies in the tropics, and household farming decisions strongly influence fallow areas and

management. Zimmerer (2004) notes that household-scale analysis is a cornerstone of the cultural-ecological research tradition (e.g., Brookfield 1964, Grossman 1977, Denevan 1983, Coomes et al. 2000, Bassett and Zimmerer 2003), as well as prominent in a subset of spatially explicit human dimensions research (e.g., see Moran and Brondizio 1998, Geoghegan et al. 2001, Fox et al. 2003). Household demography and life cycles strongly determine land use (Walker and Homma 1996, McCracken et al. 1999, Walker et al. 2000, Perz 2001). However, empirical studies of household and policy determinants of tropical forest fallows remain sparse, as do studies that further trace the ecological consequences of household fallow management.

Smith et al. (1999) and Coomes et al. (2000) linked forest fallow intervals and extents to population density and smallholder households in Peruvian Amazonian colonies, and Perz and Walker (2002) have investigated the role of household life cycles specifically in relation to fallows and farm abandonment in the Brazilian Amazon. Southworth and Tucker (2001) analyzed satellite imagery and interview data to identify anthropogenic and environmental determinants of reforestation in western Honduras, linking forest regrowth to topography, local logging bans, and processes of agricultural intensification. Abizaid and Coomes (2004) studied how household demography and policy factors affected fallow extents and duration in the southern Yucatán peninsular region of Mexico. This study extends the body of knowledge on household-policy-fallow relationships through a case study in southern Mexico, wherein regional rates of recent deforestation are exceeded by those of secondary succession on agricultural lands. The study uses field survey data to analyze (1) how socioeconomic and institutional factors influence household areas allocated to forest fallows, and (2) examines tree species diversity under two distinct fallow management regimes: forest successions on traditional crop-fallows, vs. enriched forest successions.

STUDY REGION

Historical and biophysical context

The southern Yucatán peninsular region (SYPR) in Mexico is located adjacent to the Lacandón forests of Chiapas, the Maya Biosphere Reserve of Petén

in Guatemala, and the forested ecosystems of Belize (Fig. 1). The region houses the oldest remaining tropical forests in Middle America, and underwent significant deforestation and agricultural use by the ancient Mayan civilization (ca. 1000 BC to AD 1000, Turner 1983). After the collapse of the Classic Maya, the area regained its sub-humid tropical forest and remained sparsely occupied until the mid 1800s and early 1900s, when trading in *chicle* resin and hardwoods commenced (Lundell 1933, Edwards 1957, Klepeis and Turner 2001). Seasonally humid, medium-tall statured (*mediana*) forests prevail on well-drained uplands and relatively short-statured forests within large wetlands or *bajos* (Lundell 1934, Flores Guido 1987, Turner et al. 2004). Lowland, flooding vertisols characterize the *bajos* and shallow, alkaline redzinas (mollisols) are found in the well-drained uplands, the latter is preferred for agricultural use (Turner et al. 2001, 2004). Elevation ranges from 100 to 300 m above mean sea level, with little or no surface drainage (Fig. 2). Mean annual rainfall varies from 900 to 1400 mm, gradually increasing to the south and east.

## Land tenure, use and policy responses

Infrastructure development in the 1960s and the establishment of communal land grants or *ejidos*, initiated population growth and rapid land use change in the 18,700 km² SYPR, resulting in the region's designation as a hotspot of deforestation (Achard et al. 1998, Turner et al. 2004). *Ejidos* remain the dominant land tenure, along with some private holdings, i.e., 5% total land area, and federally owned lands (25–35%) (Reforma Agraria 1988, SEFA 1991). Analysis of Landsat imagery in collaboration with the Land Cover and Land Use Change (LCLUC) SYPR research project yielded a 0.4% annual rate of deforestation over 1987–1996 (see Fig.1, Roy Chowdhury and Schneider 2004). Overall, rates of succession exceeded those of agricultural deforestation. Compared with 2.34% of secondary forest and 0.36% of mature/primary forests felled for agricultural use, 3.95% of former agricultural lands transitioned into young forest fallows, mid-late successional forests, approximately 7–15 yr of age, expanded in area by 425 km² (2.5%), and 5.98% of older secondary successions matured further (Roy Chowdhury 2006*b*). Most successional transitions occurred on *ejido* lands.

The 723,185 ha Calakmul Biosphere Reserve (CBR) was established in 1989, and aims to regulate the activities of *ejidos* within its buffer zones, restricting all agricultural deforestation in the long term (Acopa and Boege 1998). The dominant land use in the region remains *milpa* agriculture for subsistence, involving a rotation of maize, beans, and squash. The maize crop typically is grown during the summer wet season, traditionally with fallow periods of over 10 yr after 2–3 successive annual maize harvests (Turner 1983, Vogeler 1974, Klepeis et al. 2004). Other land uses include market cultivation of jalapeño chili, citrus crops, pasture, agroforestry, and/or fallows enriched with timber and fruit trees.

Regional policies address conservation concerns as well as local livelihoods, entailing efforts for economic development, sustainable land use planning, forest conservation, ecotourism, nontimber forest products (NTFPs), and fallow enrichment through agroforestry and reforestation projects. The *Programa de Apoyo Directo al Campo*—Direct Rural Assistance Program (PROCAMPO) agricultural policy entails a fixed payment to farmers, based on area cultivated in staple food crops during 1994. The state and NGO subsidized *roza-pica-siembra* (RPS, or zero-burn) agricultural project is also a conservation intervention. RPS provides a hectare-based subsidy for intercropping green fertilizers (*Canavalia ensiformis* or *Mucuna* spp.) with *milpa* to prolong soil fertility, sedentarize agriculture, and reduce deforestation, and sometimes also distributes tree saplings for reforestation. A more widespread agroforestry/reforestation program dates to the 1990s, is promoted by the CBR, agriculture and environment secretariats, and environmental NGOs, and also entails a modest hectare-based subsidy and extension services for existing agroforestry plots or fallow enrichment.

Ejidos in Calakmul vary in their demographic and socioeconomic characteristics, and in their engagement with institutional projects, giving rise to variations in land management and crop-fallow decisions. Forest fallows fall under two dominant regimes: traditional fallow management, as part of crop-fallow cycles, or enrichment with economically valuable tree species. Enriched fallows entail a distinct disturbance regime from traditional fallow forests, entailing the opening of gaps in the secondary forest for plantings, weeding, and other maintenance activities. To date, no study has undertaken a linked, comparative analysis of

Fig. 1. Land use changes in the study region (adapted from Roy Chowdhury 2006*a*,Turner et al. 2004). Polygons in the area-detailed inset represent boundaries of *ejidos*.
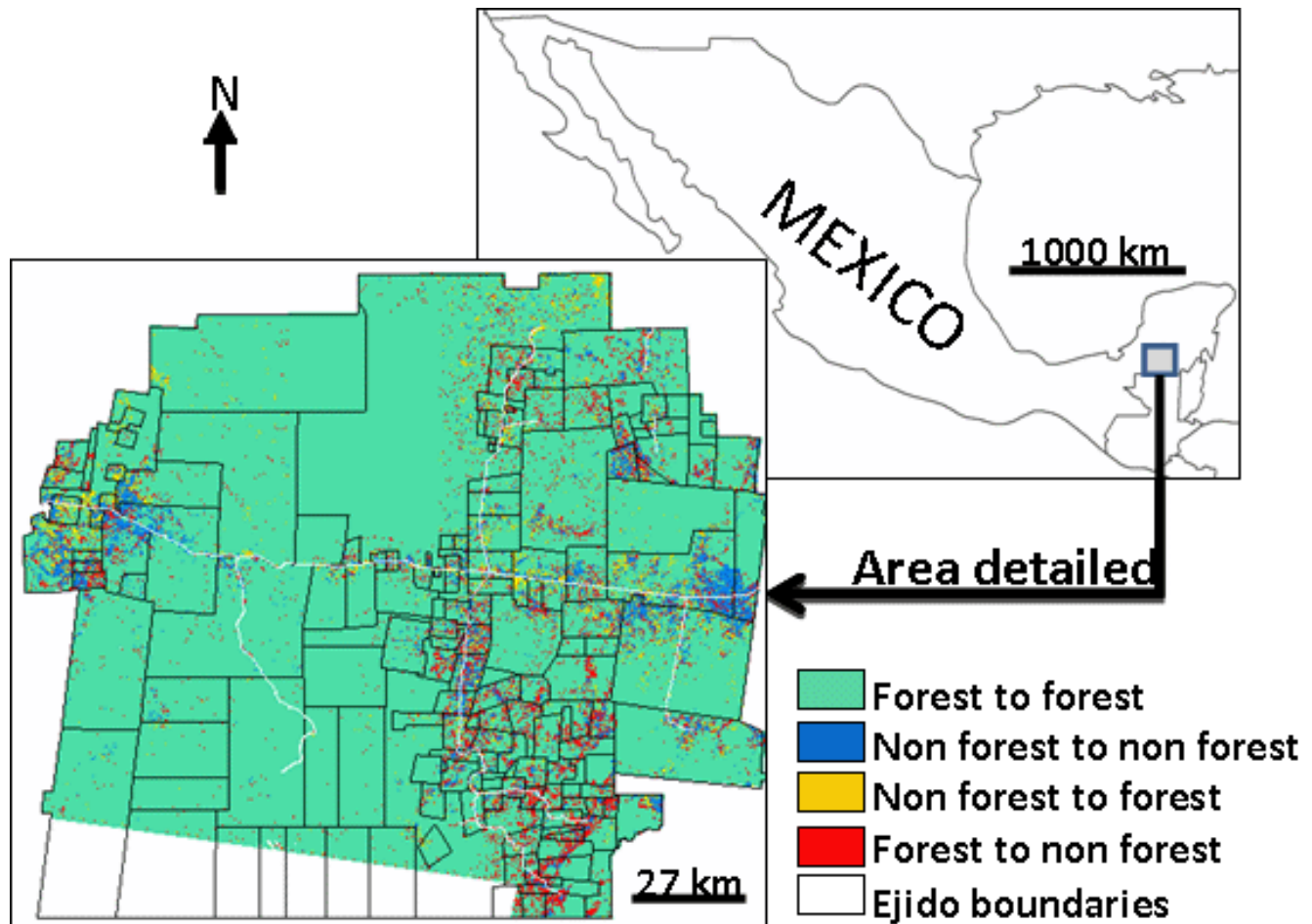
socioeconomic determinants and diversity patterns in traditional vs. enriched fallows in Calakmul's *ejidal* lands.

