Modeling Opportunity Costs of Conservation in Transitional Landscapes

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Abstract: *Conservation scientists recognize the urgency of incorporating opportunity costs into conservation planning. Despite this, applications to date have been limited, perhaps partly because of the difficulty in determining costs in regions with limited data on land prices and ownership. We present methods for estimating opportunity costs of land preservation in landscapes or ecoregions that are a changing mix of agriculture and natural habitat. Our approach derives from the literature on estimating land values as opportunity costs of alternate land uses and takes advantage of general availability of necessary data, even in relatively data-poor regions. The methods integrate probabilities of habitat conversion with region-wide estimates of economic benefits from agricultural land uses and estimate land values with a discount rate to convert annual values into net present values. We applied our method in a landscape undergoing agricultural conversion in Paraguay. Our model of opportunity costs predicted an independent data set of land values and was consistent with implicit discount rates of 15–25%. Model-generated land values were strongly correlated with actual land values even after correcting for the effect of property size and proportion of property that was forested. We used the model to produce a map of opportunity costs and to estimate the costs of conserving forest within two proposed corridors in the landscape. This method can be applied to conservation planning in situations where natural habitat is currently being converted to market-oriented land uses. Incorporating not only biological attributes but also socioeconomic data can help in the design of efficient networks of protected areas that represent biodiversity at minimum costs.*

Key Words: agricultural conversion, conservation planning, deforestation, developing country, landscape, mapping, Paraguay

Modelado de Costos de Oportunidad de la Conservación en Paisajes de Transición

Resumen: *Los científicos de la conservación reconocen la urgencia de incorporar costos de oportunidad en la planificación de la conservación. A pesar de ello, las aplicaciones han sido limitadas a la fecha, en parte quizás por la dificultad en la determinación de costos en regiones con datos limitados sobre los precios y pertenencia de tierras. Presentamos métodos para la estimación de costos de oportunidad de la preservación de tierra en paisajes o ecoregiones con una mezcla cambiante de hábitat agrícola y natural. Nuestro método deriva de la literatura sobre la estimación de valores de tierras como costos de oportunidad de usos de suelo alternativos, y aprovecha la disponibilidad general de los datos necesarios, aun en regiones relativamente pobres en datos. Los métodos integran las probabilidades de conversión de hábitat con estimaciones regionales de los beneficios económicos de los usos de suelo agrícolas, y estiman los valores de tierras con una tasa de descuento para convertir los valores anuales en valores actuales netos. Aplicamos nuestro método en un paisaje en proceso de conversión agrícola en Paraguay. Nuestro modelo de costos de oportunidad predijo un conjunto de datos independientes de valores de tierras y fue consistente con las tasas de descuento implícito de 15–25%. Los valores de tierras generados por el modelo estuvieron muy correlacionados con los valores actuales aun después de corregir por el efecto del tamaño de la propiedad y la proporción forestada de la misma. Utilizamos el modelo para producir un mapa de costos de oportunidad y para estimar los costos de conservar bosques dentro de*

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dos corredores propuestos en el paisaje. Este método se puede aplicar en la planificación de la conservación en situaciones en las que el hábitat natural esta siendo convertido a usos de suelo orientados al mercado. La incorporación no sólo de atributos biológicos sino también de datos socioeconómicos puede ayudar al diseño de redes eficientes de áreas protegidas que representen la biodiversidad al mínimo costo.

Palabras Clave: conversión agrícola, deforestación, mapeo de paisaje, país en desarrollo, Paraguay, planificación de la conservación

Introduction

Resources for the conservation of biodiversity are scarce and therefore should be allocated as efficiently as possible. The science of conservation planning aims to address this need and has moved forward rapidly in the last decade (Margules & Pressey 2000). Initial contributions focused on calculating optimal or close-to-optimal solutions for the placement of conservation reserves based on presence/absence of species (Pressey et al. 1993; Church et al. 1996). In recent years advances in statistical modeling, reserve design algorithms, and the availability of new software tools have all served to broaden both the theory and the practice of this relatively new discipline (Cabeza & Moilanen 2001). Current research includes modeling probabilities of persistence (Williams &Araujo 2000), incorporating spatial considerations (Onal & Briers 2002), using environmental surrogates (Faith et al. 2003), and using metapopulation models (Cabeza & Moilanen 2003).

The impressive gains made in conservation planning belie the fact that all these leaps forward have been biological or computational in nature. For all the intensity of research focus that has been placed on biological aspects of systematic conservation planning, corresponding research on economic facets has been largely missing. Yet it is in this area that the most significant gains in insight are likely to come from (Moore et al. 2004), and at some level economic factors enter into every policy decision, including (and sometimes especially) those involving conservation of natural habitats. As the use of methods and theory from social sciences such as economics, sociology, and political economy continues to increase in conservation biology, we expect that a similar infusion of techniques from such disciplines will occur in conservation planning. For the moment, however, there appears to be a significant lack of investigation into how techniques from the economic sciences can be used to increase the efficiency of conservation planning.

