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Will buying tropical forest carbon benefit the poor? Evidence from Costa Rica

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Abstract

We review claims linking both payments for carbon and poverty to deforestation. We examine these effects empirically for Costa Rica during the late 20th century using an econometric approach that addresses the irreversibilities in deforestation. We find significant effects of the relative returns to forest on deforestation rates. Thus, carbon payments would induce conservation and also carbon sequestration, and if land users were poor could conserve forest while addressing rural poverty. We note that the poor appear to be marginalized in the sense of living where land profitability is lower. Those areas also have more forest. We find that poorer areas may have a higher supply response to payments, but even without this effect poor areas might be included and benefit more due to higher (per capita) forest area. They might be included less due to transactions costs, though. Unless the Clean Development Mechanism of the Kyoto Protocol is modified in its implementation to allow credits from avoided deforestation, such benefits are likely to be limited.

Keywords: Land use; Deforestation; Poverty; Carbon sequestration; Development; Costa Rica

Introduction

Net carbon emissions are lowered by reducing deforestation and soil degradation and increasing afforestation, reforestation, agro-forestry and forest rehabilitation (Niles et al., 2001; Tipper, 1996; Trexler and Haugen, 1995). Reducing deforestation in developing countries may have the most potential, while agricultural land management also offers significant gains, especially in Asia. Such land uses could reduce atmospheric carbon by 2.2 billion tons by 2012 (Niles et al., 2001).

These land uses involve a wide range of practices on the ground, including systems for small-holders. Some are already policy foci. Agro-forestry and community forestry management have been promoted for reducing rural poverty and achieving sustainable economic development.

Payments for carbon sequestration appear attractive for local incomes and for ecosystem services. Yet tradeoffs may exist. The policies that most alleviate poverty may not most cost-effectively sequester carbon. In reviewing policies and considering implications of our analyses, below we ask whether paying poor land users for carbon seems especially efficient or inefficient.

There is little empirical research on the supply response of poor land users, and we know of no econometric studies which explicitly consider the impact of poverty upon supply response. Economic analyses of the supply response to carbon payments exist, employing approaches from point estimates of average costs to engineering least-cost models to revealed preference within past land use (Parks and Hardie, 1995; Callaway and McCarl, 1996; Stavins, 2000; Plantinga et al., 1999; Kerr et al., 2006) but they are focused solely upon the US and/or not upon the poor.

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Revealed-preference methods have resulted in higher estimates of the marginal costs of carbon sequestration than the other methods (Stavins, 2000; Plantinga et al., 1999). The least-cost engineering models may not capture all of the costs landowners face, including option values or non-market benefits not captured in benefit-cost analyses or various barriers to switching. Econometrics (behavior-based) estimates of marginal cost curves for carbon sequestration have also indicated considerable heterogeneity in land quality and in carbon productivity and thus also considerable variation in marginal costs of carbon sequestration (Plantinga, 1999; Stavins, 2000).

We adopt a revealed preference approach to estimating the potential supply response to sequestration payments and the effects of poverty. We use Costa Rican forest data for five points in time, mapped into the 436 districts, with a FAO district poverty index and data on land-use returns. We estimate the responsiveness of deforestation to returns and integrate the results with ecological analysis of carbon to simulate a supply curve for avoided deforestation (on integration see also Kerr et al., 2003). We find that land users respond to payments. Carbon payments would induce conservation and carbon sequestration and, for poorer land users, could address poverty.

We note that the poor appear to be marginalized, in the sense of living on worse land for production. Those poorer areas, however, also have more forest on average than do richer areas. We find that in the poorer areas deforestation might be more responsive to (carbon) payments, but even if this effect is not sufficiently robust to matter the poorer areas might be included in and benefit more from carbon policy due to their greater (per capita) forest area. They might be included less in policy implementation due to practical issues such as transactions costs, though. Further, unless the Clean Development Mechanism of the Kyoto Protocol is modified within its implementation to allow credits from avoided deforestation, any benefits are likely to be limited.

Below, Section Payments for sequestration, reviews potential sources for payments. Section Land use, payments and poverty, provides a land-use model and reviews literature on constraints poorer land users face in responding to payments. They suggest our integrated (economics and ecology) empirical approach to estimating impacts of payments and poverty upon carbon. The next Section describes our data, including ecological work on carbon densities in Costa Rica, while the following Section presents our results and then concludes at the end.

Payments for sequestration

There are several potential sources of payment for carbon sequestration through land use. They differ in terms of focus on carbon and/or poverty. Their specific criteria will determine what activities are considered for funding and with which other activities they compete for funds.

The Clean Development Mechanism (CDM), under Article 12 of the Kyoto Protocol, allows investors from industrialized countries with binding emissions-reduction commitments and with greenhouse-gas emission levels that are above their commitment levels to obtain carbon credits from developing countries who have cut their emissions or increased carbon sinks.