DATA AND METHODS

Study *ejidos* and sample design

In order to understand how households allocate parcel areas to forest fallows, the study analyzes empirical data from two *ejidos* located along the northeast and southeast buffer zones of the Calakmul Biosphere Reserve (CBR). The northern and southern *ejidos* were settled by the 1980s by

Chol communities from Chiapas and Mestizo families from Tabasco respectively, and have similar total land areas, i.e., 4340 ha northern, 3979 ha southern. Both communities practice a parcel-based land tenure system, which enables a pooled analysis of land management with a dummy variable to control for *ejido*. Detailed surveys were administered during 2001–2002 to randomly selected households in each *ejido*, i.e., ~30% of all households, recording data on demography, land tenure, socioeconomics, political-institutional engagement, and land management. Surveys included GPS-assisted land parcel mapping, deriving areas in summer- and winter-cycle *milpa*, jalapeño chili, pasture, traditional fallows, enriched

Fig. 2. Regional elevation above mean sea level (m) and study *ejido* locations. Polygons represent boundaries of *ejidos* in the region.
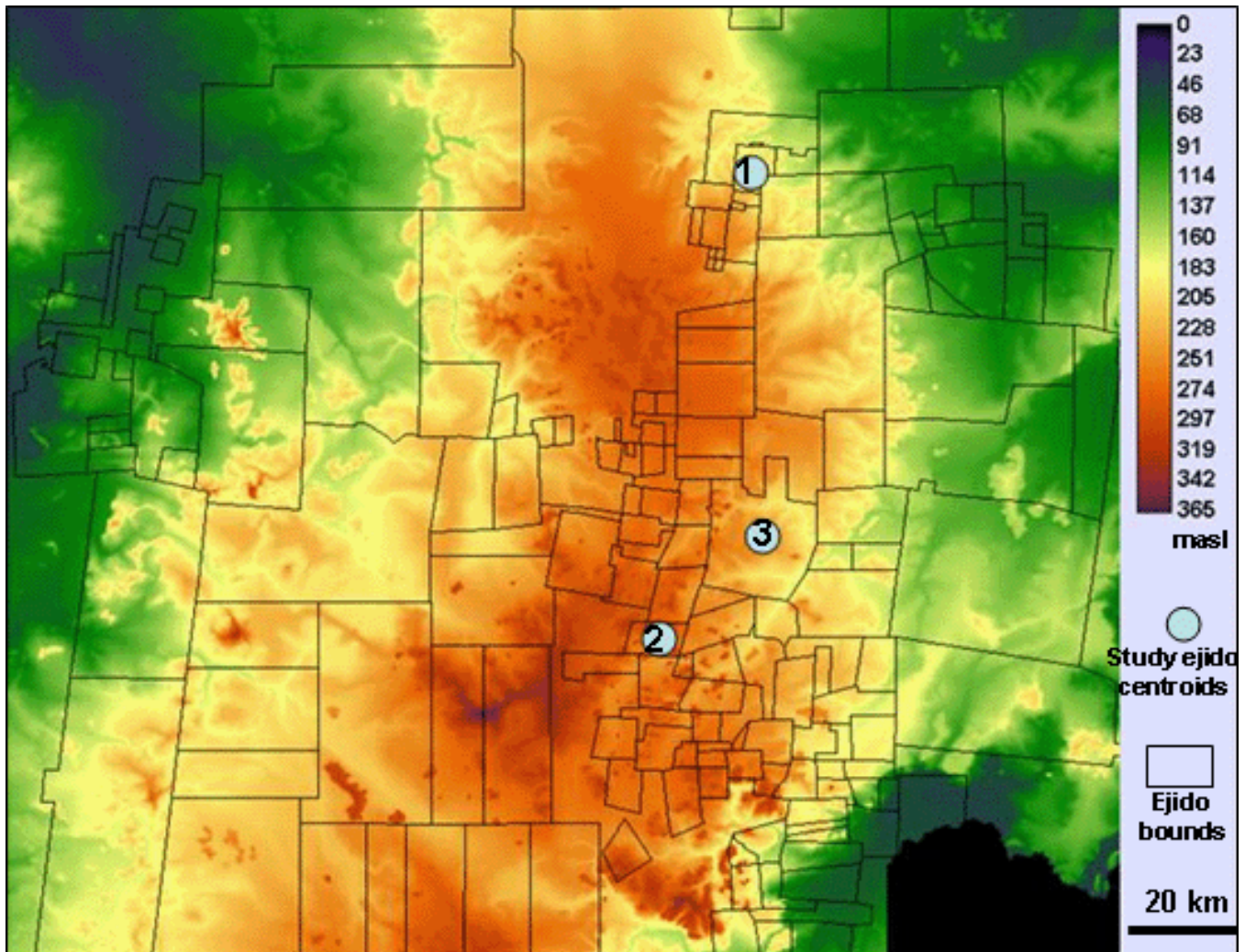
fallows, and state and NGO subsidized *roza-pica-siembra* (RPS, or zero-burn) *milpa*. linear regression model was used to analyze the number of hectares of traditional and enriched fallows for each household as a function of the listed explanatory variables (see Table 1). Zellner's Seemingly Unrelated Regression (SUR), a multiple equation modeling technique, adjusts for the fact that allocations to different land uses are not independent of one another, and their residuals may be contemporaneously correlated. SUR produces asymptotically more efficient estimates of

regression coefficients than equation by equation estimates produced by classical least squares approaches, with greatest gains in efficiency when residuals are highly correlated while independent variables are not (Zellner 1962, 1963).

Tree species diversity was surveyed in regenerating fallows on selected household parcels in the above two *ejidos*. Additional fallow plots were selected from a third, older *ejido* located midway and slightly to the east of the aforementioned *ejidos*, and established by Maya immigrants from Dzitbalché,

Campeche. Agricultural lands in the third *ejido* are non-parcelized; therefore, it was not included in the pooled household model. Once area allocations are made, diversity patterns are then structured by fallow management practices, e.g., traditional vs. enriched, and site-specific environmental factors. All three *ejidos* are broadly similar in topography and elevation, and abound in both traditional and enriched fallows, facilitating biodiversity comparisons between the two management treatments.

From all possible fallow forest sites considered initially, a total of 6 traditional and 11 enriched sites were selected for biodiversity surveys. These sites satisfied control criteria to reduce site-specific variations in (1) fallow stages between 6 and 12 yr, (2) land use history, e.g., location on previous *milpa* agricultural plot eliminating former pastures, chili, and other non-*milpa* croplands; at most two previous crop-fallow cycles on the fallow plot, (3) minimum areas of at least 1 ha, and (4) local topographic-edaphic factors such as their location on upland mollisols.