Studies that have incorporated economic costs into conservation planning have followed two general tracks. One approach focuses on using surveys of protected area managers to estimate the costs of establishing a network of conservation reserves that protects 10–15% of natural regions (James et al. 1999) and then integrates these data with biological values to guide priority setting (Balmford et al. 2000). Results of this approach highlight the considerable cost-efficiency that can be obtained by integrating cost data in conservation strategies. They also suggest that global conservation costs are relatively low, less than one-fifth the value of environmentally damaging government subsidies. Similar results were obtained for conservation in Africa (Moore et al. 2004) and for marine protected areas (Balmford et al. 2004). It is important to note, however, that this approach does not account for the costs of actually acquiring land for protection, which is likely to exceed by many times subsequent management costs (Balmford et al. 2003).

Building on algorithms that predict the "best" network based on the spatial pattern of species distributions, several researchers have asked how incorporating costs of various reserve configurations or land-use patterns affects conservation of biodiversity (Ando et al. 1998; Montgomery et al. 1999; Polasky et al. 2001; Ferraro 2003; Nalle et al. 2004; Polasky et al. 2005). These studies used the explicit or implicit market value of land to estimate the opportunity costs involved in setting aside protected areas. Opportunity costs are the costs associated with foregone alternatives, in this case the benefits that could have been derived from natural habitat had it been converted to a profitable land use. Results showed that reserve design algorithms that include costs can result in similar numbers of species protected but substantially lower expenditures than conservation programs that ignore opportunity costs.

A key limitation of extending these approaches beyond the few case studies above is data scarcity. Although countries such as the United States collect and frequently update detailed information on land values down to the county level (Veneman et al. 2004), such fine-scale and systematically organized data are more difficult to find in developing countries. Yet these countries are often of highest priority for conservation (Balmford et al. 2003). In addition, official markets for land transactions may not exist in many developing countries with weak property rights and institutions. In these situations, what approaches can serve as a proxy for or estimate of land values in areas where conservation planning is under way or imminent?

We estimated spatially explicit land values by integrating the spatial probability of land conversion with known returns to agriculture. We applied our model to a forested landscape in Paraguay that is undergoing agricultural conversion and validated it with a small reference set of property values. Using the resulting map of land values, we estimated the cost of conserving a proposed corridor design. We conclude with a discussion of the opportunities and limitations of the methodology and the importance of continued incorporation of economic costs into conservation planning.

Methods

We focused on the opportunity costs of acquiring or otherwise protecting land for conservation. Our method for estimating opportunity costs is based on two components: observed spatial patterns in conversion of natural habitat to agricultural land use and observed net benefits ("rents") from typical agricultural plots in the region.

Theoretical Framework

The literature on calculation of agricultural land values has generally followed a net present value or asset value framework. In this approach it is assumed that the annual rental value of a parcel of agricultural land is equal to its annual net revenue flow, whereas the sale value of a parcel of land is equal to the discounted flow of net revenue that the parcel is expected to generate into the future (i.e., its net present value; Weersink et al. 1999; Cavailhes & Wavresky 2003; Goodwin et al. 2003). In relatively remote agricultural areas, where urbanization is not an issue, and in situations in which government subsidies to market-oriented farmers are absent, these net rents are influenced exclusively by agricultural input and output prices and crop productivity. We assessed land values as the expected value of land arising from agricultural uses, where the expectation is taken over the probability that the land is converted to agricultural use.

Suppose one observes a number of parcels of land and wishes to estimate the expected land value for the entire set of parcels. If the economically "best" use of the land is in agriculture, the land value is expected to be the capitalized value of the annual net economic benefit derived from agriculture. Across the whole set of parcels there may be several different types of agriculture practices, each with their own associated economic benefits.

Suppose also that much of the land is forested and can, in any year, be converted to one of the several agricultural uses that occur in the region. There is a probability that a parcel of land will be converted to an agricultural use in any given year, as a function of characteristics of the parcel (e.g., soil conditions, proximity to markets, ownership). This probability is P_{itk}, where P is probability, i is parcel, t is time period, and k is type of agriculture (hereafter referred to as land use).

