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World Development Vol. 32, No. 1, pp. 173–190, 2004

Printed in Great Britain
0305-750X/$ - see front matter

doi:10.1016/j.worlddev.2003.06.014

Deforestation and Carbon Emissions at Tropical Frontiers: A Case Study from the Peruvian Amazon

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Summary. — This paper analyzes the impact of national development policy on land cover change and associated carbon fluxes at a Peruvian Amazon frontier. Remote sensing and field transects reveal changes in forest carbon stocks and accumulation rates. Deforestation was most rapid along the Interoceanic Highway during 1986–91 when credit and guaranteed markets were available, resulting in emissions of 708,000 $Mg\,C\,yr^{-1}$, of which 14% was offset by secondary regrowth. Despite continued population growth, deforestation slowed during 1991–97 when fiscal austerity measures were imposed, resulting in emissions of 389,000 $Mg\,C\,yr^{-1}$, of which 41% was offset by regrowth. Strategies to conserve frontier forests are compared in terms of carbon, biodiversity and economic costs and benefits.

Key words — climate change, deforestation, credit, biodiversity, Kyoto Protocol, Peruvian Amazon

1. INTRODUCTION

Every year, between 1.6 and 2.4 Pg of carbon [1] are released to the atmosphere from tropical forest clearing (De Jong, Ochoa-Gaona, Castillo-Santiago, Ramirez-Marcial, & Carins, 2000; Fearnside, 2000). By these estimates, tropical deforestation accounts for roughly 20–29% of global anthropogenic greenhouse gas (GHG) emissions (Watson *et al.*, 2000). Nowhere is more forest cleared than at Amazon frontiers where >2 million hectares (ha) are lost each year (Laurence, Vasconcelos, & Lovejoy, 2000; Nelson, Kimes, Salas, & Routhier, 2000; Steininger *et al.*, 2001). International environmental groups and government agencies decry the loss of these carbon-heavy, biodiverse forests, yet they cannot agree on how or if tropical forest conservation should be included in climate mitigation strategies (Fearnside, 2001a). Their debate hinges on technical uncertainties as well as political dilemmas.

The direct contribution of tropical deforestation to atmospheric GHG is universally acknowledged, but the *net* role of the tropics in global carbon cycles remains uncertain (De Jong *et al.*, 2000; Houghton *et al.*, 2000; Schimel *et al.*, 2001). For example, some experts identify the Amazon as a carbon sink, potentially absorbing 0.44–0.56 Pg C per year (Grace *et al.*, 1995; Phillips *et al.*, 1998). Others claim the Amazon is neutral with respect to carbon because emissions from deforestation, decomposition, and fire, are balanced by uptake in forests and vegetative regrowth (Houghton *et al.*, 2000; Schimel *et al.*, 2001). Uncertainty regarding the Amazon's role as a net carbon source or sink reflects the limited availability of data on forest biomass stock and uptake rates

* This research was approved by the Peruvian Institute of Natural Resources (INRENA) and was funded by the Wildlife Conservation Society and Conservación Internacional-Peru. The Princeton Environmental Institute and the Center for Applied Biodiversity Science provided support during manuscript preparation. N. Alvarez and M. Steininger generously contributed their expert advice on remote sensing analysis and carbon flux calculations. J. Mena and A. Pinedo helped collect field data. C. Ascorra, D. Bradford, D. Klooster, C. Kremen, E. Ortiz, F. Putz, E. Raez-Luna, A. Treves and two anonymous reviewers improved this paper with their comments. Any errors or omissions are the author's sole responsibility. Final revision accepted: 4 June 2003.

across this ecologically heterogeneous region (Phillips *et al.*, 1998; Watson *et al.*, 2000). In addition, interannual variations in climate and atmospheric CO_2 concentrations alter carbon uptake rates and forest flammability (Nepstad *et al.*, 2001; Watson *et al.*, 2000).

Beyond these technical challenges, our understanding of the Amazon's role in climate change is confounded by the dynamic nature of frontier land use. Amazonian forest is not being cleared in a steady, irreversible sweep. Rather, patterns of deforestation and forest regrowth vary dramatically between regions, and ebb and flow over time according to shifts in national policy (Browder, 1994; Coomes, 1996; Hecht, 1997). Amazonian land use is often characterized by high rates of land abandonment resulting in a mosaic of fields, pasture and secondary growth (Lambin *et al.*, 2001; Steininger, forthcoming). Fast-growing secondary forests may be abundant and have the potential to offset significant levels of carbon emissions from mature forest clearing (Houghton *et al.*, 2000). Understanding the dynamics of land use and associated carbon flux at tropical frontiers is key to more accurate modeling of global carbon flux (De Jong *et al.*, 2000; Schimel *et al.*, 2001; Steininger, forthcoming). Measuring carbon flux has additional importance given that proposed international agreements (e.g., Kyoto Protocol) require individual nations to inventory GHG sources and sinks (Brown *et al.*, 1995; Noble & Scholes, 2001).

Tropical forest conservation as a climate mitigation strategy is also a highly political issue. Many analysts fear such an approach will deflect attention away from the root cause of climate change, that is, GHG emissions from fossil fuel combustion in developed countries (Baer *et al.*, 2000). Several tropical countries are also wary of any attempt to "internationalize" their frontier territories. A third major concern is that maintaining frontier forests to protect global citizens from climate change will compromise the urgent economic needs of the rural poor living in these forests. Finding appropriate strategies for balancing local economic development with international carbon and biodiversity concerns depends on better understanding the driving forces of current deforestation, and identifying the potential beneficiaries of future forest management scenarios.

In this paper, I assess land cover change and the resulting carbon flux in Tambopata, Peru, a frontier area of high biodiversity importance (Figure 1). First, I analyze patterns of deforestation and regrowth along roads and rivers in Tambopata during an 11-year period (1986–97) when Peru's national agrarian policy shifted drastically, from macroeconomic populism to neoliberal austerity. In Tambopata, as in other tropical frontiers, changes in economic policy alter deforestation rates and relocate clearing activities, leaving some forested areas newly vulnerable and others to regenerate (Coomes, 1996; Gómez & Ortiz, 1998). I rely on remote sensing analysis to measure areas under mature forest, secondary regrowth, cultivation, and pasture during two intervals: (a) 1986–91, when agricultural credit and subsidies were readily available; and, (b) 1991–97, when structural adjustment and fiscal austerity measures were imposed. I connect these area estimates with field data on vegetation biomass density and accumulation rates to calculate the amount of carbon emitted and absorbed. In this way I link national policy shifts to local patterns of land cover change and associated carbon emissions. I also study deforestation along the Interoceanic Highway, and forecast the impact of road expansion and improvement in the region.

In the remainder of the paper, I consider possible strategies to mitigate future carbon emissions in Tambopata, especially projects proposed within the Clean Development Mechanism (CDM) of the Kyoto Protocol. These projects offer a means of placing economic value on the ecological services provided by forests, especially carbon storage and biodiversity conservation (Kremen *et al.*, 2000; Portela & Rademacher, 2001). But these projects are fraught with political conflict and technical uncertainties. Drawing from the carbon flux estimates for Tambopata during previous years, I project the carbon benefits and financial costs of promoting agroforestry or reduced-impact logging in the future. I also evaluate forest preservation, a strategy rejected for the Kyoto Protocol's first commitment period (2008–12), but still prominent in climate mitigation debates.