Field sampling was done during February–April 2002 using a nested strategy (see Read and Lawrence 2003), which recorded data at two distinct scales: 10×10 m plots and 2×2 m quadrats. The larger plots were placed randomly within each traditional or enriched fallow forest site that was at least 1 ha in size, at a minimum of 10 m distance from the edge of the fallow (Fig. 3). Plot measurements recorded all trees with diameters at breast height (dbh) greater than 5 cm, identifying their species with the help of a local botanical expert. Additionally, four quadrats measuring 2×2 m were systematically placed within the 100 m² plots for intensive sampling of smaller trees with dbh greater than 1 cm (Fig. 3). Relative estimates of tree species abundance in all subsequent indices of diversity presented are measured in tree basal area (m², normalized to 1 ha). For quadrat-scale comparisons, areal abundances of tree species for the four quadrats were summed to derive one value for each species identified at the site level, which was then used to calculate one quadrat-based index/site to avoid sample pseudoreplication. Finally, tree species diversity in traditional and enriched fallows was compared using indices of species richness and diversity, including Shannon-Wiener and Simpson's indices, and analyses of variance (ANOVA). Soil samples were also derived at each site, the results of which will be discussed elsewhere.

Diagnostic tests included variable and residual plots, Smirnov-Kolmogorov tests for normality, Breusch-Pagan and Levene's tests for equality of variances, and a Durbin-Watson test for residual autocorrelation. Outlier removal eliminated 2 households, and yielded 29 households for analysis. The diagnostics largely indicated that the sample data satisfied basic conditions for statistical analyses, i.e., normality, linearity, homogeneity of variance, and residual autocorrelation. Exceptions are noted and addressed in the ensuing section.

RESULTS

Household and institutional determinants of fallow forest allocations

The household fallow allocation model investigates household and institutional determinants of spatial extents in traditional and enriched fallows. Summary statistics are reported in Table 1. At the time of the surveys, households had been residing in the two *ejidos* for an average of 15 yr, with approximately 60 ha of land access. The ratio of laborers to consumers in an average household approximated 0.33, relative to a possible maximum value of 1. Households followed diverse livelihood options, including animal husbandry, which ranged from 175 pesos to over 130,000 pesos in value of livestock holdings, market sales of jalapeno chili, with an average annual income of 6567 pesos, and up to 12 additional sources of off-farm wage income. They varied widely in their dependence on primary and secondary forests for timber, fuel wood and nontimber products. They also differed in their access to federal and local subsidies/programs for various land uses. For instance, PROCAMPO inscriptions ranged from 1.5 to 9 ha, RPS inscriptions from 0 to 4 ha, and agroforestry/ reforestation inscriptions for fallow enrichment from 0 to 8 ha.

Overall, a household's hectares in traditional (R^2=0.8501, Chi-square=164.48, *P*=0.0000) and enriched fallows (R^2=0.8876, Chi-square=229.07, *P*=0.0000) are well explained by the model (Table 1). Table 1 reports the SUR results, including standardized coefficients of estimation for each variable's effect on fallow area. A household's tenancy is a strong, positive, and significant predictor of its area under traditional fallows. Tenancy appears to be positively linked to enriched fallow areas as well, although the relationship is not

Fig. 3. Nested sampling schema.
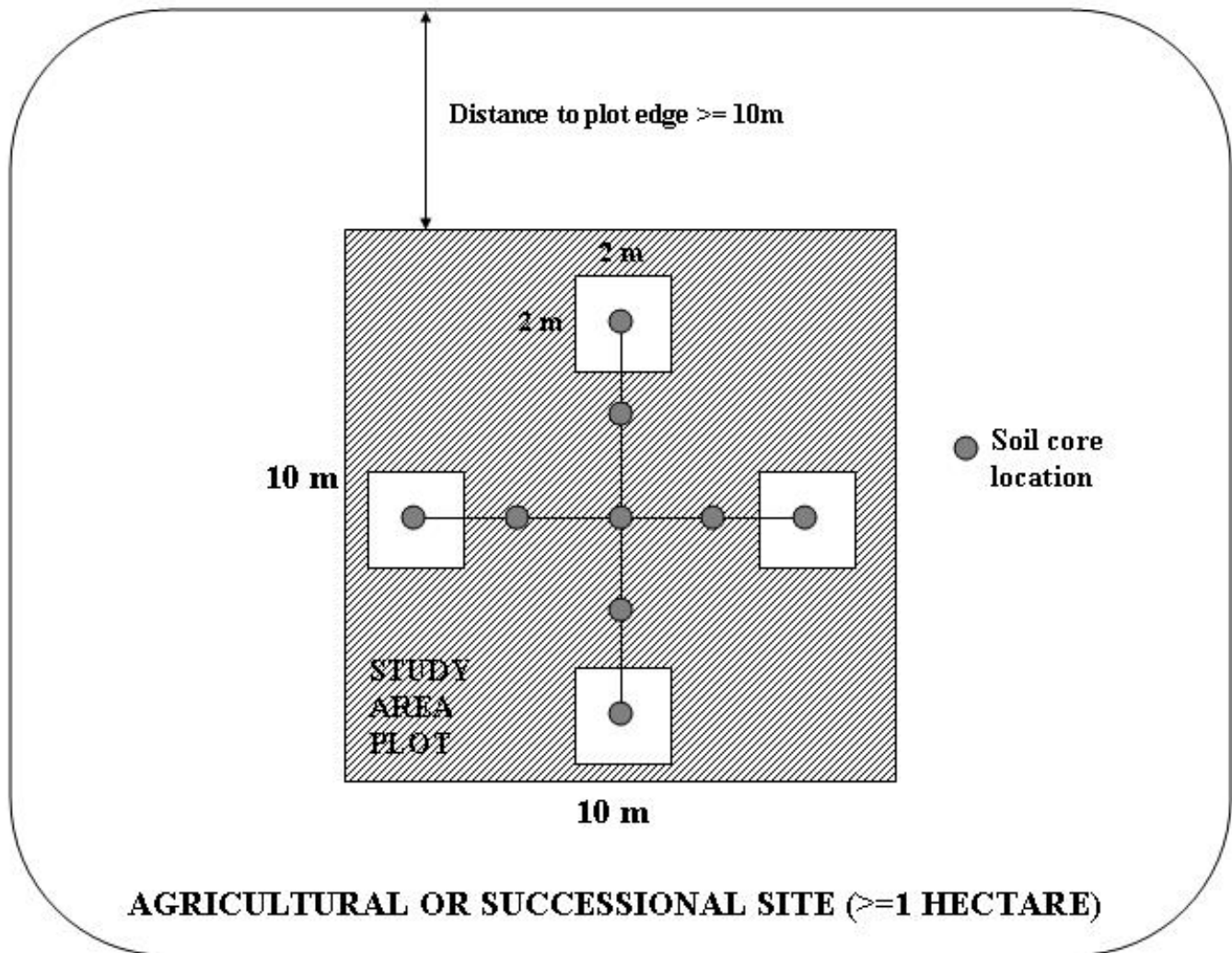
statistically significant. Households' total land entitlement, although positively linked to both types of fallows, was not statistically significant. Labor-consumer ratios, on the other hand, strongly and significantly explain traditional and enriched fallow allocations. Traditional fallows are significantly more extensive among households with larger holdings in livestock, and positively but not significantly related to their previous years' sales of chili. Enriched fallows are more extensive among households that are less invested in chili farming as indicated by their chili sales, although the relation is not statistically significant either. However, only

household forest dependence had a statistically significant effect on successional forest, with links to larger traditional fallows.

Institutional programs have interesting implications for areas under fallow. For instance, households with greater areas subsidized by PROCAMPO since 1994 held significantly larger areas in traditional fallows, but smaller areas in enriched fallows. Household parcel areas inscribed under the subsidized RPS program were significantly linked to smaller traditional fallow holdings, with an apparent positive effect on enriched fallows that was

Table 1. Summary statistics of household and institutional variables and results of Seemingly Unrelated Regression for traditional and enriched fallow allocations. Cells list standardized coefficients. *P=0.10, ** P=0.05, *** P=0.01.