Over 30 AIJ (i.e., joint implementation) land-use projects may qualify (Nasi et al., 2002), including some that have targeted small and low-income producers. Costa Rica's environmental payments program could affect up to 700,000 hectares at full operation (Chomitz et al., 1999) including some within smaller holdings. The Scolel Té Pilot Project in Chiapas, Mexico (De Jong et al., 2000) brokers credits from small-farmer forestry through a trust fund which also provides technical and financial assistance. Others that may qualify include Profafor in Ecuador (Cacho et al., 2002) and the Rewarding Upland Poor for the Environmental Services (RUPES) they provide program funded by International Fund for Agricultural Development (IFAD).

In November 2001, the Marrakesh Accords confirmed reforestation and afforestation as generating such credits but excluded conservation of standing forests (i.e., avoided deforestation) and farming-based soil carbon sequestration for the first commitment period ending in 2012. Yet there is pressure to change CDM rules to include avoided deforestation starting in 2012. Avoided deforestation has been controversial due to the risk of leakage but this risk is being reevaluated in light of the potential for mitigation at relatively low cost and, for some, gains such as biodiversity from maintaining forest (Wunder, 2005; Aukland et al., 2003) A proposal to include credits for avoided deforestation at the last meeting of the Conference of Parties to the UN Framework Convention on Climate Change in December 2005 held in Montreal received wide support. This issue seems likely to remain on the international climate change agenda (Spotts, 2005).

For poorer (likely small-holder) participants, key questions include whether agro-forestry is accepted and whether small suppliers can be competitive (significant uncertainties exist on the demand side, e.g. due to the US withdrawal from Kyoto, and on the supply side, e.g. regarding when and how Russia will enter (Black-Arbelaez, 2002)). Supply of carbon credits under the CDM may be large relative to demand. Even if that is the case in general, though, niche demands for credits that satisfy particular rules (e.g., specific definitions of "sustainability") may exist.

The Biocarbon Fund recently established by the World Bank is another source of funds. It will consider not only land use that qualifies under the CDM but also a broader menu of land uses including both avoided deforestation and soil-carbon sequestration. It explicitly requires that projects contribute to improved local livelihoods and yield cost-effective environmental impacts.

Outside of Kyoto, the US could generate significant demand through bilateral programs given investor pressures and pending state-level legislation that requires emissions reductions. The Chicago Climate Exchange facilitates carbon-credit transfers including land sequestration.

The Global Environmental Facility (GEF) is a source of grant financing that has 'sustainable development' as an objective. Its climate-change area is limited to energy and technological efficiency but its integrated ecosystem management and sustainable land management consider sequestration from land-use change. They fund activities that generate multiple environmental benefits including biodiversity conservation, water conservation, pollution prevention and net emissions reduction (GEF website). The GEF council has estimated that \$200 million annually will be needed by 2010 to support the integrated ecosystem management operational category.

Land use, payments and poverty

Within our formal model of land-use decisions below, in which owners maximize private net benefits from the land, without receiving payment for carbon land users have no incentive to provide carbon sequestration. Doing so must generate not only public but also private benefits. A carbon payment raises the private returns to forested land uses (ecotourism, non-timber forest product extraction, etc.) relative to those from cleared land uses (e.g. agriculture, cattle pasture). Thus, along with many other factors (Pfaff, 1999), such payments can induce changes in land use. Then the level of poverty can affect the way a given owner or household responds to payments.

Land use and payments

We use a dynamic theoretical model, like others (Stavins and Jaffe, 1990) but empirically emphasize key irreversibilities and the dynamics of development. We feel both are important for understanding deforestation within a developing country, including the effects of payments.

Each forested hectare j has a risk-neutral manager who selects T, the time when that land will be cleared of forest, in order to maximize the expected present discounted value of returns:

$$\mathrm{Max}_T \int_0^T S_{jt}\mathrm{e}^{-rt}\,\mathrm{d}t + \int_T^\infty R_{jt}\mathrm{e}^{-rt}\,\mathrm{d}t - C_\mathrm{T}\mathrm{e}^{-rt}, \qquad (1)$$

where S_{jt} is the expected return to forest uses of the land, R_{jt} the expected return to non-forest land uses, C_T the cost of clearing net of obtainable timber value and including lost option value and r the interest rate. Two conditions are necessary for clearing to occur at time T. First, clearing must be profitable.

Second, even if that is so, it may be more profitable to wait and clear at t+1, so (2) must hold:

$$R_{jt} - S_{jt} - r_t\,C_t + \frac{\mathrm{d}C_\mathrm{T}}{\mathrm{d}t} > 0 \qquad (2)$$

and if a second-order condition holds this necessary condition is also sufficient for clearing. Population and economic growth along a development path may lead the second-order condition to hold for land-use change in a developing country. Yet it may be violated at higher levels of development, e.g. if environmental protection becomes more stringent, returns to ecotourism rise, and capital-intensive agriculture requires less land. If it is violated, our reduced form empirical specification should be interpreted in terms of expression (2) plus the profitability condition.