2. REGIONAL CONTEXT AND RESEARCH METHODS

Peru offers an important case for understanding the relationship between national policy, deforestation, and carbon flux at tropical frontiers. Peru ranks second to Brazil in the extent of intact lowland tropical forest (Itur-

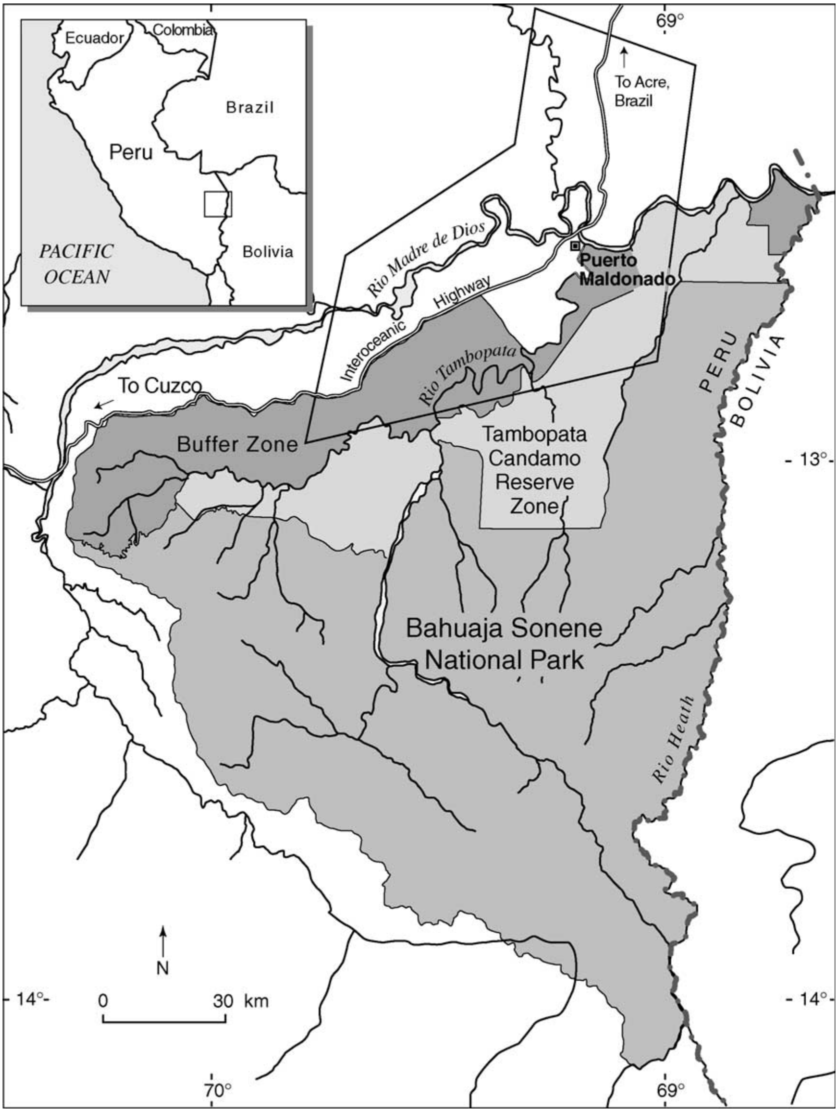

Figure 1. *Map of Tambopata study area.*

regui, Seminario, & Encinas, 2001). Peru also holds world records for biodiversity and species endemism (Terborgh, 1999). Like other Latin American countries, Peru has experienced political instability and extreme economic volatility over the past two decades. Poverty alleviation remains a national priority, but there is also increasing awareness of the country's vulnerability to climate change (Iturregui *et al.*, 2001). Under these conditions, conservationists must balance carbon and biodiversity concerns with local economic development.

(a) *Study site*

(i) *Biophysical characteristics*

Tambopata Province is located in the Department of Madre de Dios, Peru (Figure 1). This remote area is dominated by lowland humid forest (altitude: 200–350 masl; annual rainfall: 2,000–3,000 mm; Ascorra *et al.*, 1999). Tambopata contains extraordinary biodiversity due to its location in a transition zone between subtropical and tropical regions; e.g., biologists recorded 575 species of birds and >2,000 species of butterflies in a 5000 ha sample area (CDC, 1995; Foster, 1994). Tambopata also retains mature floodplain forests containing "giant" specimens of several tree species (e.g., *Ceiba pentandra*; Foster, 1994). Mature forests of this type are rare elsewhere in the Amazon due to the accessibility and fertility of riverside soils. Tambopata's forests are also highly dynamic, an attribute of special interest for carbon research. Tambopata ranked second in an international study of rainforest turnover rates, a measure correlated with rapid woody growth (Foster, 1994; Phillips & Gentry, 1994).

Peruvian and international conservationists have long lobbied for protection of Tambopata's forests, and in the year 2000, one million ha were set aside as Bahuaja Sonene National Park, with another 272,000 ha designated as a Forest Reserve (Figure 1). The Mayor of Puerto Maldonado then declared the city the "Biodiversity Capital of Peru." More commonly, the ecological services provided by Tambopata's old-growth forests (e.g., species preservation and carbon storage) are not factored into land use decisions. As elsewhere in the Amazon, forest is cleared for agriculture and cattle ranching to meet urgent local needs and national development goals (Alvarez & Naughton-Treves, 2003; Portela & Rademacher, 2001).

(ii) *Socioeconomic dimensions of land use in Tambopata*

Deforestation in Tambopata was negligible until the mid 1960s, when a road was constructed to connect Puerto Maldonado with the Andean highlands (Figure 1). This road was part of a national plan to populate remote areas and relieve social pressure from the land-poor highlands, or as President F. Belaunde then declared, "to conquer, occupy and exploit the Amazon" (Dourojeanni, 2001). In 1979, Peru signed an agreement with Brazil to construct a 5,739 km Interoceanic Highway passing from Acre through Madre de Dios (CTAR, 1998). Later, under the macroeconomic populist policies of president Alan Garcia (1985–90), the amount of agricultural credit flowing into the Peruvian Amazon tripled from mean levels for the previous 10 years (Coomes, 1996). Garcia's regime also promoted farmers' cooperatives and offered guaranteed markets for crops (Alvarez & Naughton-Treves, 2003; Coomes, 1996). Immigrants came by the thousands in response to these opportunities and to escape political violence in the highlands (Chicchón, 2000). Newcomers were granted 40 ha landholdings along roads and rivers, and they often used government credit to employ forest clearing crews to open up land for crops (Alvarez, 2001). After two or three planting seasons, many agriculturalists abandoned farming for extensive livestock production (0.5–0.8 head ha^{-1}), a more secure investment given Peru's hyperinflation during this period (Coomes, 1996). During 1985–90, rice production in Madre de Dios increased 30%, maize 72%, and cattle 29% (INEI in Ascorra *et al.*, 1999).

Agrarian and economic policies changed drastically when Alberto Fujimori was voted into office in 1990. In response to Peru's economic crisis and hyperinflation, Fujimori's administration imposed a radical austerity program based on structural adjustments. They dismantled the Agrarian bank, removed credit and subsidies, and imposed taxes. As a result, agricultural production and forest extraction activities declined (Varcarcel, 1993). Rice production in Madre de Dios declined 64%, and maize, 26%, during the first four years of Fujimori's regime (INEI in Ascorra *et al.*, 1999). But, Tambopata Province's population continued to grow by 5–6% per year during the 1990s, reaching 76,610 in 1997, with roughly half the population residing in the capitol city of Puerto Maldonado (GESUREMAD, 1998). The amount of titled land also grew steadily throughout the two study periods (Figure 2).