In a particular period, the expected annual return (R_t) can be written as the product of the probability of being in agriculture (P_{itk}) times the return associated with agricultural land use k (R_{itk}), summed over all land-use types. Over the set of parcels this expected return is

$$R_t = \sum_{k=1}^{K} \sum_{i=1}^{I} P_{itk}\, R_{itk}. \qquad (1)$$

The expected capitalized value (EV) of the parcels is the sum of the expected discounted returns over time (discount rate $= \delta$).

$$\mathrm{EV} = \sum_{t=0}^{T} \sum_{k=1}^{K} \sum_{i=1}^{I} (P_{itk}\, R_{itk}) * (1/(1+\delta)^t). \qquad (2)$$

If the probability of conversion of a parcel is constant over time and the returns associated with agriculture are assumed invariant with regard to spatial location, then the value becomes

$$\mathrm{EV} = \sum_{k=1}^{K} \sum_{i=1}^{I} P_{ik} \sum_{t=0}^{T} (R_{tk})(1/(1+\delta)^t). \qquad (3)$$

As t goes to infinity the expected value becomes

$$\mathrm{EV} = \sum_{k=1}^{K} \sum_{i=1}^{I} P_{ik} \frac{R_k}{\delta}, \qquad (4)$$

which is the probability of conversion times the capitalized value of the parcel, summed over the set of parcels and land uses. In such an analysis the expected value of the set of parcels can be calculated using only information on the returns, discount rate and probability of land being in agriculture (or probability of land being converted from forests to an agricultural land use). We used Eq. 4 as our model of land values, or opportunity costs of conservation.

There are several assumptions associated with a model of this type, which can be relaxed in the following ways. First, we assumed the value of land not in agriculture is zero. If not, then the value of parcels has an additional term that adds the value of nonagricultural land times the probability of land being in nonagricultural uses, or

$$R_t = \sum_{k=1}^{K} \sum_{i=1}^{I} P_{itk}\, R_{itk} + (1 - P_{itk})\mathrm{NR}_{itk}, \qquad (5)$$

where NR_{it} is the return from nonagricultural uses (e.g., sustainable forestry, collection of nontimber forest products).

Second, we assumed that conversion costs are zero. If not, then the returns from agriculture should be adjusted to include the conversion costs:

$$R_t = \sum_{k=1}^{K} \sum_{i=1}^{I} P_{itk}\, R_{itk} - C_i + (1 - P_{itk})\mathrm{NR}_{itk}, \qquad (6)$$

where C_i is the one-time cost of converting forest to agricultural use, net of potential benefits such as timber sales.

Third, for the capitalized value version of the expression to be valid, the rents from agriculture must not change over time. If they do, then a more complicated expression must be used.

Fourth, the probability of conversion is assumed to be fixed over time. This could be valid if technologies do not change and if major changes in infrastructure (e.g., roads) do not occur. If major changes in factors affecting the probability occur over time, probabilities need to be re-estimated with updated variables that reflect the period of interest. The probabilities are a function of land rents, so the calculations above can be expressed in a more complex fashion $[P_{itk}(R_{itk}) * R_{ikt}]$. But because the data we have for rent are not spatial, it is more reasonable to assume that the aspatial rent is a constant, whereas the probability of conversion captures the spatial variation around that estimate.

Finally, the model is highly stylized. It does not include social or cultural factors that may be determinants of land conversion in particular areas, such as clearing forested land as a strategy to claim territory. It also does not reflect situations in which farmers are pursuing goals other than maximizing their household's economic welfare, such as when farmers cultivate land for communal benefit or for social obligation (e.g., Laney 2002). In areas where these issues may be important in explaining land conversion, the model would have to be expanded to account for them or an entirely different approach should be taken. In these cases the opportunity cost of conservation will not be directly related to the land value arising from the production of crops.

Example of Opportunity Cost Modeling

The Mbaracayu Forest Biosphere Reserve in eastern Paraguay is within the highly threatened Upper Parana Atlantic Forest ecoregion. In 1973 this reserve (which follows the boundaries of the Upper Jejui watershed, an area of 2920 km^2) was 90% forested. In 2004 the area of forest in the watershed was down to 56% and, with the exception of a central core of protected area and several forests on adjacent private lands, was becoming highly fragmented. As of 2004 there were three dominant land uses in the region: smallholder agriculture (12% of land surface), large-scale cattle ranching (14%), and soybean production (2.4%).

We used the following steps to calculate the opportunity costs of conservation in the Mbaracayu Forest Biosphere Reserve. (1) We obtained satellite images of the study area for two dates: 1990/1991 and 2004. (2) We obtained spatial coverages on factors expected to influence probability of agricultural conversion on a plot of land. These were developed through interpretation of imagery and existing data sets. (3) We built a statistical model of probability of habitat conversion with methods for model selection and evaluation. (4) We obtained data on output prices and input costs of agricultural production for all land uses in the study area. (5) We calculated an average net rent or land value for the area in question based on data from 4. (6) We multiplied (and summed for different land uses) results from 5 by results of 3 to obtain a spatial map of land values. (7) We validated the model results with reference data.