Consistent with this model, we assume that deforestation has irreversibilities, since trees take time to grow and incurring the costs of development changes marginal returns to land uses. We separate deforestation from (still rare) reforestation and empirically examine deforestation, i.e. examine where forest present at the beginning of a period is cleared by the end of the period.

In the model, deforestation occurs when Eq. (2) is satisfied for the first time. When it occurs differs due to variation across space in exogenous land quality and access to markets and due to both exogenous and endogenous temporal shifts. The model's individual decisions are discrete, while we observe continuous rates of loss in districts, so we aggregate the model's predictions.

Specifically, in our aggregated data we do not perfectly observe the variables in Eq. (2) since deforestation and the factors that explain it (X_{it}, i = district, t = time) are measured for districts. The X_{it} vector generates a single estimated net clearing benefit for an entire district, though the actual returns and changes in costs of course vary across the parcels within each district. Thus, we imperfectly measure parcels' net benefits from clearing in Eq. (2), such that clearing occurs if

$$R_{ijt} - S_{ijt} - r_t\,C_t + \mathrm{d}C_\mathrm{T}/\mathrm{d}t = X_{it}\beta - \varepsilon_{ijt} > 0, \qquad (3)$$

where again i refers to an area, j to a specific parcel, ij to a specific parcel j known to be in area i, and ε_{ijt} is a parcel-year-specific term for the unobserved relative returns to forested land uses, so

$$\text{Probability (satisfying Eq. (3) so that cleared if currently in forest)} = \mathrm{Prob}(\varepsilon_{ijt} < X_{it}\beta). \qquad (4)$$

The predicted district-level clearing rates depend upon X_{it} and the distribution of the ε_{ijt}. If the cumulative distribution of the ε_{ijt} is logistic, then we have a logit model for each parcel

$$F(X_{ijt}\beta) = 1/(1+\exp(X_{ijt}\beta)). \qquad (5)$$

For our grouped data, we estimate this model using the minimum logit χ^2 method also known as "grouped logit" (see Maddala, 1983 and also see Greene, 1990 for an explicit discussion of the heteroskedasticity here). If h_{it} is an area's measured rate of forest loss, then we estimate

$$\log(h_{it}/(1-h_{it})) = X_{it}\beta + \mu_{it}. \quad (6)$$

The variance of the μ_{it} (referring to areas, not parcels) can be estimated by $(1/I_{it}\ h_{it}\ (1-h_{it}))$, where I_{it} represents the number of forested parcels in area i at the beginning of interval t and the estimator is consistent and asymptotically normal. This is estimated by weighted least squares.

Poverty and responses to payments

Existing studies of the relationship between poverty and land use have found multiple linkages although not a single, unambiguous conclusion regarding the direction of causal effects. Wunder, 2001 summarizes macroeconomic literature on poverty and deforestation and concludes that two effects are in opposition: capital endowments rise with income, enabling deforestation; but as returns to other economic activities rise with development, deforestation is less attractive. Kaimowitz and Angelsen, 1998 also summarize microeconomic work on causes of deforestation. They find that income levels, or poverty, have indeterminate theoretical and empirical effects. Pfaff et al. (2006) offers additional extended discussion of poverty's direct effect on land use.

A related but distinct issue is poverty's effect on responses to carbon payments. Many poor people can be found on the forest frontier ((World Resources Institute, 2005) estimates that about 1.6 billion people are dependent on forests in some way, including perhaps 500 million smallholders using forests for some fraction of their income and 60 million indigenous people who are extremely dependent upon the forest). This proximity has been noted per the potential for carbon services payments to the poor, though various constraints may restrict such flows.

Before considering such constraints, we note that the poor being located near deforested locations must be distinguished from the poor causing that forest clearing. Other factors can be sufficient for clearing independent of poverty. While the poor may be found on marginal frontiers it may be that the lack of institutions in such locations, for instance, truly cause the deforestation. Alternatively, smaller and poorer producers sometimes inhabit cleared and then abandoned land.

Returning to constraints on carbon supply response, poverty is associated with all of the following barriers to action: risk; high cost of capital and lack of investment capacity; poor rights to property; transactions costs; and efficiency in the production of carbon sequestration (Lipper and Cavatassi, 2004). These factors may cause the poor's carbon supply to be lower than others'.

Risk

Giving up the right to 'liquidate a forest asset' for income during difficult times could be an important cost of receiving payments for forest carbon (on managing risk see, e.g., Rodriguez-Meza et al., 2003). This has implications for the design of carbon-services payments. If they can increase security, like insurance, then they will be more widely adopted. If instead they represent a source of uncertainty, they may be ignored by risk-averse land users (e.g. Lemos et al., 2002).

Another risk and payments issue is that the reversibility of sequestration activities (e.g., if forests succumb to fire) may cause credits to be discounted. The poor may be discounted more as suppliers if they are perceived as offering higher-risk sequestration. Still, though, accepting any up-front discounts could be easier on the poor than being held liable in the future for carbon loss.