(b) *Research methods*

(i) *Remote sensing analysis of land cover change*

Satellite images from 1986, 1991, and 1997 were acquired for the 414,759 ha study region (Figure 1). These dates were selected for analysis based on the assumption that land use changes lag policy changes; e.g., to test the impact of the Garcia's regime (1985–90), images from 1986 and 1991 were employed. The 1986 image came from a 79 m resolution Landsat MSS sensor, and the 1991 and 1997 images from 30 m resolution Landsat TM sensors. The 1997 image was geometrically rectified and registered to a UTM coordinate system based on 1:100,000-scale topographic maps of the study region. The 1986 and 1991 images were georectified using an image-to-image approach with the 1997 image as the reference. The rectification process resulted in 0.71 and 0.63 pixel-RMS error for the 1991 and 1986 images, respectively. All the images were

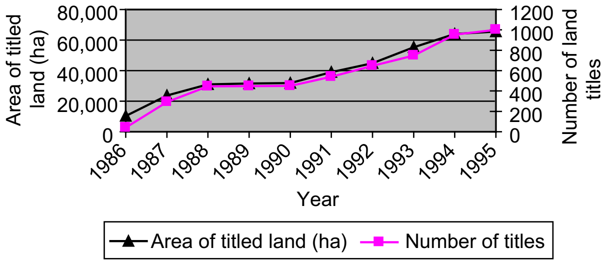

Figure 2. *Land titling in Tambopata.*

resampled to a 30 m pixel size. Data on land-use and land-cover were collected and geo-referenced during field work in 1999 and 2000 (see Alvarez, 2001; Alvarez & Naughton-Treves, 2003 for more details and an analysis of a portion of the study area considered here).

The land cover classes identified in a supervised classification of the images were: mature forest (including all closed canopy forest, seasonally flooded forest, and swamp forest); vegetative regrowth (all natural regrowth <15 years old on fallowed agricultural and pasture land); pasture or cultivated areas; and, water. Some secondary forest >15 years was likely included in the mature forest class, due to its spectral similarity. This is a common challenge in remote sensing research on deforestation (Steininger, 1996; Steininger, forthcoming). Old secondary forest is however relatively rare in the study area given Tambopata's recent history of agricultural expansion.

Land cover change was assessed by comparing the amount of land under each class in the 1986, 1991, and 1997 images. Forest clearing and regrowth were also assessed within two dynamic areas: within eight km of the Interoceanic Highway, and within 3 km of principal rivers (Tambopata and Madre de Dios) (Figure 3).

(ii) *Measuring biomass in mature and regenerating forest*

The amount of carbon held in mature forest in Tambopata was derived from field measurements of the stem diameter of 1802 trees >10 cm DBH (diameter at 1.3 m or above buttresses). Thus all biomass estimates reported here refer only to aboveground live biomass. The mass of dead trunks and belowground biomass may reach 14–34% of total biomass (Brown *et al.*, 1995), but is not included here. Thirty-seven transects (each 10×100 m) were placed randomly in mature forest at the edge of frontier settlements along rivers ($n = 29$) and roads ($n = 8$). These forests were largely unaffected by human activities during the past century, or were subject only to rubber tapping >40 years ago (Foster, 1994). To convert the stem diameter (D) to standing biomass, I used regression equations derived by Brown (1997) for moist tropical forest. For trees <160 cm dbh, biomass $= \exp\{-2.134 + 2.530 \times \ln(D)\}$; for trees >160 cm dbh, biomass $= 42.69 - 12.800(D) + 1.242(D^2)$.

I calculated the rate of biomass accumulation during vegetative regrowth in areas left fallow for 2–25 years after cultivation using 25 additional 10×100 m transects ($n = 453$ trees). Relic trees found in fallows were not included in basal area measurements. Following Brown (1997), I calculated biomass from stem diameter (D) and ran a regression of biomass (y) against vegetation age (x).

(iii) *Data analysis*

To calculate regional carbon flux during 1986–91 and 1991–97, I used a model in which field assessments of aboveground carbon stocks of mature forest and biomass accumulation rates of secondary growth are multiplied by the area under each land cover class measured in the satellite images (De Jong *et al.*, 2000; Fearnside, 1996; Steininger, 1996). Total carbon flux is calculated by summing carbon emissions from total areas of mature forest clearance (Eqn. (1)) and secondary forest clearance (Eqn. (2)), and subtracting carbon uptake from forest regrowth in fallow areas and newly abandoned fields and pastures (Eqn. (3)). Specific calculations are as follows:

Carbon emissions from mature forest clearance:

$$C_{\text{f-c}} = (A_{\text{f-c}} + A_{\text{f-r}})B_{\text{f}} \cdot F_{\text{c}} \cdot E_{\text{f}}, \tag{1}$$

where $C_{\text{f-c}}$ is carbon flux from mature forest clearance (measured in Mg of C, 1 Mg $= 10^6$ g),

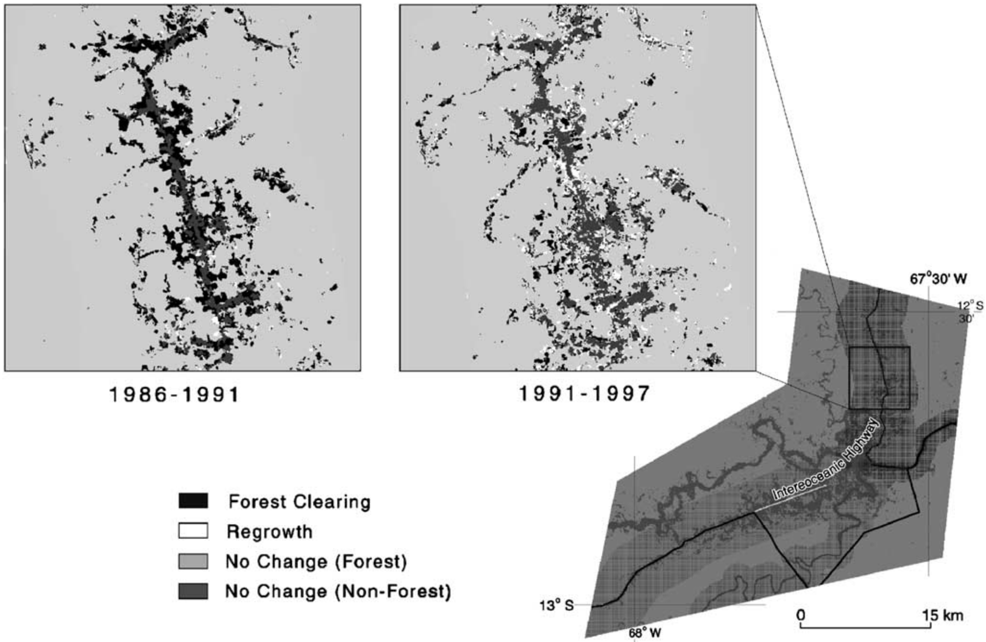

Figure 3. *Land cover change in Tambopata, 1986–91 and 1991–97.*

$A_{\text{f-c}}$ and $A_{\text{f-r}}$ are the areas of transition from forest to clearance and forest to regrowth (ha), B_{f} is aboveground live biomass of mature forest (407 Mg ha^{-1}, derived from transects in mature forest), F_{c} is carbon fraction of dry biomass and E_{f} is the burning efficiency of mature forest clearance (unitless fraction). I employed a value of 0.5 for F_{c} (Houghton *et al.*, 2000). For the burning efficiency of mature forest, I used 0.27 (Kauffman, Till, & Shea, 1992).