Of the two key parameters in our model (P_{ik} and R_k), the probability of conversion to agriculture P_{ik} was the more complicated to estimate because it varies spatially, not just among land-use categories. To estimate P_{ik} we needed information on both the spatial pattern of habitat conversion over recent time and spatially referenced information on factors that are likely to have influenced this pattern. If not already available as GIS coverages, much of this information can be derived from satellite imagery (Turner et al. 2003). We used Landsat satellite imagery from 1990/1991 (the study area spanned two scenes) and 2004 to classify land cover into a variety of forest and nonforest types (forest classification methods, Naidoo & Hill, 2006). For our purposes we reclassified land cover into two classes: forest and nonforest. We also identified four different land-use classes from both spectral and spatial patterning elements of land cover in the watershed: cattle ranching, smallholder "colonies," soybean farming, and unknown or indeterminate land use. For each of the three known land-use classes, we produced coverages containing forested and deforested areas and features likely to have influenced deforestation.

There is a large literature on empirical modeling of deforestation (reviewed in Angelsen & Kaimowitz 1999), which we consulted to develop a list of candidate explanatory variables. These variables included biophysical conditions, human infrastructure, and known land-tenure boundaries (Table 1). Spatial data layers for these variables were either interpreted from satellite images (1990 road network, 2004 land-use classes) or obtained from archives of the Fundacion Moises Bertoni, a local nongovernmental organization that works extensively in the area (land-tenure information, soil types, rivers).

We modeled the probability of deforestation for each land-use class with a multinomial logistic regression analysis. Such models estimate the probability of an event occurring in each of the multiple classes being considered, with probabilities across classes summing to 1. Robust estimation of probabilities from binary logistic regression modeling requires that observations in the two classes not be overly unbalanced. King and Zeng (2001) and Cramer (1999) make suggestions as to how unbalanced classes can be before severe biases in prediction probabilities occur. It is unclear how these rules of thumb relate to multinomial models (Cramer 1999), but only one of our three agricultural classes (soybean, $n = 27$) was greatly underrepresented in relation to the largest class (still forest, $n = 958$), and subsequent receiver operating characteristic analyses (see Results) showed that this class

Table 1. Variables used in multinomial logistic regression analysis of forest conversion, Mbaracayu Forest Biosphere Reserve.

			Entire study area (n = *228,225*)				*Sample for regression model* (n = *1,304*)			
Variable[a]	*Units*	*Group*[b]	*mean*	*SD*	*min*	*max*	*mean*	*SD*	*min*	*max*
x	UTM	spatial	667,494	19,035	625,349	704,549	666,921	18,588	625,967	703,272
y	UTM	spatial	7,317,440	12,948	7,287,911	7,348,711	7,317,695	13,275	7,288,593	7,345,904
Elev	meters	biophysical	235	53	132	476	236	54	134	465
Slope	degrees	biophysical	3	2	0	32	3.34	2.30	0	22.2
Dist.roads	meters	infrastruc.	2,694	2,667	0	12,762	2,833	2,745	0	12,233
Dist.rivers	meters	biophysical	445	377	0	2,648	457	373	0	2,200
Dist.centroid	meters	infrastruc.	24,157	11,842	0	54,764	23,659	11,958	583	54,099
Dist.ruta10	meters	infrastruc.	19,525	12,667	0	51,182	19,941	12,846	200	48,301
Dist.towns	meters	infrastruc.	27,068	10,500	316	49,341	27,026	10,517	2,236	48,846
Dist.foredge	meters	spatial	321	450	0	3,204	352	461	0	2,759
Are.rho.pale	dummy	biophysical	0.536	0.50	0	1	0.539	0.50	0	1
Lith.hapl	dummy	biophysical	0.006	0.08	0	1	0.005	0.07	0	1
Aquic.pale	dummy	biophysical	0.129	0.34	0	1	0.129	0.34	0	1
Rhodic.pale	dummy	biophysical	0.090	0.29	0	1	0.096	0.30	0	1
Typic.pale	dummy	biophysical	0.043	0.20	0	1	0.029	0.17	0	1
Rhodic.kandalf	dummy	biophysical	0.040	0.20	0	1	0.044	0.20	0	1
Rhodic.kandox	dummy	biophysical	0.035	0.18	0	1	0.043	0.20	0	1
Typic.quartz	dummy	biophysical	0.121	0.33	0	1	0.114	0.32	0	1
Reserve	dummy	tenure	0.262	0.44	0	1	0.291	0.45	0	1
Indig	dummy	tenure	0.063	0.24	0	1	0.071	0.26	0	1
Estancia	dummy	tenure	0.158	0.36	0	1	0.159	0.37	0	1
Colonia	dummy	tenure	0.065	0.25	0	1	0.071	0.26	0	1

[a] *Abbreviations: x, east-west UTM coordinate; y, north-south UTM coordinate; elev, elevation; dist.roads, distance to the nearest road; dist.rivers, distance to the nearest river; dist.centroid, distance to the geographical center of the core protected area; dist.ruta10, distance to the only paved road in reserve; dist.towns, distance to nearest town; dist.foredge, distance to the edge of the forest; are.rho.pale, lith.hapl, aquic.pale, rhodic.pale, typic.pale, rhodic.kandalf, rhodic.kandox, typic.quartz, dummy variables for various soil types; reserve, dummy variable for core protected area; indig, dummy variable for indigenous reserves; colonia, dummy variable for smallholder "colonies"; estancia, dummy variable for large landholdings.*