Limits on capital

Poor farmers lacking assets to invest up front may find it harder to pursue agro-forestry or reforestation. Further, the poor may not be able to borrow even when investments are justified (Fafchamps, 1999; Lipper, 2001). Perhaps payments can be structured to overcome this constraint.

Property rights

Poorer land users frequently do not hold secure individual title to their land. In addition, more than one type of property right may exist for a given parcel (e.g., rights to trees, water, post-harvest residue, etc.). Uncertain or complex property rights reduce the incentives of land users to invest in new land uses that sequester carbon, as the private rewards from this will be uncertain.

As an example of the effect this lack of clarity can have on a project, the Scolel Té Pilot Project provides evidence of communities with intractable internal conflicts being uncompetitive while communities featuring successful resource management were competitive. Observed costs ranged from a low of \$52/ha up to \$325/ha for those in more conflict (De Jong et al., 2000).

Transactions costs

High transaction costs for the poor can arise from the small scale at which they operate and from remoteness. For example, albeit with a small sample, Cacho et al. (2002) find that project costs per hectare and sequestration costs per ton were negatively correlated with project size.

Coordinating supply can reduce such cost. Its potential is illustrated by FACE Foundation projects, the largest being Profafor in Ecuador with 22,500 ha reforested (Cacho et al., 2002). Other cases are described in Smith and Scherr (2002) and Orlando et al. (2002). In many cases cost is reduced through the activities of an intermediary, most frequently an NGO.

Efficiency in sequestration

Low-income land users are expected to have a lower average rate of return to their land. Thus the payment necessary for them to forego these returns to sequester is likely to be lower. However, for supplying sequestration competitively, biophysical conditions matter as well. Whether the poor's land supports less sequestration is not well established (Lipper, 2001). If a rich owner has many parcels, their quality may vary considerably. Thus, at the margin, they could have a lower opportunity cost of sequestration, albeit a steeply sloped supply curve for carbon.

Existing studies (e.g. in IPCC Climate Change, 2001) suggest that some types of land-use change are more competitive in generating sequestration. Avoided deforestation and forest regeneration are relatively efficient, though land-use costs vary considerably across countries. Cacho et al. (2002), for four agro-forestry systems on degraded lands in Sumatra, found systems associated with smallholders to be competitive with plantations. Smith and Scherr (2002) note that costs of carbon from smallholder systems have been quite variable, with the opportunity cost of the land and scarcity of tree products as a major determinant. When smallholder costs are higher, non-carbon benefits may offset this. Tomich et al. (2001) studied costs of sequestration in land use in Sumatra and found smallholder sequestration competitive with more capital intensive land use.

Data

Deforestation variable

We observe forest cover at five points (1963, 1979, 1986, 1997 and 2000). The country has 436 political districts. Our smallest unit of observation is a form of sub-district, distinguishing different 'lifezones'. The Holdridge Life Zone System (Holdridge, 1967) assigns each location in Costa Rica to one of 12 lifezone categories. These reflect precipitation and temperature. On average there are about three lifezones present in a district so we can use up to 1229 observations per year. Yet as the poverty index described below is generated for districts, we focus on district results. Our dependent variable is the annual percentage loss of forest during a time interval.

Data for the dependent variable come from several sources (Kerr et al., 2006 for details). The 1963 data are from aerial photos digitized by University of Alberta to distinguish forest and non-forest. The 1979 data are produced from Landsat satellite images by the National Meteorological Institute of Costa Rica (IMN, 1994). The 1986 and 1997 data were also derived from Landsat satellite images (FONAFIFO, 1998) and distinguish forest, non-forest, and mangroves, while also indicating secondary forest (i.e., forest in 1997 but not 1986). The 2000 Landsat images were processed by the University of Alberta EOSL to be consistent with the 1986 and 1997 datasets.

For each district for each time interval, we calculate the area deforested. The 1986, 1997 and 2000 maps all have clouds so we calculate these areas deforested (and thus also rates of loss) from the visible portions of each observation, using pairs of images with consistent clouds. For intervals before 1986–1997 we cannot distinguish the gross from net transitions, and assume gross deforestation equals net (since reforestation was not widespread before 1986, this should not cause significant problems). If the measured gross deforestation is negative, we assign a zero.

Our dependent variable is the area deforested divided by the area of forest "at risk" at the start of the interval. We assume national parks and biological reserves are not at risk (they were not cleared (Sanchez-Azofeifa et al., 2003)). We also drop areas for which we do not have the poverty data. Because our time intervals are of varying lengths, we use annualized rates of deforestation. If λ_{it} is the area deforested over a given interval divided by the area at risk, and n is the number of years in that interval, our annualized dependent variable (assumed constant during an interval) is

$$h_{it} = 1 - (1 - \lambda_{it})^{1/n}. \tag{7}$$

Explanatory variables

Direct measure of economic returns

The annual return r_{jkt} to a given hectare j in crop k at time t is the crop price p_{kt} times the annual yield per hectare y_{jkt} minus the costs of production $cost_{jkt}$ minus the transport cost t_{jkt}. For each year, we estimate the returns for the four major export crops: coffee, bananas, sugar and beef. We have data from 1950 onward although its quality improves significantly in later years. For each interval, returns are averaged across the years for an average return (in 1997 US$) to crop k on one hectare of cleared land during that interval (see Appendix on data and techniques).