Carbon emissions from secondary growth clearance:

$$C_{\text{r-c}} = A_{\text{r-c}} \cdot B_{\text{r}} \cdot F_{\text{c}} \cdot E_{\text{r}}, \tag{2}$$

where $C_{\text{r-c}}$ is the carbon flux from secondary forest clearance (Mg C), $A_{\text{r-c}}$ is the area of regrowth to clearance transition (ha), B_{r} is the average biomass (Mg) for regrowth at the time of clearing. $B_{\text{r}} = G_{\text{r}} \cdot P$ (where G_{r} is average accumulation of aboveground biomass during fallow period P). Based on the field biomass surveys (see results below) I set 11.5 $\text{Mg C ha}^{-1}\,\text{yr}^{-1}$ for G_{r}. I used 3.2 years as the average fallow period based on field interviews (Alvarez & Naughton-Treves, 2003). E_{r} is the burning efficiency of secondary forest clearance (unitless fraction). I used 0.6 for E_{r} (Kauffman *et al.*, 1992).

Carbon uptake by regrowing secondary forest:

$$C_{\text{r}} = (A_{\text{r}} + 0.5(A_{\text{c-r}} + A_{\text{r-c}})) \cdot G_{\text{r}} \cdot F_{\text{c}} \cdot T, \tag{3}$$

where C_{r} is the carbon uptake by regrowth (Mg), G_{r} is again the rate of aboveground biomass accumulation for regrowth ($\text{Mg ha}^{-1}\,\text{yr}^{-1}$) and T is the time interval between satellite image dates (y). Because I did not know precisely when an area was cleared or abandoned during T, I used $0.5(A_{\text{c-r}} + A_{\text{r-c}})$ and assumed these change occurred at the midpoint of the interval.

(iv) *Accuracy assessment*

The accuracy of measurement for each independent variable was either calculated or drawn from the literature (Table 1). Then, following Morgan and Henrion (1990) and Steininger (forthcoming), carbon flux accuracy was estimated by error propagation (Table 1).

3. RESULTS

(a) *Deforestation patterns*

Approximately 291 km^2 of mature forest were cleared across the study area during 1986–

Table 1. *Accuracy assessment*

	Symbol	Range	Accuracy[a] (%)	Source of error
Average aboveground biomass ($Mg\,ha^{-1}$)	B_f	195–1,045	93.4	Variation in field estimates[b]
Average aboveground biomass increment of regrowth ($Mg\,ha\,yr^{-1}$)	G_r	12.3	89.0	r^2 for slope, age vs. biomass
Average fallow period (yr)	P	3.2	97	Variation in field interviews[b]
Fraction of carbon[c]	F	0.48	98	Reported values in other studies
Efficiency of biomass combustion[d]	E_f, E_r	0.27, 0.60	95	Reported values in other studies
Efficiency of decomposition[e]	E_{df}, E_{dr}	0.68, 0.35	95	Reported values in other studies
Mature forest area	A_f	*	98.0, 96.0, 90.0	Jackknife image classification
Cleared area	A_c	*	82.9, 90.0, 92.6	”
Secondary forest area	A_r	*	96.9, 86.0, 95.9	”
Mature forest clearance area	$A_{r\text{-}c}$, $A_{f\text{-}r}$	*	93.6, 88.7	Transition error matrix
Secondary forest clearance area	$A_{r\text{-}c}$	*	94.0, 86.8	”
Field abandonment area	$A_{c\text{-}r}$	*	70.5, 73.8	”
Secondary forest regrowth area	$A_{r\text{-}r}$	*	70.5, 82.5	”
C flux from deforestation	$C_{f\text{-}c}$	**	59.0, 63.8	Error propagation
C flux from regrowth clearance	$C_{r\text{-}c}$	**	65.0, 57.4	”
C flux from forest regrowth	C_r	**	50.6, 54.4	”

[a] When two values are listed, the first is for interval 1986–91, second is 1991–97. When three values are listed, first is for 1986, second for 1991, third for 1997.
[b] Accuracy for mean = (SE/mean) × 2.
[c] Data are from Whittaker and Marks (1975).
[d] First value is for mature forest, second for secondary forest (Kauffman *et al.*, 1992; Fearnside & Guimaraes, 1996).
[e] First value is for decomposition of felled mature forest, second is for felled secondary forest (Kauffman *et al.*, 1992).

97, while 242 km^2 regenerated as secondary forest. These measurements yield relatively slow annual deforestation rates of 0.7% (gross) and 0.1% (net) across the 11-year study period. But, deforestation was highly variable over space and time (Table 2). The greatest amount and fastest rates of mature forest clearance were observed within eight km of the Inter-oceanic Highway during 1986–91 when 166 km^2 of forest were cleared, at a rate of 2.4% per year (accuracy 93.6%, Table 1). Deforestation along the highway slowed to 1.4% during 1991–97 during which time 110 km^2 were cleared (accuracy 88.7%, Table 1). Deforestation activities extended further from the highway during this second time period (Figure 4). For example, 4.7% of mature forest was cleared on land 8 km from the highway during 1986–91, while 10.8% were cleared at this distance during 1991–97. Forest clearing along rivers was slower and more constant throughout 1986–97 (Table 2).

There was substantial secondary regrowth along roads and rivers throughout the study period (Table 2). Along the highway, the area under secondary regrowth expanded at 21.9% per year during 1986–91, and 24.3% per year during 1991–97 (calculated as % area present in previous year). Secondary growth along rivers expanded more slowly; 3.9% per year during 1986–91, 1.7% per year during 1991–97. By 1997, secondary growth covered 20% of land along roads and rivers, and mature forest covered 68% and 62% of land along roads and rivers, respectively.

(b) *Biomass density and accumulation rates*

Based on data from 37 transects, the average aboveground biomass density for mature forest at Tambopata is 407 $Mg\,ha^{-1}$ (range: 195–1,048, SD: 262). Four trees with dbh > 200 cm were disproportionately important in biomass calculations. They represented 14% of the biomass but only 0.2% of the sample of 1,802 trees. When I removed these “giants” from calculations, average biomass density estimate fell to 326 $Mg\,ha^{-1}$.