Variable	Mean (SD)	Range	Traditional fallows Z-score, Estimated coefficient, (95% Confidence Interval)	Enriched fallows Z-score, Estimated coefficient, (95% Confidence Interval)
Constant term			**-3.49*** -17.48 (-27.30, -7.67)	**-1.80*** -1.21 (-2.52, 0.11)
Tenancy (# years with land rights)	15.00￼(6.02)	8-25	**2.83*** 0.75 (0.23, 1.27)	1.20 0.04 (-0.03, 0.11)
Land entitlement (ha)	59.31 (13.61)	40-80	1.61 0.12 (-0.03, 0.27)	1.03 0.01 (-0.01, 0.03)
Labor-consumer ratio	0.33 (0.23)	0.13-1.00	**6.00*** 24.02 (16.17, 31.86)	**2.39**** 1.28 (0.23, 2.33)
Net worth in livestock holdings (Mexican pesos)	8153.28 (24,090.33)	175-13,0595	**2.95*** 0.00012 (0.00004, 0.00020)	1.27 0.000007 (-0.000004, 0.000018)
Income from chili sales in previous year (Mexican pesos)	6566.90 (8399.34)	0-32,400	0.99 0.0001 (-0.0001, 0.0004)	-1.01 -0.00002 (-0.00005, 0.00002)
# Off-farm direct wage sources tapped	2.93 (3.17)	0-12	1.38 0.37 (-0.15, 0.89)	0.94 0.03 (-0.04, 0.10)
Relative intensity and commercialization of household forest use, e.g., timber, fuel wood, nontimber forest products (qualitative index)	3.79 (1.99)	1-8	0.14 0.07 (-0.94, 1.08)	**2.18**** 0.15 (0.01, 0.29)
PROCAMPO (ha inscribed)	4.36 (2.03)	1.5-9	**2.72*** 1.27 (0.35, 2.18)	**-2.45**** -0.15 (-0.28, -0.03)

RPS (ha inscribed)	1.62 (1.05)	0-4	**-1.80*** -1.63 (-3.40, 0.14)	1.40 .17 (-0.07, 0.41)
Fallow enrichment and agroforestry (ha inscribed)	0.69 (1.61)	0-8	-0.14 -0.09 (-1.37, 1.18)	**10.23***** 0.89 (0.72, 1.06)
Ejido dummy	0.45 (0.51)	0-1	-1.35 -4.58 (-11.20, 2.05)	-0.03 -0.01 (-0.90, 0.88)

not statistically significant. The largest effect detected in the model was that of participation in local agroforestry/reforestation programs, with a standardized score of 10.23, significantly increasing areas under enriched fallows. There were no statistically significant differences in overall fallow allocations between the *ejidos*.

Biodiversity in traditional and enriched forest fallows

As mentioned previously, site-specific environmental and land use variations were minimized during the selection process. It was not possible to completely eliminate variation in numbers of *milpa* crop-fallow cycles (up to 2 cycles), or in fallow age (6–12 yr), among the sites. However, rank-abundance plots and regression analyses revealed no significant confounding effects by either of these two factors (age effect in traditional fallows: R^2=0.005, $F(1, 41)$=0.212, P=0.648, crop-cycle effect in traditional fallows:R^2=0.002, $F(1, 41)$=0.099, P=0.755, age effect in enriched fallows: R^2=0.002, $F(1, 62)$=0.114, P=0.737, crop-cycle effect in enriched fallows:R^2=0.001, $F(1, 62)$=0.047, P=0.828). The plots were therefore considered suitable for a comparison of species diversity differences due to their management as traditional or enriched fallows.

Species richness

Quadrat-scale counts of trees with dbh&gt1 cm revealed no major differences in species richness between traditional (maximum observed richness of 10 species) and enriched fallows (9 species). At the plot scale, enriched successions had the maximum observed species richness (13 species, represented among 38 trees with dbh&gt5 cm). The site that recorded this high number of species was also the oldest successional site studied (12 yr), and had experienced one previous crop-fallow cycle. Within the traditional fallows sampled, a 6-yr-old fallow site that had undergone two previous crop-fallow cycles recorded the highest species richness (10 species represented among 22 trees with dbh&gt5 cm). At the plot scale, enriched fallows had a mean of 5.82 (SE=1.10) tree species vs. 7.17 (SE=1.05) in traditional fallows, whereas at the quadrat scale, enriched fallows had a mean of 4.45 (SE=0.98) trees species to 6.5 (SE=0.76) tree species in traditional fallows. However, enriched and traditional fallows did not differ significantly in species richness at plot (ANOVA, $F(1, 15)$=0.636, P=0.438) or quadrat scales (ANOVA, $F(1,15)$=1.993, P=0.178).

Species evenness

Rank-abundance curves are useful as simple representations of species richness and evenness. As species abundance ranks, (rank=1 indicating the most abundant species), are plotted on the X-axis and their relative abundance on the Y-axis, steeper curves indicate low evenness, as high ranking species are much more abundant than low-ranking species. Rank-abundance curves for traditional (Fig. 4) and enriched fallow plots (Fig. 5) are comparable. The Shannon-Wiener index (H') of species heterogeneity was used to assess species richness vs. abundance, as it captures both dimensions of biodiversity (Magurran 1988). Figure 6 captures differences in mean values of H' between traditional and enriched fallows at the plot

and quadrat scales. There were no significant differences in H' between traditional and enriched fallows at the scale of the larger plot ($F(1, 15)=1.075$, $P=0.316$, Fig. 6). Diagnostic tests revealed that traditional and enriched fallows sampled at the quadrat scale had unequal variances in H' (Levene statistic $(1, 15)=8.108$, $P=0.012$), however, robust tests for equality of means revealed that traditional fallows had significantly higher heterogeneity (Brown-Forsythe statistic=13.77, $P=0.002$). Simpson's index of diversity (1-D, where D=Simpson's Dominance), was also analyzed, although the data revealed no statistically significant patterns (Fig. 6). Traditional fallows displayed lower species dominance at plot ($F(1, 15)=1.086$, $P=0.314$) and quadrat ($F(1, 13)=1.781$, $P=0.205$) scales, whereas enriched successions were more variable, and most sites exhibited higher species dominance than in traditional fallows.

DISCUSSION

Household and institutional determinants of fallow forest allocations

Household tenure, demography, and livelihood strategies play important roles in decisions to allocate parcels lands to traditional and enriched fallows. Roy Chowdhury and Turner (2006) provide a more in-depth discussion of how household and structural factors drive agricultural land uses. Tenancy reflects an aspect of the household's life cycle in the region, a factor frequently found to strongly influence fallowing decisions in the neotropics (e.g., see Moran 1989, Walker and Homma 1996, Coomes et al. 2000, Perz and Walker 2002). The positive link between tenancy and traditional fallows is paralleled in a study by Abizaid and Coomes (2004) in another Calakmul *ejido* that linked the age of the *ejidatario* head of household to fallow holdings. Larger labor-consumer ratios imply greater labor availability for agricultural and other activities, and thereby explain the higher cumulative holdings of traditional fallows. The maintenance tasks that fallow enrichment entails also require additional labor, explaining the significant relationship between a household's labor-consumer ratio and its fallow enrichment.

Other aspects of households' economic strategies are also linked to their fallowing decisions. Livestock translate into larger traditional fallows,

presumably through larger feed demand (and planted, then fallowed crop area) over time, although it is important to note that the data do not distinguish among recent livestock acquisitions vs. long-term holdings. The fact that the previous years' chili sales do not significantly affect fallow allocations is unexpected, other aspects of land management practices under chili may nonetheless have important implications for fallows. The absence of a statistically significant connection between forest dependence and traditional fallows may reflect the fact that enriched fallows, which are positively and significantly linked to forest dependence, or old-growth forests are often the sources for such products. Also, secondary forest dependence may be more related to the quality of traditional fallows than their spatial extent.

The models reveal that institutional policies strongly influence fallowing decisions in Calakmul's *ejidos*. PROCAMPO has been linked to deforestation and total land holdings in the region (e.g., see Klepeis and Vance 2003, Abizaid and Coomes 2004). Although Abizaid and Coomes (2004) did not find a statistically significant relationship between PROCAMPO payments and parcel fallow area, this analysis finds a positive relationship. Despite the fact that PROCAMPO payments are made ostensibly for a spatially fixed area under cultivation, most households have practiced and continue shifting cultivation in the region, and continue to receive the fixed subsidy payment despite having relocated their agricultural fields. Over time, the shifting pattern generates a cumulatively larger area in traditional fallows. This spatially extensive agricultural practice correlates with the link between PROCAMPO inscriptions and smaller areas in the more intensively managed enriched fallows.