Any parcel is used for one crop at a time. We define s_{jk} as the probability of a crop being chosen as the use of newly cleared land. For larger areas, these probabilities imply expected shares of the area in each crop, to be used as weights in our measure of expected annual return

$$R_{it} = E(r_{ijt}) = \sum_{k} s_{jk} r_{jkt}. \tag{8}$$

We calculate the s_{jk} using data on production patterns in the 1970s and 1980s and information on the suitability of lifezones for crops. In a humid, lower-montane area we represent land choices by assuming that cleared land will be used for coffee or a similar return. The resulting R_{it} is our returns measure, AgReturn. Under most circumstances (though see Angelsen and Kaimowitz, 2001 on effects of both elasticities and endowments), higher returns to agriculture should raise clearing. Forest payments that lower relative returns to agriculture then lower deforestation.

Poverty index

Here we summarize Cavatassi et al.'s (2003) poverty index estimation for Costa Rica. Without sufficient household-level data for a 'small area estimation' approach, they chose to use 'principal components analysis' (PCA). The necessary data are available from the census over

4 decades, permitting a poverty index that can be matched with the deforestation observations.

The data are variables common to multiple censi, at district level. Seventeen variables are common to 1973, 1984 and 2000, of which 12 are common to the 1963 census as well. See Cavatassi et al. (2003) for discussion of judgments about variables' economic meanings and roles in explaining the overall variance in these data. Variables chosen include demographic, labor, education, housing, infrastructure and consumer durable variables. Some examples are the percentage of dwellings without heaters, or without bathrooms, or without electricity. Others are the average number of occupants per bedroom and percent of people receiving job remuneration.

In PCA, eigenvectors of the correlation matrix indicate the direction and the weight of the variables in the index. Cavatassi et al. (2003) find that greater values of variables that should be positive correlated with poverty (% with dirt floor, % without refrigerators) have positive signs in the index, as expected, while wage remuneration and education variables have negative signs, as makes sense. The weights are used to create a poverty index (see Filmer and Pritchett, 1998):

$$\text{Marginality (or Poverty) Index}_j = W_1^*(a_{j1} - a_1)/(s_1) + \dots + W_n^*(a_{jn} - a_n)/(s_n), \tag{9}$$

where W is the weight for a variable (among variables 1–n in Eq. (8)), a_j is the jth district's value for that variable and a and s are, respectively the mean and standard deviation of the variable across the districts.

This method is first used to create year-specific poverty indices for 1963, 1973, 1984 and 2000. Such indices, however, are not comparable over time. Each is based on a scale relevant only to that year. In other words, the indices' units vary, precluding comparison between years. Thus, as a second step, Cavatassi et al., 2003 also pool all years' data to estimate a single PCA for 1973–2000 using the seventeen variables and one for 1963–2000 using the twelve variables. For these pooled PCA estimations, a change in the marginality index arises only from changes in the levels of variables over time, not changes in the relative importance of each variable in the index.

As noted above, some observations must be dropped because of a lack of poverty data. The reason is that the number of districts changes each census year (from 334 in 1963 to 406 in 1973, to 459 in 2000) as older larger districts are split to form newer smaller districts. When they knew how such a split has occurred, Cavatassi et al. (2003) are able to use the poverty values for older larger districts for each of the smaller newer districts into which they split. However, for some districts they were unable to track these changes over time, and thus districts are dropped.

We use the 1963–2000 and the 1973–2000 indices to explore the tradeoff between more years of data and more observations per year. We match to intervals as follows. For the 1963–2000 index, for 1963–1979 we use the 1963 values, while for 1979–1986 we use the 1973 values, for 1986–1997 we use the 1984 values and for 1997–2000 we use the 2000 values. We try using the 1984 values for the 1997–2000 interval also, to have the option of using lagged values. For the 1973–2000 measure the difference is that for 1963–1979 we have only the 1973 values.

A final matching step is to the 436-district structure used by the University of Alberta to organize the forest data and some explanatory data. For years before 2000 we must match the smaller number of census districts to these 436 while for 2000 we match the 459 districts to 436.

We use the indices in continuous form. To focus on greater poverty, e.g. subsistence, we also use quartiles, e.g., to allow for non-linearities within the poverty-deforestation relationship.

Carbon density values

We estimate potential carbon storage in primary forests with the general ensemble biogeochemical modeling system (GEMS) created to incorporate spatially and temporally explicit information of climate, soil, and land cover (Liu and Schimel, 2006). GEMS is a model developed to better integrate established ecosystem biogeochemical models with various spatial databases for the simulations of the biogeochemical cycles over large areas. The well-established model CENTURY (Parton et al., 1987; Schimel et al., 1994) was used as the underlying plot-scale biogeochemical model. It uses a Monte-Carlo-based ensemble approach to incorporate variability (as measured by variances and covariance) of state and driving variables of the underlying biogeochemical models into simulations. The mean values and their corresponding standard deviations of aboveground biomass carbon density simulated by GEMS are listed in Table 1.