Table 2. *Land use and land cover change in Tambopata Province, Peru (1986–97)*

Land cover	Land <8 km from Transoceanic Highway area in km^2 (% of total area)			Land <3 km from Tambopata or Madre de Dios river area in km^2 (% of total area)		
	1986	1991	1997	1986	1991	1997
Mature forest	1,381 (77.2)	1,251 (69.9)	1,216 (68.0)	516 (65.0)	500 (63.1)	492 (62.0)
Secondary regrowth	144 (8.0)	339 (21.8)	365 (20.4)	94 (11.9)	166 (21.0)	162 (20.5)
Fields/pasture	231 (12.9)	113 (6.3)	177 (9.9)	106 (13.4)	50 (6.2)	69 (8.6)
Waterbodies	35 (1.9)	34 (1.9)	31 (1.7)	76 (9.6)	77 (9.5)	70 (8.8)

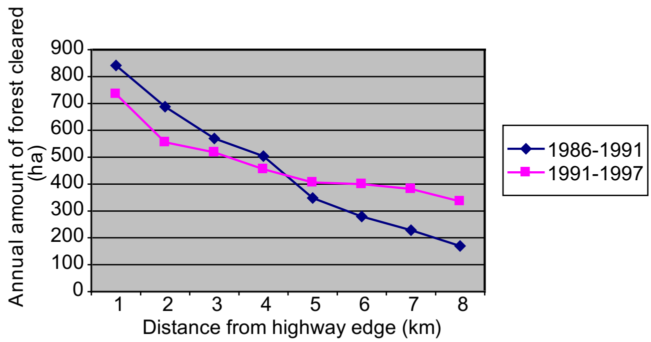

Figure 4. *Deforestation vs. distance to highway.*

Data from 25 transects in regenerating fallows (age 2–15 years), revealed an average biomass accumulation rate in regenerating forest of 11.47 $\mathrm{Mg\,C\,ha^{-1}\,yr^{-1}}$. Regressing biomass density on forest age yields the following equation: $Y = -0.08 + 11.47$ (years fallow) ($r^2 = 0.89$, $p = 0.0001$). There was no significant difference between accumulation rates for land used for pasture (short term, low stocking) versus shifting cultivation.

(c) *Net carbon fluxes*

During 1986–97, the clearing of mature forest released an estimated 7,739,000 Mg C. Secondary clearing released an additional 189,000 Mg C. The greatest volume and rate of emission occurred during 1986–91 along the Interoceanic Highway when 3,569,000 Mg C were released (59% accuracy, Table 1), of which 14% was offset by carbon uptake in secondary regrowth (Table 3). During 1991–97, 2,335,000 Mg C were emitted from forest clearing along the highway (63.8% accuracy), of which 45% was offset by regrowth. Land cover change along rivers produced 1,077,000 and 1,009,000 Mg of carbon during the two study periods, during which time uptake from regrowth offset 25% and 46% of emissions respectively (Table 3).

4. COMPARING DEFORESTATION AND CARBON FLUX AT TAMBOPATA TO OTHER AMAZONIAN SITES

(a) *Observed patterns of deforestation*

The observed annual rate of forest clearance (0.1% net) in Tambopata is below the estimated national level of 0.4% (1990–2000; FAO, 2000). Deforestation in Tambopata was also slower than neighboring frontier regions; e.g., ~0.3% for Acre, Brazil (1988–97; INPE in Laurence *et al.*, 2001), and ~0.2% for the adjacent lowland forests in Bolivia (1992–94; Steininger *et al.*, 2001). On average, 26.5 km^2 of mature forest was cleared in the study region each year, or ~1% of the total amount cleared each year across the Peruvian Amazon (Iturregui *et al.*, 2001). But, while deforestation rates averaged for the entire Tambopata study region were relatively slow, land use was concentrated

Table 3. *Estimates of carbon flux at deforestation frontiers in Tambopata, Peru (fluxes to atmosphere are positive, uptake is negative)*[a]

	Average annual rate (Mt)				Total affected land
	Land <8 km from Interoceanic Highway		Land <3 km from rivers		Total (Mt)
	1986–91	1991–97	1986–91	1991–97	1986–97
Carbon emitted from mature forest clearance					
Initial burn	0.201	0.111	0.061	0.046	2.199
Decomposition	0.507	0.279	0.153	0.116	5.539
Subtotal	0.708	0.389	0.213	0.162	7.739
Carbon emitted from regrowth clearance					
Initial burn	0.003	0.014	0.001	0.004	0.119
Decomposition	0.002	0.008	0.001	0.002	0.070
Subtotal	0.005	0.023	0.002	0.006	0.189
Carbon uptake from fallow regrowth					
Assimilation	−0.108	−0.184	−0.056	−0.077	−2.257
Net flux	0.606	0.228	0.160	0.091	5.671

[a] 1 Mt = 1,000,000 Mg = 10e12 g.

along roads and rivers where 1.1% and 0.4% yr^{-1} of mature forest were cleared (respectively) leading to losses of 12% and 4.7% of forest area over 11 years. This pattern matches observations elsewhere in the Amazon that land use intensity increases exponentially with proximity to access routes, and is typically greater along roads than rivers (Imbernon, 1999).

The proximate causes of deforestation in Tambopata parallel those for many other tropical frontiers, particularly those in Latin America. Agricultural expansion was associated with 96% of 152 deforestation cases sampled across the tropics (Geist & Lambin, 2002). Roughly half of these cases were attributed to shifting cultivation, and nearly all clearing occurred near new roads (Geist & Lambin, 2002). Underlying these proximate causes are national goals of economic growth and territorial control, which lead to policies favoring credit, subsidized markets, and low taxation at frontiers (Coomes, 1996; Geist & Lambin, 2002; Steininger *et al.*, 2001). In the case of Tambopata, roadside deforestation was most rapid in the second half of the 1980s when credit and guaranteed markets were available for colonists. Later, when the Fujimori administration removed guaranteed markets and credit, roadside deforestation slowed significantly, even as road improvements and population growth continued. Land was titled at a relatively constant rate throughout 1986–97 (Figure 2), suggesting that access to credit, not land, was the constraining factor for clearing. Many peasants responded to the austerity measures of the 1990s by moving to Puerto Maldonado, or turning to subsistence production along rivers. A similar slow-down and shift in clearing activities occurred elsewhere in the Peruvian Amazon during the 1990s (Coomes, 1996; Maki, Kalliola, & Vuorinen, 2001). But the same type of neoliberal policies that constrained deforestation in Tambopata facilitated massive clearing for soybeans in Santa Cruz, Bolivia (Hecht, 1997; Steininger *et al.*, 2001). This contrast underscores the need to consider regional context in predicting the impact of agrarian policy. In the future, as Tambopata's Highway is improved and its economy grows, a flow of private credit could potentially cause rapid forest clearing for soybeans or other industrial crops, as occurred in Bolivia (Steininger *et al.*, 2001). Indeed, soybeans are featured in future development plans for Tambopata (CTAR, 1998).

(b) *Biomass densities and uptake rates*

The biomass measured in this study (407 $Mg\,ha^{-1}$) roughly matches other local studies of mature floodplain forest (e.g., 395–414

Mg ha^{-1}; Chambi Condori, 2001; 392 Mg ha^{-1}, Phillips *et al.*, 1998), but exceeds values for most other Amazonian sites (commonly 300–372 Mg ha^{-1}, range 169–860; Nelson *et al.*, 2000; Steininger, forthcoming). This may be attributable to Tambopata's favorable growing conditions (Phillips *et al.*, 1998). The high standard deviation (262) reveals however the variability in biomass density within the region and the disproportionate influence of rare giant trees (biomass density falls to 326 Mg ha^{-1} when four trees >200 cm dbh were excluded). Malhi and Grace (2000) estimated 50% of aboveground biomass in a Brazilian forest was contained in the largest 10% of trees. Brown *et al.* (1995) found that 3% of the trees accounted for 50% of the total biomass, but warned that giant trees can inflate estimates when sample units are <1000 m^2 (our sample units were 1000 m^2). Moreover, published regression equations for estimating biomass are typically derived from measurements of smaller trees (Brown, 1997). On the other hand, traditional size-structured approximations may discount the C stored in emergent, large canopy trees and "severely underpredict" the aboveground biomass in mature forests (Moorcroft, Hurtt, & Pacala, 2001). Chambers, Higuchi, and Schimel (1998) argued that giant (>200 cm) Amazonian trees (including species measured in this study) absorb and store carbon over centuries. Such giants can be 600–1400 years old (Chambers *et al.*, 1998). In any case, biomass density of mature forests remains one of the greatest sources of uncertainty in carbon flux studies (Brown, 1997; De Jong *et al.*, 2000; Iturregui *et al.*, 2001).