[b] *Each variable was defined as belonging to a group. Individual groups and all possible combinations were used as candidate models for model selection by an information-theoretic approach (Burnham & Anderson 2001) (spatial, pure spatial effects on habitat conversion; topographic, topographical effects; infrastruc., human infrastructure effects; soil type, soil condition effects; tenure, land tenure effects).*

had very high predictive power. Observations of forest conversion to smallholder colonies ($n = 161$) and cattle ranching ($n = 158$) were frequent enough to not qualify as unbalanced or "rare events" according to King and Zeng (2001). Therefore, our selection of satellite image dates resulted in a data set of land-use conversion that allowed robust estimation of conversion probabilities.

We used an information-theoretic approach and Akaike's information criteria (Burnham & Anderson 2001) to select a best-fitting model from among a priori models based on variable types (Table 1). We checked these models for spatial autocorrelation in the residuals. Significant autocorrelation was present, which we attempted to account for in two ways. First, we added the x and y coordinates and the xy product as a measure of broad spatial trends (Legendre & Legendre 1998). Further polynomial expansion resulted in complete separation and nonconvergence of some regression models in preliminary analyses. Second, we identified the distance at which observations became spatially independent by examining spatial correlograms of model residuals and identifying distance thresholds at which the correlation coefficients had declined to very low (nonsignificant) levels. We then randomly sampled the landscape for points at distances greater than this threshold and used only these observations in multinomial models. This approach significantly reduced, but did not entirely eliminate, spatial autocorrelation effects.

We then used the best-fitting model to generate probabilities across the whole study landscape. Most of variables stayed constant over time; distances to roads and to forest edges, however, changed substantially because of increases in the road network and forest clearance, respectively. We therefore used updated (to 2004) values for these variables when calculating spatially explicit conversion probabilities across the landscape.

To estimate annual economic rents (R_k) we used agricultural data from a variety of sources, most of them specific to the Mbaracayu Forest Biosphere Reserve itself (Table 2). The primary reference was Jazmin (1995), which provided detailed estimates of market prices and production costs for smallholder crops grown in the department of Canindeyu in Paraguay, of which about one-third is the reserve. We used information from Renshaw et al. (1989) for data on crop distribution. For cattle ranching, detailed surveys of two cattle ranches

Table 2. Capitalized net economic benefits (i.e., R_k/δ) associated with agricultural land uses in the Mbaracayu Forest Biosphere Reserve, Paraguay.*

	Net present value (%), δ							
Land use	*5*	*10*	*15*	*20*	*25*	*30*	*35*	*40*
Smallholder agriculture	770	385	257	192	154	128	110	96
Cattle ranching	1124	562	375	281	225	187	161	141
Soybean farming	4040	2020	1347	1010	808	673	577	505

**All values are U.S. dollars per hectare of land. More detailed data on net economic rents for each land use are available upon request from the authors.*

within the reserve were used to estimate annual cattle production and costs (Fundacion Moises Bertoni, unpublished data). The business pages of an Asuncion daily (*Diario ABC*; www.abc.com.py) were consulted for current prices of live cattle. For soybean farming, we took information on average production costs in areas bordering Paraguay (northern Argentina and southern Brazil) from Muzzi (2002). Soybean commodity prices have been volatile over the past 4 years, peaking at US$364 in April 2004 before declining to US$210 in September 2004. We therefore used the 3-year average price (Chicago Board of Trade) in our net rent calculations.

The final parameter in our model represented in Eq. 4 was the discount rate, δ. Choosing an appropriate discount rate in applications such as cost-benefit analysis is the source of much controversy (Kasting & Schultz 1996). Because the choice of any one particular discount rate is highly uncertain and because calculated land values will be highly sensitive to the chosen rate, we estimated land values based on a range of discount rates (from 5 to 40%) and assessed the range for which our results are robust.

We collected a small reference set of property values ($n = 21$) against which to verify our model results. Most of these ($n = 16$) were from landowner surveys, in which we asked the property owner to assess the sale value of his or her property. The remaining values were from recent market transactions. In all cases we estimated the boundaries of properties by collecting a number of GPS points along the property boundary. Properties ranged from 6 to 4750 ha. Most were owned by smallholders ($n = 15$), although we also surveyed three cattle ranches, two soybean farms, and one forested property that was originally purchased for cattle ranching but later sold to the Paraguayan government as an indigenous reservation. The amount of forest on properties varied from none to 92% of surface area, with a mean of 38%. We estimated property values by summing within property boundaries the predicted rent values for areas still forested and the average agricultural rents of relevant land-use categories for area that was already converted.