Table 1
Carbon stocks

Life zone	GEMS	
	Mean	Standard deviation as (%) of mean
Premontane moist	159	43
Lower montane moist	134	45
Tropical moist	156	24
Premontane wet	156	32
Lower montane wet	113	54
Montane wet	119	34
Tropical wet	336	40
Tropical dry	96	41
Premontane rain	120	49
Lower montane rain	116	58
Montane rain	96	86

Results

Carbon supply

In Table 2, column I addresses whether carbon payments will induce changed land use and sequestration. The significance of the direct returns variable is evidence that they will.

Note the absence of other variables (we do include time dummies and previous clearing, the latter varying over time). Since our returns measure does not capture all possible elements of actual returns, we might wish to include measures of potentially omitted elements. For instance, as transport costs are missing from our returns measure, we might include the distance to the closest of three major Costa Rica markets (which is found to be significant in Kerr et al., 2006). However, these regressions include district fixed effects, so we can not include factors which vary only over space, such as distances and ecological conditions. Fixed effects should control for their effects and for the effects of fixed spatial differences that we cannot directly observe.

Since payments can induce carbon supply, and some land users are poor, the hypothetical 'win–win' is feasible. Carbon payments could generate carbon gains and lessen rural poverty. Whether to target the poor or not, e.g. because they respond more or less, is another question.

Carbon supply by the poor

Poverty's baseline effects

Below we examine whether clearing in poor areas is more or less responsive to carbon rewards. First we note that poorer areas contain more forest per district and also forest per capita. Thus the same rate of deforestation means more forest is cleared. Also, the same responsiveness of clearing to carbon payments, i.e. lowered deforestation, would generate more sequestration.

One reason more forest remains in the poorest areas appears to be that the characteristics of the land the poor

Table 2
Carbon supply by the poor[a,b]

Dependent variable:	I Annualized Def. prob.	II Annualized Def. prob.	III Annualized Def. prob.	IV Annualized Def. prob.
Returns	0.00015 (2.3)	0.00014 (2.2)	0.00004 (0.6)	0.00009 (1.4)
Poverty[c]		0.05 (0.35)	−0.24 (−1.4)	0.17 (3.2)
Returns × poverty			0.00025 (2.7)	−0.000066 (−2.5)
%Cleared	2.1 (2.4)	2.1 (2.4)	2.0 (2.3)	1.2 (1.3)
%Cleared^2	−4.1 (−4.4)	−4.1 (−4.5)	−4.2 (−4.5)	−3.3 (−3.5)
Timedummy 1979–1986	0.79 (9.8)	0.79 (9.8)	0.81 (10)	1.1 (8.8)
Timedummy 1986–1997^2	−0.46 (−4.2)	−0.46 (−4.2)	−0.41 (−3.7)	0.095 (0.44)
Timedummy 1997–2000	−1.9 (−11)	−1.9 (−10)	−1.8 (−9.9)	−1.1 (−3.6)
Constant	−3.2 (−19)	−3.2 (−18)	−3.0 (−16)	−3.6 (−16)
Fixed effects	$F = 6.2$ ($P = 0.00$)	$F = 6.2$ ($P = 0.00$)	$F = 6.3$ ($P = 0.00$)	$F = 6.4$ ($P = 0.00$)
Adjusted R^2	0.75	0.75	0.76	0.76
N	973	973	973	973

Redoing Column III using the 1973–2000 index for the 1979–1986, 1986–1997 and 1997–2000 periods (dropping 63–79), to get more observations of poverty, yields a positive significant result for the interaction with the poorest-quartile dummy while retaining returns and poverty coefficients as in II. Column IV uses the continuous version of the 1963–2000 poverty index for comparing both the direct and the interaction effects of poverty.

[a]All regressions are Grouped Logit regressions, following expression (6) in the manuscript text above.

[b]Coefficients reported with t statistics below them in parentheses (except for fixed effects, where F statistics reported with P values below).

[c]In all columns, 1963–2000 pooled index. Columns II and III use a dummy for the poorest quartile, to focus on the poorest for the interaction.

are on discourage clearing. Pfaff et al. (2006) provide evidence that the land in poorer and richer districts differs enough to affect clearing choices. It appears that the poor are marginalized, i.e. are on land with characteristics that, all else equal, lower deforestation rates.

Controlling for these characteristics, they find that relatively poor areas are more cleared. This is consistent with Table 2's poverty result in Column IV, using a continuous poverty index. However, Pfaff et al. (2006) find a much weaker result for a poorest-quartile dummy variable. That is consistent with Table 2's columns II and III, which employ the poorest-quartile version of the poverty index to focus on the most poor in thinking about who benefits from carbon payments and who might respond differently to them, which is the focus of column III, discussed below.