Conservation interventions such as RPS inscriptions have a weakly significant, negative effect on traditional fallows, and no significant effect on enriched fallows. This can be attributed to several factors: (1) the program was launched in the late 1990s and thus did not have enough time to appreciably impact fallow holdings by 2001–2002, (2) the program focus emphasizes green fertilizers over fallow enrichment, and discourages the establishment of traditional fallows, and (3) the program had various structural failures, e.g., poor quality saplings provided at the wrong time for planting during the dry season, and at one point created an artificial market for the promoted green

**Fig. 4.** Plot-scale (dbh>5 cm) dominance-diversity curves in traditional fallows.
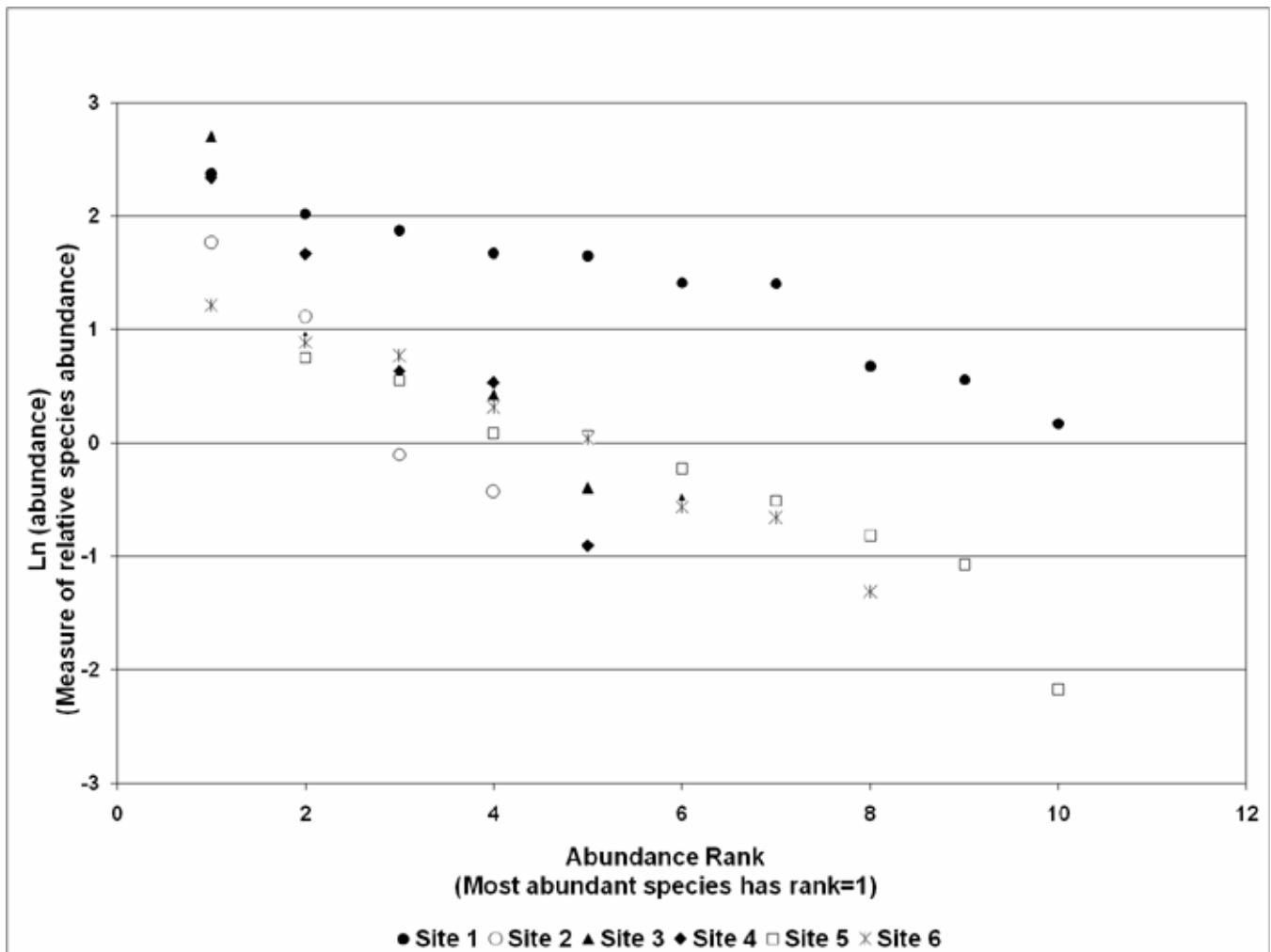
fertilizer crops such that farmers adopted the practice to raise and sell the crop for the state, rather than for its purported fertilization or agroforestry benefits. Not surprisingly, the data indicate that household inscriptions in agroforestry/reforestation projects, which also promote fallow enrichment, are strongly and significantly linked to enriched fallow area.

Biodiversity in traditional and enriched forest fallows

Rank-abundance plots generally reveal important information about species communities. Lawrence (1998) found, for instance, that the slope of the log (rank)/log(abundance) plot becomes increasingly negative as relative abundances of dominant species increase and community evenness decreases with repeated cycles of swidden agriculture in Kalimantan. The similarly shaped curves for traditional and enriched fallows indicate that their patterns of tree species evenness are comparable at the plot scale. The statistical results obtained for

**Fig. 5.**. Plot-scale (dbh>5 cm) dominance-diversity curves in enriched fallows.
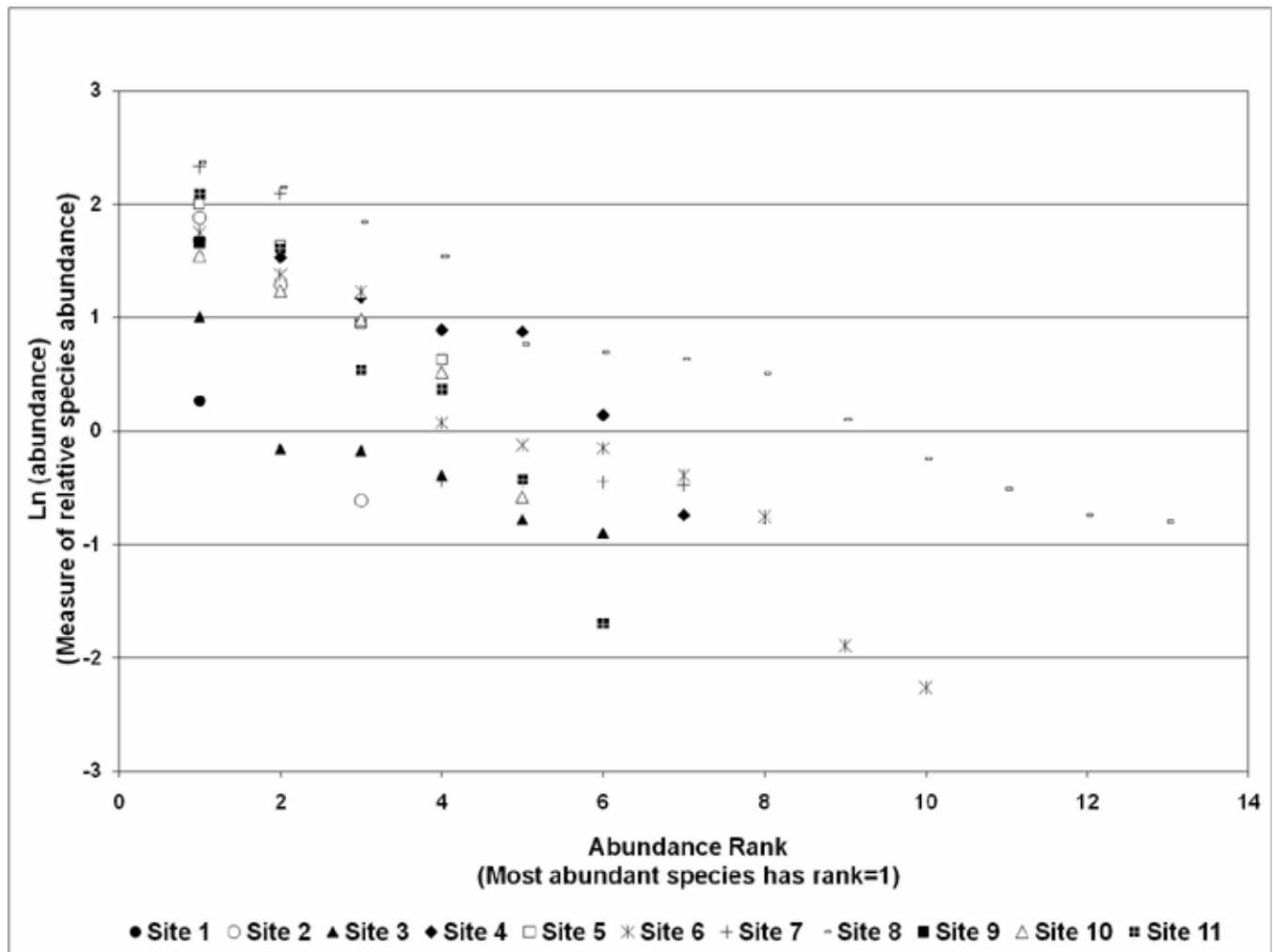
Shannon-Wiener and Simpson's diversity indices confirm the plot-scale similarity between traditional enriched successions. However, the analyses also reveal that traditional fallows have significantly higher species heterogeneity at the scale of the quadrats, in which smaller trees were also recorded.