Results

The information-theoretic procedure of model selection identified the best-fitting model as one that contained all explanatory variables except for the spatial coordinates (Table 3). The multinomial logistic model performed much better than independent logistic models for each land-use class based on preliminary analyses, and the area under the curve of receiver operating characteristics for each land-use class was indicative of a very good fit (smallholder, 0.90; ranching, 0.85; soybean, 0.96; compared with random, 0.50). First-order spatial autocorrelation was reduced from an average of 0.55 to an average of 0.20 after sampling of points >1000 m apart.

The strongest explanatory variables in the model were those coding for soil type, land tenure, and topography. Typic pale and rhodic kandalf soil types were strongly and negatively associated with conversion to soybean farming, whereas areas deforested for smallholder and ranching land uses had similar soil type associations, with the exception of aquic pale soil (negative for smallholder, positive for ranching). Coefficients on dummy variables coding for core area of the reserve and indigenous reservations were significantly negative across all land uses, indicating greatly lowered probabilities of deforestation in these areas. As expected, *colonias* (smallholder colonies) were associated with smallholder deforestation, whereas large landholdings were associated with conversion to ranching and soybean farming. Slope was a strong negative explanator of forest conversion to all three land uses. Measures of distances to the nearest roads, towns, and forest edges were mostly in the expected direction (negative), but were not as strongly related to forest conversion for any of the three land uses.

Model-estimated property values at discount rates from 5 to 40% were compared with self-assessed or sale property values with simple linear regression. Models yielding unbiased predictions were indicated by slopes not significantly different from 1 and y-intercepts not significantly different from 0. For discount rates of 5, 10, 30, 35, and 40%, the null hypothesis of no significant difference in slope (from 1) and intercept (from 0) was rejected, and therefore these models generated biased predictions (e.g., linear hypothesis test, discount rate of 5%: sum of squares = 9.63, $F = 11.83112$, df = 2,19, $p = 0.0005$). Estimates from models at 5% and 10% were significantly greater than actual property values, whereas model estimates with discount rates of 30, 35, and 40% were smaller than actual values. For discount rates of 15, 20, and 25%, hypothesis tests revealed intercepts not different from 0 and slopes not different from 1, indicating that model-predicted estimates of property values were unbiased predictors of actual property values (e.g., linear hypothesis test, discount rate of 20%: sum of squares = 0.965, $F = 1.29$, df = 2,19, $p = 0.30$; Fig. 1). Regardless of whether models were biased or unbiased, all predictions

Table 3. Logistic regression model of forest conversion to agricultural habitat, 1990–2004, for three land-use types in Mbaracayu Forest Biosphere Reserve.

	Smallholder		*Ranching*		*Soybean*	
*Variable**	*estimate*	*SE*	*estimate*	*SE*	*estimate*	*SE*
Intercept	0.98	1.73E-06	−1.22	2.57E-06	−18.47	2.06E-06
Dist.roads	−1.59E-04	8.49E-05	−6.09E-05	7.67E-05	2.84E-06	1.18E-04
Dist.centroid	7.36E-06	1.28E-05	−8.02E-05	1.36E-05	3.80E-05	3.46E-05
Dist.ruta10	−7.05E-05	1.09E-05	−9.82E-06	1.04E-05	1.43E-06	3.37E-05
Dist.towns	−1.18E-05	1.13E-05	8.58E-06	1.05E-05	−1.72E-05	2.61E-05
Are.rho.pale	1.14	5.14E-06	2.14	1.93E-06	3.99	1.90E-06
Lith.hapl	−6.53	1.01E-11	−16.26	2.34E-15	−15.97	1.50E-16
Aquic.pale	−0.43	2.38E-06	1.45	3.17E-06	6.34	5.27E-06
Rhodic.pale	1.88	5.57E-06	2.41	2.37E-06	5.12	4.51E-06
Typic.pale	0.21	6.22E-07	1.52	6.89E-07	−12.93	9.67E-15
Rhodic.kandalf	2.36	8.23E-07	1.25	2.28E-07	−12.49	5.17E-14
Rhodic.kandox	1.45	2.64E-06	3.42	3.53E-06	4.78	1.05E-05
Typic.quartz	0.91	2.14E-06	2.85	5.08E-06	2.70	3.22E-06
Elev	−0.0087	2.62E-03	0.00097	2.20E-03	0.032	4.36E-03
Slope	−0.016	1.14E-05	−0.13	2.19E-05	−0.011	1.44E-05
Dist.foredge	−0.0012	8.04E-04	−0.00032	7.69E-04	−0.00040	1.70E-03
Dist.rivers	0.0007	3.03E-04	0.00031	2.99E-04	0.00083	7.01E-04
Reserve	−22.86	4.79E-16	−24.57	2.51E-15	−32.95	6.54E-19
Indig	−2.73	1.33E-06	−1.88	6.75E-07	−14.83	7.68E-13
Estancia	−25.49	1.05E-16	0.75	4.14E-06	2.65	1.03E-05
Colonia	2.19	5.77E-06	−25.61	3.55E-17	−13.32	4.41E-13
n	1304					
AIC	1514					
Deviance explained	0.35					
Chi-square	754					
df	20					
p	<0.00001					

**Variables defined in footnote of Table 1.*

were strongly correlated with actual property values, with R^2 values of about 0.89 (e.g., Fig. 1).