Poverty's supply effect

Columns III and IV present novel evidence concerning poverty's effect on response to returns and, by implication, to carbon payments that change net agricultural returns. Column III's interaction of the poorest quartile dummy with returns is positive and significant. Taken alone, that coefficient would suggest that poorer people are more responsive to payments. However, comparisons with columns II and IV, and other runs, suggest this result is not robust.

Comparing with column II, which uses the same poverty variable, shows that the returns variable has lost it significance. Perhaps the interaction term is picking up the effect of returns. Also, the poverty variable has shifted to a negative estimated effect in III, inconsistent with other runs for this paper and evidence in Pfaff et al. (2006). Note, though, that the same positive result for the interaction plus consistency with column II arises if using the 1973–2000 poverty index.

Column IV's results for a continuous 1963–2000 poverty index are relatively robust as long as the index is continuous. Most importantly, the interaction is negative and significant. While we acknowledge that poorest-quartile and continuous results may be expected to differ since different stories apply across the income/poverty range, in our view this reversal of sign for the interaction indicates instability. Thus, there may well be no robust significant result here for the poorest group of land users who are usually the focus of "win–win" policy discussions.

This result suggests the lack of an empirical basis for focusing carbon payments programs on the poorest *if* the goal is efficient generation of carbon supply. Of course if a program directly favors poverty reduction as an objective, then targeting the poor will remain the obvious choice.

Simulating carbon supply

We simulate a deforestation baseline and then compare to that the deforestation forecast when the net returns to land clearing for each location are lowered by the carbon price times the carbon per hectare in that location (using Table 1's values). The difference between projections arises solely from the carbon payment and indicates the carbon supply induced by the payments. The baseline assumption for each of the explanatory variables is of course subject to debate (e.g., what output prices will obtain, etc.) but each applies to both the baseline and the payments cases.

We project forward the carbon supply in this way for each of a range of carbon prices. This yields supply or cost curves (Figs. 1 and 2). These convey the estimated responsiveness of sequestration in tropical forest to rewards (the horizontal distance to the curve at each price) and permit an estimate of the cost of sequestering a given level of additional carbon (the integral under the curve up

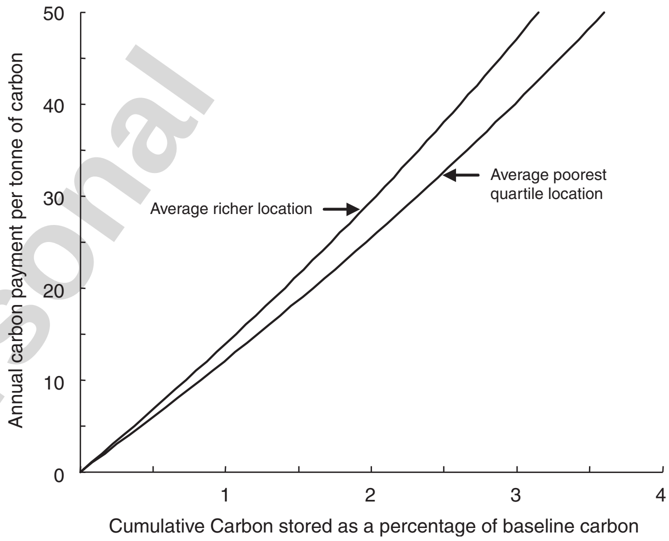

Fig. 1. Carbon supply in 2020 for poorest and less poor districts by assuming no behavioral difference in responsiveness to carbon payments.

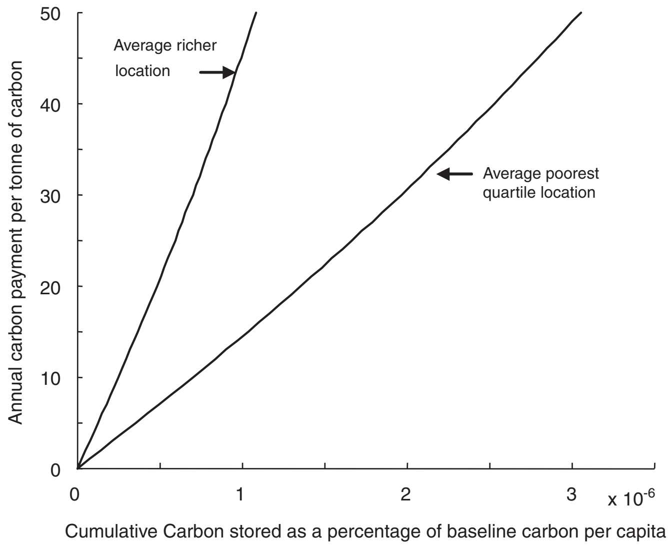

Fig. 2. Carbon supply *per capita* in 2020 for poorest and less poor districts by assuming no behavioral difference in responsiveness to carbon payments.

to that level of carbon). The carbon cheaper to sequester is supplied earlier, and then the carbon gets increasingly expensive as more valuable agricultural land is protected.