The rapid rate of biomass accumulation measured in Tambopata's young fallows (11.5 Mg ha^{-1} yr^{-1}) is also at the high end of estimates for moist forest (typically 5–10 Mg ha^{-1} yr^{-1} after short-term shifting cultivation) (Phillips *et al.*, 1998; Nelson *et al.*, 2000). Again, the rate we measured accords with previous studies in Tambopata (e.g., 12.5 Mg ha^{-1} yr^{-1}; Chambi Condori, 2001).

(c) *Net carbon flux and the importance of secondary forest*

The high volume of C released during 1986–91 was produced by the rapid clearing of mature forest along the Interoceanic Highway under government subsidies. Only 14% of total emissions were offset by secondary regrowth during this time. The subsequent surge in secondary growth offsets (45% of total emissions during 1991–97) reflects the decline in mature forest clearing *and* regrowth on abandoned lands. Carbon offsets from secondary regrowth at Tambopata, while highly variable, are roughly comparable to other Amazonian frontiers populated by shifting agriculturalists. In Brazil, Houghton *et al.* (2000) assumed that 18–50% of deforestation flux could be offset by secondary growth. Steininger (forthcoming) found that secondary forest at a Bolivian site offset 41–64% of emissions, and 25–48% of emissions in two Brazilian sites. But, when Steininger extrapolated his findings for regrowth uptake and compared them to deforestation emissions for the whole Brazilian and Bolivian Amazon, this offset fell to a range of 4–27% offset by secondary growth. Similarly, Fearnside (1996) offers a much lower estimate (0.8%) for carbon offsets by secondary forest across the entire Brazilian Amazon. Estimates of carbon offsets are site dependent and scale-sensitive. Studies from remote, sparsely populated tropical frontiers (e.g., Tambopata) do not accurately represent basin-wide offsets. At soybean frontiers or in areas of large-scale ranching, secondary growth will offset negligible amounts of carbon emissions from forest clearing (Fearnside, 1996). Even under optimal regrowth scenarios, the high biomass stored in Tambopata's mature forest dwarfs the amount of biomass accumulated in secondary forests.

5. STRATEGIES FOR MITIGATING CARBON EMISSIONS AND SLOWING DEFORESTATION

The patterns of deforestation and regrowth observed in this study show the sensitivity of frontier land use to macroeconomic policies. Tambopata reveals that carbon flux cannot be predicted solely on projected population growth or the extent of road networks. Rather, the fate of carbon-heavy and biodiverse mature forests at tropical frontiers will also be shaped by agrarian policy, credit programs and local struggles to claim and control forest use.

Like many frontier regions, Tambopata is subject to contradictory national strategies to both exploit and conserve forests (Turner *et al.*, 2001). Peru has devoted 60% of Madre de Dios to protected areas and indigenous reserves (CONAM, 2003). The government is also building roads and subsidizing agriculture in the same region. Madre de Dios is now free from taxes on agriculture, forestry, and fossil fuel use (Iturregui *et al.*, 2001). The state agency building the Interoceanic Highway (CTAR) promises to open 800,000 ha for agriculture in Madre de Dios, and another 4,500,000 ha for mining, logging, and tourism in order to generate ~US$700 million GDP yr^{-1} (CTAR, 1998). They intend to repay the cost of road construction by charging tariffs on the four million Mg of soybeans and grain they hope will be eventually trucked out of the region each year (CTAR, 1998). CTAR acknowledges the ecological values of Peru's forests but states that the highway is the "best guarantee for a rational use of our Amazon and its resources" (CTAR, 1998, Section 7.2).

Ecologists predict severe environmental and social impacts from the road (Dourojeanni, 2001; Nepstad *et al.*, 2001). In the Brazilian Amazon, 29–58% of forests were cleared within 50 km of paved roads in <20 years (Nepstad *et al.*, 2001). This level of clearing along the Interoceanic in the Tambopata study area alone would result in 54,000–103,000 km^2 of additional mature forest lost and 328,000,000–623,000,000 Mg C emitted, worth US$6.5–12.5 billion. [2] The indirect environmental impacts of the road potentially extend further, and include biodiversity loss via illegal logging, hunting, and fishing (Nepstad *et al.*, 2001). Indigenous reserves and protected areas near the road are at risk of invasion by loggers and miners. [3] Land ownership tends to be severely concentrated in soybean production, and land scarcity may drive the rural poor to forest margins and protected areas (Fearnside, 2001b). Despite these threats, few environmentalists are attempting to stop the Interoceanic. Rather, they hope to mitigate the road's impact due to the "unanimous, irrevocable decision" by the Peruvian government and local citizens in favor of the highway (Dourojeanni, 2001). [4] Their efforts are constrained by poor funding and politically weak environmental institutions (Dourojeanni, 2001). While Brazil has received US$250 million from multilateral banks to mitigate the environmental impact of roads, Peru has received nothing to date (Dourojeanni, 2001). Many environmentalists look to the CDM Forest and Land Use projects proposed within the Kyoto Protocol as a potential source of funds for mitigating the environmental impact of road development (Fearnside, 2001a). Ideally, these projects would provide a mechanism for industrialized countries to pay for the global environmental services of Tambopata's forests (e.g., carbon storage and biodiversity maintenance) while safeguarding the livelihoods of the rural poor. In essence, forests (including agroforests) could better compete with industrial soybean plantations and cattle pastures if carbon storage and sequestration generated significant revenue.

The Peruvian government endorses the Clean Development Mechanism (CDM) of the Kyoto Protocol [5] and has laid out strategies for reducing GHG emissions (Iturregui *et al.*, 2001). The largest source of GHG in Peru is land use and land cover change [6] (23,593,792 Mg of CO_2 carbon equivalent in 1994, of which an estimated 43% was offset by regrowth on abandoned land) (Iturregui *et al.*, 2001). Under the proposed strategies, a total of 40.4 million Mg C would be mitigated between 1999 and 2020 (Table 4) (Iturregui *et al.*, 2001). Land use and forestry projects are featured prominently, particularly agroforestry, reforestation and reduced-impact logging projects (Table 4). [7]

(a) *Agroforestry*

Agroforestry can increase C sequestration on agricultural land by enhancing and expanding forest fallows, and adding woody perennial crops and timber species to farms (Bass *et al.*, 2000). Agroforestry projects also protect watersheds and soil, and promise to break the cycle of land clearing and abandonment at forest frontiers (Iturregui *et al.*, 2001; Smith & Mourato, 2002). For example, in Thailand, shifting cultivators who adopted agroforestry systems reportedly slowed forest clearing and thus offset 1 Pg C over 20 years (Dixon *et al.*, 1994). But the major advantage of agroforestry over other CDM forest projects is their potential to improve livelihoods for the rural poor (Smith & Mourato, 2002). In the prototype CDM agroforestry project, farmers participate on a voluntary basis and receive direct payments for improving carbon storage on their land (Bass *et al.*, 2000). This strategy avoids political conflict and allows local farmers to capture some of the positive global externalities of forest conservation, while also earning income from crop and timber sales. Given that 54% of Peruvians live below the official poverty line, it is no surprise that agroforestry projects have priority in government plans (Iturregui *et al.*, 2001).