To confirm the robustness of the relationship between model-predicted and actual property values, we estimated multiple regressions that included property size and proportion of property that was forested as covariates. For discount rates of 15, 20, and 25%, property size was a significant positive explanator of actual property value (e.g.,

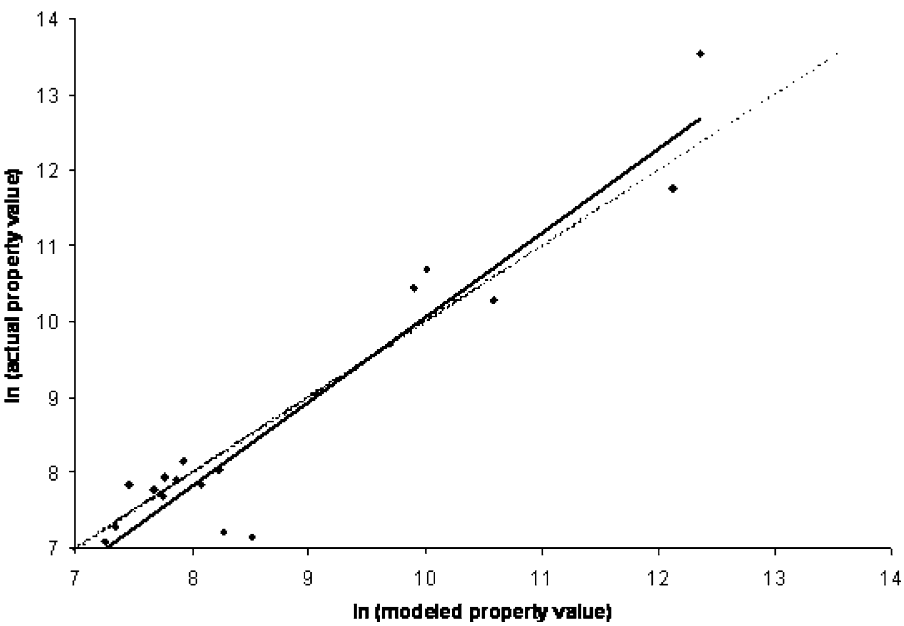

Figure 1. Actual versus modeled property values (logged axes), Mbaracayu Forest Biosphere Reserve, discount rate = 20% (solid line, least-squares linear relationship [y = 1.14x − 1.16, R^2 = 0.89]; dashed line, 1-to-1 line).

discount rate of 20%: coefficient = 0.57, p = 0.019), whereas the coefficient on proportion of property that was forested was not significantly different from 0 (coefficient = −0.18, p = 0.73). The relationship between model-predicted and actual property value was still positive and significant (coefficient = 0.62, p = 0.017), whereas the overall model explained 92% of the variance and was highly significant. Restricting the analysis to the set of properties where forest was at least 48% of the surface area did not change our results substantially (e.g., discount rate of 20%: property size, coefficient = 0.53, p = 0.09; model-predicted values, coefficient = 0.62, p = 0.07, $R^2 = 0.94$, $n = 9$).

Mapping of the average spatial net rents over the three dominant land uses (smallholder agriculture, cattle ranching, and soybean farming) showed a high degree of spatial heterogeneity in estimated land values for the Mbaracayu Forest Biosphere Reserve (Fig. 2). Most of the high-value land was concentrated in the extreme east of the reserve. Forested land in this area is highly likely to be converted to soybean plantations (the most economically valuable land use) because it is at the edge of the extensive soybean belt that overlays the fertile soils shared by some of the most productive parts of Paraguay, Brazil, and Argentina. Most of the rest of the land is of substantially lower economic value but variability in price is still apparent. Large "blocks" of similar economic value, such as the indigenous reserve directly south of the core protected area, reflect the strong effects of land tenure on land price.