These results highlight the complex and changing relationship of species heterogeneity to fallow management over time. Enriched successions are typically characterized by a higher frequency of disturbance in the form of clearing light breaks along planted rows, and/or weeding around economically desirable species. The effects of such disturbance may be expected to penetrate to other areas of the succession, producing microenvironments in which (1) species that are selectively managed may increase in dominance, or (2) light-loving or early successional stage species favored by the high disturbance frequency increase in dominance. Aside from planted tree species such as cedar, mahogany, allspice and fruit trees, farmers often preferentially maintain other naturally occurring trees species for their economic value. These include palms, naturally occurring allspice (*pimienta*), chicozapote (*Manilkara zapota*), and other tree species that provide material for thatching, lesser-known timber species, or

Fig. 6. Shannon-Wiener and Simpson's indices of diversity at plot (dbh>5 cm) and quadrat (dbh>1 cm) scales.
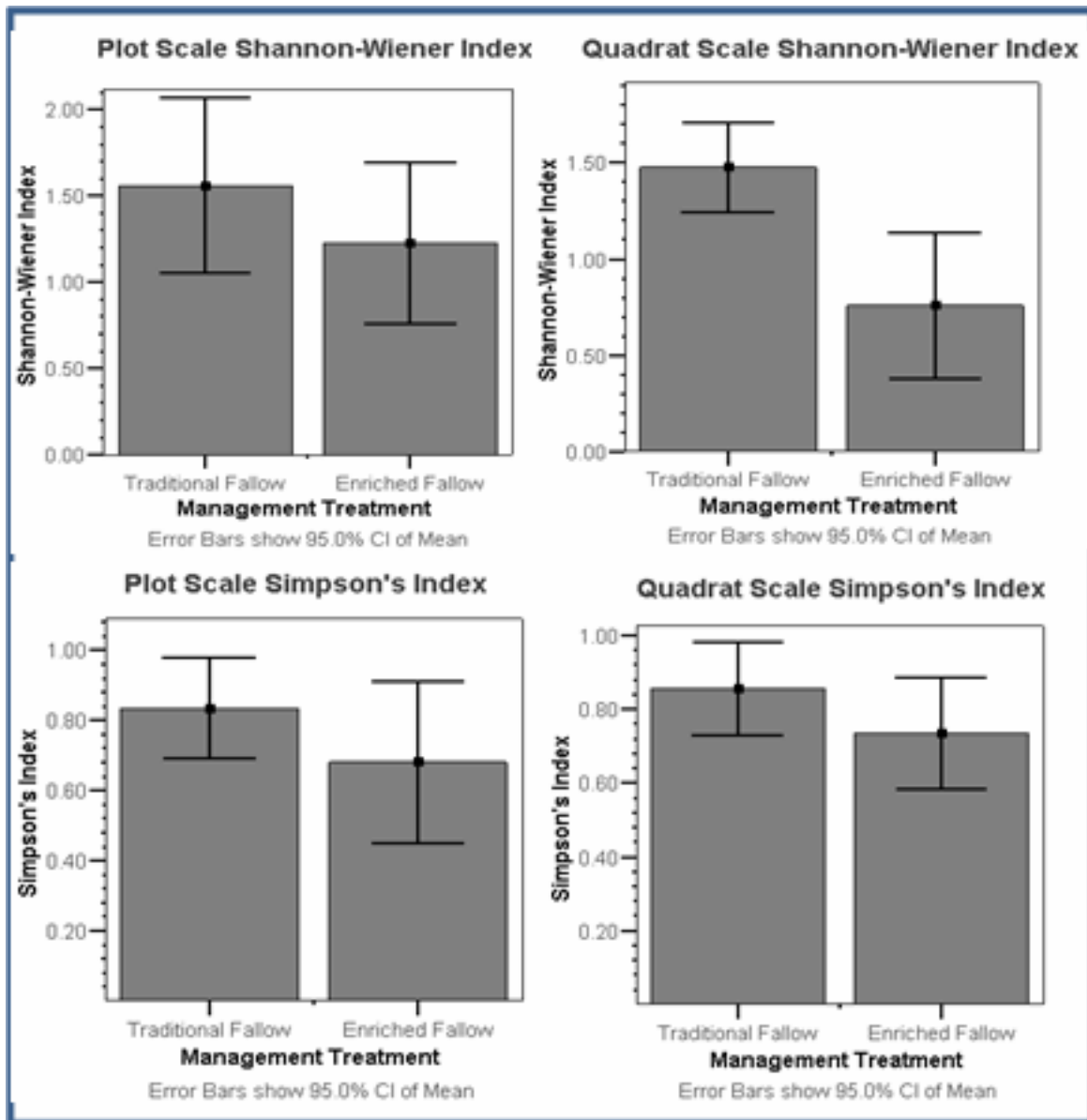
nontimber forest products. Many of the farmers that were interviewed engaged in apiculture, and maintained apiaries near or in successional forests, because several of the flowering trees that are found in such sites are valued for the quality and profusion of their pollen.

CONCLUSION

The southern Mexican landscape, a hotspot of deforestation as well as biotic diversity, is experiencing increasing rates and extents of secondary succession following the deforestation waves that ensued in the 1960s. Successional forests hold important implications for the sustainability

and restoration of the region's forested ecosystems and the livelihoods of Calakmul's largely ejidal communities. The findings of the data and analysis presented here indicate the complexity of factors driving the generation, maintenance and composition of traditional and enriched successions in the region. Fallow management decisions have biodiversity consequences, leading to reduced species heterogeneity among smaller, early-stage trees in enriched forest plots compared to traditional fallows.

Numerous studies caution against overestimating the ecological benefits of successional communities, citing divergence between mature and successional sites in floristic composition (Packham et al. 2005), slowing rates of biomass accumulation in older successions (Vieira et al. 2003), low accretion of species diversity, and/or lower heterogeneity despite high species richness in successions (Turner et al. 1994, Ohsawa 2005, Silva et al. 2007). Nevertheless, secondary forests represent valuable biodiversity and regeneration potential if properly managed, even in landscapes of shifting cultivation (Makana and Thomas 2006). Fallow conservation interventions such as those examined in this study are also common in neighboring Mayan *ejidos* of Quintana Roo, where enrichment plantings of commercially valuable timber species have been found to aid in the conservation of older successions and their eventual transition to mature forests (Dalle et al. 2006).

Up to 80% of Mexico's forests are communally held (Bray et al. 2003), this further necessitates a focus on both successional and old growth forests under *ejidal* land tenure systems. As the fundamental decision-making agents driving land uses in the region, *ejido* households merit particular attention in studies aiming to explicate relationships among land use, landscape, and the fate of the forest. This study shows that household socioeconomic characteristics influence their parcel-scale fallow forest extents. Structural forces such as agricultural and conservation-based subsidy programs and interventions also strongly mediate fallow production and significantly, management. The region's biodiversity potential and the reach of national and international conservation regimes have engendered a suite of conservation experiments with varying degrees of success. Among such interventions are institutionally promoted soil conservation and fallow enrichment practices, that have been adopted to varying degrees

by smallholding households. Formal household inscriptions in such projects are the strongest predictor of whether households undertake fallow enrichment practices, yet, there remains significant variation in household fallow management even after controlling for program inscriptions.

Land change research as well as conservation regimes need to carefully examine household demographics and economic strategies before they can predict how conservation interventions may affect decision making by local land managers. The ecological results presented here indicate that species diversity of traditional and enriched fallows are broadly similar, however, there is some evidence that enriched fallows exhibit lower species heterogeneity and higher species dominance compared to traditional forest successions, at least among younger and smaller tree species. It is noteworthy that although there is evidence of successful adoption and practice of fallow enrichment, spatially explicit studies of land change indicate that such programs may also be linked to greater probabilities of deforestation elsewhere in parcels (Roy Chowdhury 2006*b*). The impacts of structural factors and policy interventions vary across households and spatial/ecological scales, necessitating both social and ecological analyses for policy evaluation and design in the interest of ecological and livelihood sustainability.