Table 4. *Projected CO_2 reductions from proposed CDM type projects in Peru*

Sector	Type of project	Estimated cumulative reduction by 2020[a] (in million Mg C equivalent[b])
Land use and forestry		
	Shade coffee cultivation with valuable hardwoods[c]	12.5
	Reforestation with exotic species (*Pinus radiata*)	2.7
	Reforestation with native species (*Polylepis* sp.)	1.2
	Heart of palm cultivation with yuca	0.7
	Forest management	0.65
	Subtotal	17.8
Transportation		8.5
Energy		14.1
Total		40.4

Source: Iturregui *et al.* (2001).
[a] Some projects are scheduled to be complete by 2010.
[b] Iturregui *et al.*'s original estimates were in CO_2 gas equivalent. Here they are converted to carbon equivalent so they can be compared with results from Tambopata.
[c] Includes carbon from avoided deforestation.

Currently, few farmers in Tambopata manage multi-strata, species-rich agroforestry systems. Instead, their 40 ha farms are covered on average with 21 ha mature forest, seven ha fallow land, five ha annual crops, and five ha pasture (Alvarez, 2001). As in other frontier regions, labor constraints, land insecurity, and lack of technical knowledge hinder the adoption of agroforestry (Coomes, Grimard, & Burt, 2000; Perz & Walker, 2002; Smith & Mourato, 2002). Direct payments and technical assistance would likely be necessary for farmers to adopt agroforestry in Tambopata. Farmers elsewhere in the Peruvian Amazon reported they would charge \$138 $ha^{-1} yr^{-1}$ to convert their land to agroforestry, or \$218 $ha^{-1} yr^{-1}$ to preserve mature forest (Smith & Mourato, 2002). Assuming these estimated charges would be sufficient incentive for Tambopata residents, [8] and that biomass accumulation in agroforestry matched the high rate measured in regenerating fallows in this study, a 15-year agroforestry project could sequester 172 Mg C ha^{-1} at \$12 per Mg, and the C would have a net present value of \$5–22 ha^{-1} (Table 5). If farmers agreed instead to forgo clearing one ha of mature forest on their farms for 15 years, the carbon benefit in terms of avoided deforestation would be 407 Mg C ha^{-1} at \$8.00 per Mg, with a net present C value of \$200–800 (Table 5). In either case, if by participating in this project, farmers opted to stay on their land and avoid colonizing new forest, the C benefits would increase significantly and the price per Mg would decline.

Agroforestry projects are unanimously endorsed by Peruvian agencies and nongovermental organizations (NGOs). The government's high projected carbon offsets from agroforestry include avoided deforestation because they believe these projects will break the cycle of land clearing and abandonment (Iturregui *et al.*, 2001). They do not specify how this added benefit is calculated. Measuring and verifying the C offsets of agroforestry via avoided deforestation would be challenging, given the complex local and national forces driving deforestation. For example, in areas of Brazil where farmers incorporated small areas of high value perennial crops, forest-clearing rates initially declined, but when credit became available, farmers resumed rapid clearing (Smith *et al.*, 1999). In Tambopata, subsidies for rice, soybeans or cattle would undermine agroforestry projects.

(b) *Reduced impact logging (RIL)*

Reduced impact logging mitigates C emissions by avoiding unnecessary damage to the surrounding forest during timber harvest, enhancing subsequent forest regeneration, and guarding against fire (Pinard & Putz, 1996). RIL techniques are ignored in most logging

Table 5. *Net present value of carbon per ha under four land use scenarios (US$)*

Project length (yr)	Discount rate (%)	Agroforestry (on land slated for annual crops)	Reduced impact logging (on land slated for conventional logging)	Forest preservation (on land slated for conventional logging)	Forest preservation (on land slated for annual crops)
15	5	45.68	198.61	476.67	806.36
	10	23.10	100.41	240.99	407.68
	20	11.55	50.23	120.54	203.92
30	5	42.65	204.17	490.00	828.92
	10	22.19	102.07	244.96	414.39
	20	11.30	50.68	121.63	205.76

operations (Johnson & Cabarle, 1993), but there are promising results from a few CDM pilot projects. A US utility sponsored a project in Malaysia that reduced logging damage by 50%, thereby saving 40–60 Mg C ha^{-1} (Pinard & Putz, 1996). A key advantage of RIL is that substantial C offsets are earned upfront during the early years of the project when logs are cut. The costs of RIL schemes tend to be lower than agroforestry projects, but the benefits flow primarily to skilled laborers, subcontractors, concessionaires and the national government, not peasant farmers (Bass *et al.*, 2000; Kremen *et al.*, 2000; Klooster & Masera, 2000).

Tambopata holds substantial amounts of mahogany and other valuable hardwoods. Logging has become one of the top three sources of income for Madre de Dios, and there is intense local and national pressure to expand logging in the region (Ascorra *et al.*, 1999; Chicchón, 2000). Some farmers and indigenous groups have formed collectives hoping to independently manage logging concessions and retain profits in the local economy. As in the case of agroforestry, the viability of RIL projects is shaped by national policy. In 2001, Peru revised its Forestry and Wildlife Law (DS 014-2001-AG) to improve the sustainability of logging and incorporate the value of the environmental services of forests, including C sequestration. The new regulations favor RIL but they are not yet generally enforced. Illicit "cut and run" logging occurs all too often on public land, protected areas and indigenous territories containing valuable hardwoods (Kvist & Nebel, 2001).

Assuming that logging regulations will eventually be enforced in Madre de Dios, RIL projects have the potential to mitigate substantial amounts of C at low cost. The government currently charges concessionaires a flat rate of US$5.00 ha^{-1} yr^{-1}, and management costs range from US$20–100 ha^{-1} yr^{-1} (Raez-Luna, E. personal communication, May 2003). If a concessionaire agreed to a 15-year RIL project, and achieved published projected emission reductions of 50 Mg C ha^{-1}, the C per ha would have a net present value of $50–198 and would cost of $2.10 per Mg, assuming $100/ha management cost during harvest (Table 5).

(c) *Forest and biodiversity preservation*

Like other development frontiers, Tambopata's mature forests contain great stocks of carbon and globally important biodiversity (Foster, 1994). Ecological economists estimate intact Amazonian forest provides $432–$1431 ha^{-1} yr^{-1} worth of indirect environmental benefits, including genetic resources, climate mitigation, erosion control, and nutrient cycling (Portela & Rademacher, 2001; Torras, 2000). By these calculations, the forest in Tambopata's Bahuaja–Sonene Park is worth $648 million to US$2 billion per year in environmental services. In terms of C alone, projecting biomass density estimates from our study site across the park yields 203,500,000 Mg C worth >US$4 billion to the global community (assuming a damage cost of $20 Mg^{-1} as per Kremen *et al.*, 2000). Yet Bahuaja–Sonene has an annual management budget of only US$127,000 [9] (2001, C. Landeo, personal communication). In essence, carbon is being maintained at <$0.01 Mg^{-1}.