Figs. 1 and 2 show two ways to see the implications of results like Table 2's column I when joined with the carbon-per-unit-forest numbers from Table 1. In Fig. 1, carbon supply is slightly higher in the average poor location than in the average rich location at any carbon price.

In Fig. 1, *per capita* results, more forest per capita in the poor areas has a large effect and suggests that payments could significantly benefit the poor. We consider this further below.

Discussion

This paper estimated the potential supply response in Costa Rica to carbon payments and, most innovatively, tested for an effect of poverty on responsiveness to payments. We found that land users will respond to payments and that the poor may respond more, although the latter result is not robust enough to assume different degrees of response in our simulations of carbon supply.

As all types of land users respond to payments and as the poorer areas have more forest, payments could help both forests and the poor. One important caveat concerns the CDM, which excludes avoided deforestation from generating credits. Other payment sources, though, do not.

A second caveat follows from our data being at district level. It may be that in poor areas a large fraction of people are poor but those who own the land are not. If services and payments are proportional to land holdings, payments to *poor areas* would not go mostly to *poor people*.

Our results suggested neither gains nor losses in efficiency from having poor land users in carbon payments programs, a goal that is often expressed. Costa Rica has recently implemented "pro-poor" measures in its environmental service payment program, eliminating the requirement for title to land, simplifying procedures, and providing front-loaded payments (Pagiola et al., 2005). At the international level, efforts to reduce transactions costs include developing simplified methods for small-scale carbon-sequestration projects to come under the CDM (note that information on these guidelines can be found on the UNFCCC website, at http://cdm.unfccc.int/panels/ar).

Another key issue for the poor is risk. Land users may enter into a contract and follow all recommended practices but not yield the contracted sequestration. If they were paid by practice, then those purchasing their credits would assume the risk of carbon storage falling short. If paid for carbon sequestered, land users assume the risk. Equity issues aside, the costs of monitoring practices versus sequestration should affect how contracts are written (Antle and McCarl, 2001).

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Appendix. Direct measure of returns from beef, coffee, sugar and bananas

Units: Crop price is in \$/kg; yield is in kg/ha; production cost is in \$/ha; transport cost is in \$/ha.

Observations: Total of 436 districts in Costa Rica from 1900–1997 in principle, but 1950–1997 in fact. The limitations on historical data mean that we do not have good measures for years before 1950 and more generally even within 1950–1997 the quality of the data is higher for the later intervals.

Prices: Though some production is sold domestically, Costa Rica is a small country and we use exogenous export prices (in 1997 US\$). Price data are taken from two sources, the Costa Rican Ministry of Planning (Vargas and Saenz, 1994) and the Central Bank of Costa Rica website.

Yields: Crop yields vary over time because of technological change, and across space because of differences in general productivity and in suitability for particular crops. While lifezones and soils proxy for this variability, here we estimate yield. For instance, in some areas the yield for a particular crop is effectively zero since it would never be grown there. Our data is of two types.

For some crops we have data on yield per hectare: for bananas, county level for 1977–1997, and given no obvious trend we assume this to be constant before 1977; for sugar, province level for several years between 1950 and 1977 and for county level in 1998, and we apply the province-level trends to extrapolate the yields for all counties within a province before 1998.

Else we observe production (kg) and area in production (ha) and divide to get the yields. For coffee we have production from 1974–1992 and 1996 at county level and area at county level from the census for 1950, 1955, 1963, 1973 and 1986. We assume production is fixed pre-1974 and area is fixed post-1986, and then interpolate the coffee areas before calculating yield ratios. For pasture we use national production from 1950 to 1995 and divide by census estimates of area for a national yield estimate. We create county-level variation by utilizing the ratio of number of cattle to pasture in the census data, assuming this variation is related to productivity. In locations where the yields for particular crops are undefined within our data, we assume that they are zero.

Costs: We estimate operating costs on an annual basis, although the data are sparse. For coffee, we observe costs only in 1979 and 1981 by coffee zone. For beef we have a single reliable estimate from 1974 at the national level. For sugar, data is better although still at national level, with estimates from Barboza et al., 1982 and Chaves-Solera, 1994 for 1963, 1966, 1972, 1977, 1979 and 1994–1996. For bananas we have a technical estimate from Hengsdijk (pers. commun., Wageningen Agricultural University) for 1997, but no previous data. These are assumed constant outside the period within which they are observed and interpolated. For transport costs, lacking direct measures, currently we rely upon the proxy described above.

Crop shares: To predict how likely each of the four crops is to be chosen, we use a combination of census and satellite land-use data to estimate the share of each crop in each district. While the satellite data are more precise, they distinguish not crops but simply land uses (permanent crops, pasture, and forest). The data is from 1973 and 1984, and our shares do not change over time.

We combine the district shares with the crop returns for expected annual return per district-year, following Eq. (6) above. Then we average the returns across intervals to generate mean returns to which we assume our estimated constant annual interval deforestation rate will have responded.

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