Responses to this article can be read online at:
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members for their assistance. Sincere thanks are also due to the editors and anonymous reviewers of this manuscript.

LITERATURE CITED

Abizaid, C., and O. T. Coomes. 2004. Land use and forest fallowing dynamics in seasonally dry tropical forests of the Southern Yucatán Peninsula, Mexico. *Land Use Policy* **21**:71-84.

Achard, F, H. Eva, A. Glinni, P. Mayaux, T. Richards, and H. J. Stibig. 1998. Identification of deforestation hot spot areas in the humid tropics. *TREES Publications Series B. Research Report No. 4.* Luxemburg, European Commission.

Acopa, D., and E. Boege. 1998. The Maya forest in Campeche, Mexico: experiences in forest management at Calakmul. Pages 81-97 *in* R. D. Primack, D. B. Bray, H .A. Galletti, and I. Ponciano, editors. *Timber, tourists, and temples: conservation and development in the Maya forest of Belize, Guatemala, and Mexico.* Island Press, Washington, D.C., USA.

Anderson, A. B., P. H. May, and M. J. Balick. 1991. *The subsidy from nature: palm forests, peasantry, and development on an Amazon frontier.* Columbia University Press, New York, New York, USA.

Angelsen, A., and D. Kaimowitz. 1999. Rethinking the causes of deforestation: lessons from economic models. *The World Bank Research Observer* **14**:73-98.

Antinori, C., and D. B. Bray. 2005. Community forest enterprises as entrepreneurial firms: economic and institutional perspectives form Mexico. *World Development* 33(9):1529-1543.

Barrow, C. J. 1991. *Land degradation: development and breakdown in terrestrial environments.* Cambridge University Press, Cambridge, UK.

Bassett, T. J., and K. S. Zimmerer. 2003. Cultural ecology. Pages 97-112 *in* G. L. Gaile and C. J. Wilmott, editors. *Geography in America at the dawn of the new millennium.* Oxford University Press, Oxford, UK.

Bray, D. B., L. Merino-PÉrez, P. Negreros-Castillo, G. Segura-Warnholtz, J. M. Torres-Rojo, and H. F. M. Vester. 2003. Mexico's community-managed forests as a global model for sustainable landscapes. *Conservation Biology* **17**(3):672-677.

Brookfield, H. 1964. Questions on the human frontiers of geography. *Economic Geography* **40**293-303.

Brown, S., and A. E. Lugo. 1990. Tropical secondary forests. *Journal of Tropical Ecology* **6**:1-32.

Cincotta, R. P., and R. Engelman. 2000. *Nature's place: human population and the future of biological diversity.* Population Action International, Washington, D.C., USA.

Coomes, O. T., F. Grimard, and G. J. Burt. 2000. Tropical forests and shifting cultivation: secondary forest fallow dynamics among traditional farmers of the Peruvian Amazon. *Ecological Economics* **32**:109-24.

Corlett, R. T. 1995. Tropical secondary forests. *Progress in Physical Geography* **19**(2):159-172.

Daily, G. C., T. Söderqvist, S. Aniyar, K. J. Arrow, P. Dasgupta, P. R. Ehrlich, C. Folke, A. Jansson, B. O. Jansson, and N. Kautsky. 2000. The value of nature and nature of value. *Science* **289**:395-396.

Dalle, S. P., T. Johns, S. De Blois, and J. Caballero. 2006. Integrating analyses of local land-use regulations, cultural perceptions and land-use/land cover data for assessing the success of community-based conservation. *Forest Ecology and Management* **222**(1-3):370-383.

de Jong, W., U. Chokkalingam, J. Smith, and C. Sabogal. 2001. Tropical secondary forests in Asia: introduction and synthesis. *Journal of Tropical Forest Science* **13**(4):563-576.

Denevan, W. M. 1983. Adaptation, variation, and cultural geography. *Professional Geographer* **35**:399-407.

DeFries, R., C. Field, I. Fung, G. Collatz, and L. Bounoua. 1999. Combining satellite data and

biogeochemical models to estimate global effects of human-induced land cover change on carbon emissions and primary productivity. *Global Biogeochemical Cycles* **13**:803-815.

Edwards, C. 1957. *Quintana Roo, Mexico's empty quarter.* University of California, Berkeley, California, USA.

Food and Agriculture Organization (FAO). 1996. *Forest resources assessment 1990: survey of tropical forest cover and study of change processes.* FAO Forestry Paper 130, Rome, Italy.

Food and Agricultural Organization (FAO). 1999. *State of the world's forests, 1999.* FAO, Rome, Italy.

Flores Guido, J. S. 1987. *Uso de los Recursos Vegetales en la Peninsula de Yucatán: Pasado, Presente y Futuro*. INIREB, Xalapa.

Fox, J., R. R. Rindfuss, S. J. Walsh, and V. Mishra, editors. 2003. *People and the environment: approaches for linking household and community surveys to remote sensing and GIS.* Kluwer Academic, Boston, Massachusetts, USA.

Fuller, D. O. 2006. Tropical forest monitoring and remote sensing: a new era of transparency in forest governance? *Singapore Journal of Tropical Geography* **27**(1):15-29.

Geoghegan, J., S. Cortina-Villar, P. Klepeis, P. Macario-Mendoza, Y. Ogneva-Himmelberger, R. Roy Chowdhury, B. L. Turner II, and C. Vance. 2001. Modeling tropical deforestation in the southern Yucatán peninsular region: comparing survey and satellite data. *Agriculture, Ecosystems and Environment* **85**:25-46.

Grossman, L. 1977. Man-environment relationships in anthropology and geography. *Annals of the Association of American Geographers* **67**:126-44.

Hecht, S. 2004. Invisible forests: the political ecology of forest resurgence in El Salvador. Pages 64-104 *in* R. Peet and M. Watts, editors. *Liberation ecologies: environment, development, and social movements.* Routledge, London, UK.

Houghton, R. A., D. L. Skole, C. A. Nobre, J. L. Hackler, K. T. Lawrence, and W. H. Chomentowski. 2000. Annual fluxes of carbon from deforestation and regrowth in the Brazilian Amazon. *Nature* **403**:301-304.

Kasperson, J. X., R. E. Kasperson, and B. L. Turner, II, editors. 1995. *Regions at risk: comparisons of threatened environments.* United Nations University Press, Tokyo, Japan.

Kauppi, P. E., J. H. Ausubel, J. Fang, A. S. Mather, R. A. Sedjo, and P. E. Waggoner. 2006. Returning forests analyzed with the forest identity. *Proceedings of the National Academy of Sciences of the United States of America* **103** (46):17574-17579.

Klepeis, P., and Turner, B. L., II. 2001. Integrated land history and global change science: the example of the southern Yucatán peninsular region project. *Land Use Policy* **18**(1):27-39.

Klepeis, P., C. Vance, E. Keys, P. Macario Mendoza, and B. L. Turner II. 2004. Subsistence sustained: swidden or *Milpa* cultivation. Pages 189-206 *in* B. L. Turner II, J. Geoghegan, and D. Foster, editors. *Integrated land-change science and tropical deforestation in the southern Yucatán: final frontiers.* Clarendon Press, Oxford, UK.

Kummer, D. M. 1992. Remote sensing and tropical deforestation: a cautionary note from the Philippines. *Photogrammetric Engineering and Remote Sensing* **58**:1469-1471.

Lawrence, D., and D. Foster. 2002. Changes in forest biomass, litter dynamics and soils following shifting cultivation in southern Mexico: an overview. *Interciencia* **27**(8):400-408.

Lugo, A., and S. Brown. 1992. Tropical forests as sinks of atmospheric carbon. *Forest Ecology and Management* **54**:239-255.

Lundell, C. L. 1933. Chicle exploration in the Sapodilla forest of the Yucatan Peninsula. *Field and Laboratory* **21**:15-21.

Lundell, C. L. 1934. *Preliminary sketch of the phytogeography of the Yucatan peninsula.* Carnegie Institute of Washington, Publication Number 436. Carnegie Institute, Washington, D.C., USA.

Magurran, A. 1988. *Ecological diversity and its measurement.* Princeton University Press, Princeton, New Jersey, USA.

Makana J. R., and S. C. Thomas. 2006. Impacts of selective logging and agricultural clearing on forest structure, floristic composition and diversity, and timber tree regeneration in the Ituri Forest, Democratic Republic of Congo. *Biodiversity and Conservation* **15**(4): 1375-1397.