Protecting forests that are destined for agriculture or logging offers an inexpensive means to offset C (Table 5). But, the greatest ancillary benefit of avoided deforestation projects is the conservation of biodiversity (Bass *et al.*, 2000; Fearnside, 2001a; Kremen *et al.*, 2000;

Totten, 1999). A large block of mature forest not only stores significant C, it also protects forest ecosystems from the deleterious effects of fire, fragmentation and overhunting. In the Bolivian Amazon, a consortium of power companies paid a logging company \$9.5 million to retire logging rights to 640,000 ha adjacent to Noel Kempff National Park (Totten, 1999). As a result, the park doubled in size and 15 million Mg of C emissions were avoided (Totten, 1999). Similarly, in 2001, a Peruvian NGO (Asociación para la Conservación de la Cuenca Amazónica, or ACCA), used a US\$5 million donation to purchase a 135,000 ha, 40-year conservation concession near Bahuaja-Sonene National Park. Although their central aim is to protect biodiversity and ecosystem function, the forest within the concession also holds C worth >\$800 million to the global community at a cost of ~US\$0.10 Mg^{-1} C [10] (Table 5). Assuming a 15-year time horizon and that the land under concession would have been cleared for agriculture, the C per ha has a net present value of US\$200–800 (Table 5). If, on the other hand, the land would only have been selectively logged, the C saved per ha has a net present value of US\$120–476 (Table 5).

Avoided deforestation projects remain highly controversial. Although some tropical countries support avoided deforestation projects as a carbon mitigation strategy (e.g., Bolivia, Argentina, and Costa Rica), both Brazil and Peru refuse them, arguing that these projects have high opportunity costs, threaten national sovereignty, provide perverse incentives for deforestation, and produce insurmountable accounting problems (Fearnside, 2001b). Identifying baseline land use scenarios and then accurately measuring and verifying carbon offsets is a challenging exercise for *any* land use and forestry project (Noble & Scholes, 2001). It is especially difficult at tropical frontiers where land use fluctuates according to boom and bust economic cycles, migration and/or sociopolitical conflict. Any of the activities outlined in Table 5 could be undermined by national policy shifts, wildfire or other disturbances. A potential investor in carbon sequestration or storage at Tambopata, or any other tropical frontier, would have to accept significant risk given the dynamic nature of local land use, as well as national politics and economic conditions. In light of this risk, Fearnside, Lashof, and Moura-Costa (2000) propose a "ton-year" approach in which carbon storage or sequestration is not bought in perpetuity, but is rented on a yearly basis. Even if a project simply delays clearing of old-growth forest for 15 years, some atmospheric benefits will remain, i.e., these projects can buy time [11] (Noble & Scholes, 2001). The major drawback of avoided deforestation projects is the opportunity costs incurred by local populations and national governments (Bass *et al.*, 2000; Kremen *et al.*, 2000). Strict exclusion of all resource use in a forest is often impractical and/or unethical. The avoided deforestation projects implemented in Belize and Bolivia both invested funds in providing alternative income, training local labor forces, and in the case of Bolivia, securing land tenure for neighboring indigenous communities. [12] Such an integrated approach would be necessary at Tambopata to assure that projects are equitable and politically acceptable.

Ultimately, avoided deforestation projects were excluded from the Kyoto Protocol under the accords reached in Marrakech and New Delhi (UNFCCC, 2003). As the Protocol is currently interpreted for the 2008–12 period, CDM projects can account for only 1% of GHG reductions for any Annex B country (Article 12). This reflects the preeminent importance of reducing fossil fuel use emissions in industrialized countries. Carbon held in forests is inevitably at greater risk of emission than carbon stored in underground fossil fuel reserves. But, the debate over avoided deforestation projects is not over, because the inclusion of these projects will likely remain a key condition for the US to re-enter international climate treaty negotiations (M. Oppenheimer, personal communication). Better land and forestry management remains an important part of climate change mitigation. The ideal project would preserve species-rich forests and reduce carbon emissions, while simultaneously providing secure incomes for local citizens, private companies, and government agencies. No single strategy can achieve these multiple and internally contradictory goals. A complementary approach combining different strategies (e.g., agroforestry and forest conservation) would likely prove more resilient in light of the dynamic nature of land use and politics at Tambopata. Field-based projects must also be aware of the impact of national policy and infrastructure. Neoliberal politicians may be more supportive of forest conservation and sustainable use if carbon offset funding allowed Peru to earn foreign exchange and improve its balance of trade ratio (Klooster & Masera, 2000). Meanwhile, Peru's current president, Alejandro Toledo, promises to re-open Peru's Agrarian Bank. If credit opportunities are carefully crafted to support agroforestry (e.g., Perz & Walker, 2002), eco-tourism and/or reduced impact logging, Tambopata's forests will benefit. If soybeans and cattle are subsidized, the biodiversity and carbon losses will be great.

NOTES

1. Pg = 1 Petagram = 10^{15} g; 1 Mt = 1 Teragram = 10^6 g; 1 Mg = 1 Megagram = 10^6 g.

2. Assuming a global damage cost of US\$20 per Mg C (Kremen *et al.*, 2000).

3. E.g., the proposed "Marginal de Selva" road would pass through Manu National Park (Dourojeanni, 2001).

4. In fact, politicians are have lobbied fiercely to have the highway pass through their home region (Dourojeanni, 2001).

5. Peru is highly vulnerable to climate change. For example, the rapid melting of Peru's montane glaciers over the past two decades threatens the supply of water and hydroelectric power to large urban areas (Iturregui *et al.*, 2001).

6. Peru's GHG emissions are dwarfed by those of industrial countries, e.g., in 1994, Peru emitted 26,950,308 Mg C vs. 1,520,763,377 by the United States, or ~4 Mg CO_2/capita vs. 22 Mg CO_2/capita. Peru's energy sector will soon surpass land use and cover change as a source of emissions (Iturregui *et al.*, 2001).

7. The proposed reforestation projects target highland areas so they are not addressed here.

8. Charges would likely be less in Tambopata given that land is far more abundant than in Pucallpa. E.g., land values in Tambopata range from \$35–250 ha. Caution is warranted in extrapolating results from one region to the next, even within the same country (Perz & Walker, 2002).

9. The budget of NGOs working around the protected area is higher, on the order of \$500,000 per year (A. Chicchon, personal communication).

10. Conservation concessions are typically modeled after timber concession, except that instead of paying for the right to log a forest, conservationists compensate national authorities or local resource users to protect natural ecosystems (Gullison, Rice, & Blundell, 2000). In this case, \$5 million guaranteed the first five years of the concession with an option to renew.

11. Only assuming that deforestation is not simply displaced elsewhere during the delayed period. Note that benefits from sequestration projects (reforestation or agroforestry) are entirely reversible if the sequestered vegetation is later cleared and burned (Noble & Scholes, 2001).

12. Analysts sharply disagree whether the funds committed to local economic development in these projects were adequate and/or administered appropriately (Asquith, Vargas Rios, & Smith, 2002). The debate parallels arguments regarding the viability and ethics or Integrated Conservation and Development Projects around protected areas.

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