McCracken, S. D., E. Brondizio, D. Nelson, E. F. Moran, A. D. Siqueira, and C. Rodriguez-Pedraza. 1999. Remote sensing and GIS at farm property level: demography and deforestation in the Brazilian Amazon. *Photogrammetric Engineering and Remote Sensing* **65**(11):1311-1320.

Mooney, H. A., and R. J. Hobbs, editors. 2000. *Invasive species in a changing world.* Island Press, Washington, D.C., USA.

Moran, E. F. 1989. Adaptation and maladaptation in newly settled areas. Pages 20-41 *in* D. Schumann and W. Partridge, editors. *The human ecology of tropical land settlement in Latin America.* Westview Press, Boulder, Colorado, USA.

Moran, E. F., and E. Brondizio. 1998. Land-use change after deforestation in Amazonia. Pages 94-120 *in* D. Liverman, E. F. Moran, R. R. Rindfuss, and P. C. Stern, editors. *People and pixels: linking remote sensing and social sscience.* Committee on the Human Dimensions of Global Environmental Change, National Research Council. National Academy Press, Washington, D.C., USA.

Moran, E. F., E. Brondizio, and S. D. McCracken. 2002. Trajectories of land use: soils, succession and crop choice. Pages 193-217 *in* C. H. Wood and R. Porro, editors. *Deforestation and land use in the Amazon.* University Press of Florida, Gainesville, Florida, USA.

Myers, N. 1994. Tropical deforestation: rates and patterns. Pages 27–40 *in* K. Brown and D. W. Pearce, editors. *The causes of tropical deforestation.* University College of London Press, London, UK.

Ohsawa, M. 2005. Species richness and composition of Curculionidae (Coleoptera) in a conifer plantation, secondary forest, and old-growth forest in the central mountainous region of Japan. *Ecological Research* **20**(6):632-645.

Packham, J. M., A. K. Mills, T. W. May, M. J. Brown, and T. J. Wardlaw. 2002. Macrofungal diversity and community ecology in mature and regrowth wet eucalypt forest in Tasmania: a multivariate study. *Austral Ecology* **27**(2):149-161.

Perz, S. G. 2001. Household demographic factors as life cycle determinants of land use in the Amazon. *Population Research and Policy Review* **20**:159-186.

Perz, S. G., and R. T. Walker. 2002. Household life cycles and secondary forest cover among small farm colonists in the Amazon. *World Development* **30**(6):1009-1027.

Read, L., and D. Lawrence. 2003. Recovery of biomass following shifting cultivation in dry tropical forests of the Yucatan. *Ecological Applications* **13**(1):85-97.

Reforma Agraria. 1998. *Carta de polígonos ejidales, Campeche.* Campeche, Mexico.

Roy Chowdhury, R. 1999. Agroforestry: systems of diversity or divestiture? *Revista Geográfica* **125**:99-112.

Roy Chowdhury, R. 2006*a.* Driving forces of tropical deforestation: the role of remote sensing and spatial models. *Singapore Journal of Tropical Geography* **27**(1):82-101.

Roy Chowdhury, R. 2006*b.* Landscape change in the Calakmul Biosphere Reserve, Mexico: modeling the driving forces of smallholder deforestation in land parcels. *Applied Geography* **26**(2):129-152.

Roy Chowdhury, R., and L. C. Schneider. 2004. Land-cover/use in the southern Yucatán peninsular region, Mexico: classification and change analysis. Pages 105-141 *in* B. L. Turner II, J. Geoghegan, and D. Foster, editors. *Integrated land-change science and tropical deforestation in the southern Yucatán: final frontiers.* Clarendon, Oxford, UK.

Roy Chowdhury, R., and B. L. Turner II. 2006. Reconciling agency and structure in empirical analysis: smallholder land use in the southern Yucatán, Mexico. *Annals of the Association of American Geographers* **96**(2):302-322.

Rudel, T. K., D. Bates, and R. Machinguiashi. 2002. A tropical forest transition? Agricultural change, out-migration, and secondary forests in the Ecuadorian Amazon. *Annals of the Association of American Geographers* **93**:87-102.

Secretaría de Fomento Agropecuario (SEFA). 1991. *Carta de polígonos ejidales, 1:500,000.* Gobierno de Estado de Quintana Roo. SEFA, Dirección de Estudios y Proyectos, Chetumal.

Shukla, J., C. Nobre, and P. Sellers. 1990. Amazon deforestation and climate change. *Science* **247**:1322-1325.

Silva, R. R., R. S. Machado Feitosa, and F. Eberhardt. 2007. Reduced ant diversity along a habitat regeneration gradient in the southern Brazilian Atlantic Forest. *Forest Ecology and Management* **240**(1-3):61-69.

Skole, D., and C. Tucker. 1993. Tropical deforestation and habitat fragmentation in the Amazon: satellite data from 1978–1988. *Science* **260**:1905-1910.

Smith, J., P. van de Kop, K Reategui, I. Lombardi, C. Sabogal, and A. Diaz. 1999. Dynamics of secondary forests in slash and burn agriculture: interactions among land use types in the Peruvian Amazon. *Agriculture, Ecosystems and Environment* **76**:85-98.

Snook, A., and G. Zapata. 1998. Tree cultivation in Calakmul, Mexico: alternatives for reforestation. *Agroforestry Today* **10**(1):15-18.

Southworth, J., and C. Tucker. 2001. The influence of accessibility, local institutions, and socioeconomic factors on forest cover change in the mountains of western Honduras. *Mountain Research and Development* **21**(3):276-283.

Turner II, B. L. 1983. *Once beneath the forest: prehistoric terracing in the Rio Bec region of the Maya lowlands.* Westview Press, Boulder, Colorado, USA.

Turner II, B. L., S. Cortina Villar, D. R. Foster, J. Geoghegan, E. Keys, P. Klepeis, D. Lawrence, P. Macario Mendoza, S. Manson, Y. Ogneva-Himmelberger, A. B. Plotkin, D. PÉrez Salicrup, R. Roy Chowdhury, B. Savitsky, L. Schneider, B. Schmook, and C. Vance. 2001. Deforestation in the southern Yucatán peninsular region: An integrative approach. *Forest Ecology and Management* **154**(3):343-70.

Turner II, B. L., J. Geoghegan, and D. Foster, editors. 2004. *Integrated land-change science and tropical deforestation in the southern Yucatán: final frontiers.* Clarendon, Oxford, UK.

Turner, I. M., P. T. Chew, R. T. Corlett, H. T. W. Tan, Y. C. Wee, and A. Bin Ibrahim. 1994. A study of plant species extinction in Singapore: lessons for the conservation of tropical biodiversity. *Conservation Biology* **8**(3):705-712.

Unruh, J. D. 1988. Ecological aspects of site recovery under swidden-fallow management in the Peruvian Amazon. *Agroforestry Systems* **7**:161-184.

Vieira, I. C. G., C. J. Reis De Carvalho, J. B. Guerrero, A. S. De Almeida, E. A. Davidson, and T. A. Stone. 2003. Classifying successional forests using Landsat spectral properties and ecological characteristics in eastern AmazÔnia. *Remote Sensing of Environment* **87**(4):470-481.

Vitousek, P. M., H. A. Mooney, J. Lubchenco, and J. M. Melillo. 1997. Human domination of earth's ecosystems. *Science* **277**:494-500.

Vogeler, I. K. 1974. The cultural ecological setting of southeastern Campeche. Pages 110-112 *in* R. E. W. Adams, editor. *Preliminary reports on archaeological investigations in the Rio Bec area, Campeche, Mexico.* Middle American Research Institute. Number 31, Tulane University. New Orleans, Louisanna, USA.

Walker, R. T., and A. K. O. Homma. 1996. Land use and land cover dynamics in the Brazilian Amazon: an overview. *Ecological Economics* **18**:67-80.

Walker, R. T., E. F. Moran, and L. Anselin. 2000. Deforestation and cattle ranching in the Brazilian Amazon: external capital and household processes. *World Development* **28**:683-699.

Wilson, E. O. 1988. *Biodiversity.* National Academy Press, Washington, D.C., USA.

Wright, S. J., and H. C. Muller-Landau. 2006. The future of tropical forest species. *Biotropica* **38**(3):287-301.

Zellner, A. 1962. An efficient method of estimating seemingly unrelated regressions and tests for aggregation bias. *Journal of the American Statistical Association* **57**:348-68.

Zellner, A. 1963. Estimators for seemingly unrelated regression equations: some exact finite sample results. *Journal of the American Statistical Association* **58**:977-92.

Zimmerer, K. S. 2004. Cultural ecology: placing households in human-environment studies: the cases of tropical forest transitions and agrobiodiversity change. *Progress in Human Geography* **28** (6):795-806.