These spatially explicit land values can be used to evaluate the costs of conservation plans in the Mbaracayu Forest Biosphere Reserve. Environmental organizations working in the reserve are concerned about maintaining and enhancing connectivity among remaining forests within the area and between the core protected area and other protected areas within the ecoregion. One example of this type of scheme is presented in Fig. 2. The corridors shown are part of a conservation plan included in a project funded by the Global Environment Facility and the World Bank addressing sustainability issues within the reserve. Modeled land values can be used to estimate the opportunity costs of conservation within these corridor designs. Assuming land values with a discount rate of 20%, the total cost of compensating an owner of all forested land within the proposed boundaries was estimated to be \$1,457,282. Much of this land follows riparian corridors, especially in the two smaller corridors west of the core protected area. Breaking down the costs, the larger corridor in the eastern part of the reserve encompasses the lion's share of the costs, at \$1,273,493, whereas the two smaller corridors in the west would cost \$77,585 and \$106,204, respectively. These detailed estimates of the economic costs of conservation can be integrated with biological information to guide priority setting for conservation investments.

Discussion

Empirical Results and Caveats

Our predictions for property values in the Mbaracayu Forest Biosphere Reserve were consistent with actual values for discount rates of between 15 and 25%. Although these may seem high, recent studies on individual discount rates have shown rates in developed countries as high as 19–22% (United States, Coller & Williams 1999) and 28% (Denmark, Harrison et al. 2002). Rates in developing

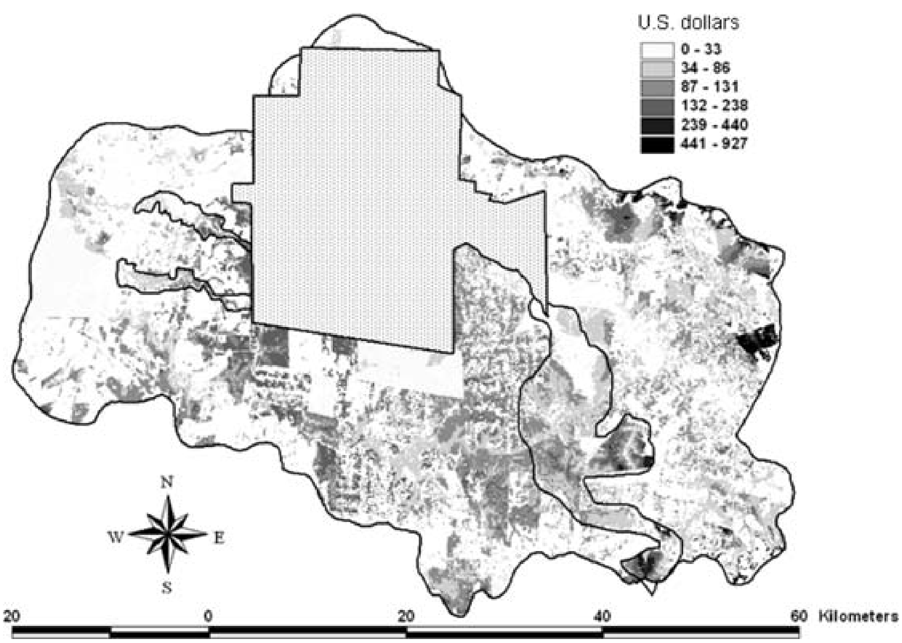

Figure 2. Predicted land values, Mbaracayu Forest Biosphere Reserve (2004 U.S. dollars per hectare). Stippled region is core protected area. Proposed corridors outlined in black.

countries where land tenure is uncertain, livelihoods are strongly tied to uncertain environmental conditions, and general economic welfare is lower, can be even higher (Holden et al. 1998). Uncertainty surrounding discount rates is one important reason to calibrate cost models with reference data whenever possible so that potential biases in estimates can be corrected. Even a relatively small reference data set such as ours allows corrections for biases in the magnitude and direction of predictions.

Although we focus on the value of land that may not currently be in agricultural use, it is worth noting that estimating values for existing tracts of agricultural land is often not a trivial exercise. Land markets can be thin and there may be barriers to sales. Such difficulties have led researchers to employ experimental economics techniques such as auctions to assess values (Stoneham et al. 2003). Our approach can provide estimates of land values at lower costs but it should be recognized that these experimental techniques probably provide greater accuracy and will be more representative of the values used in actual behavior of landowners. If estimates of these values are available, they can be used in place of the rent calculations we used and can be incorporated with the probabilistic assessment of land-use changes to form measures of expected values of land.

We did not consider the possibility that different anthropogenic land uses may transition among one another. Large-scale soybean farming has only recently come to the Mbaracayu landscape in Paraguay. If prices remain high, cattle ranches may be converted to this more profitable land use, albeit at a probability that is currently unknown. Other researchers have shown that potential conversion probabilities (e.g., from agriculture to urban land use) are capitalized into agricultural land values (Plantinga et al. 2002).

As in any empirical modeling exercise, the accuracy of modeled opportunity costs of conservation will be correlated with the quality of input data. The more closely one can estimate and account for actual economic behavior in the estimation of net rents for agriculture, the more accurate the resulting land values will be. Calculating average net rents should therefore involve a familiarization with the types of land uses and agricultural systems present in a given study area and an understanding of the economic benefits these provide to agents of land-cover change.

Rent calculation also involves recognizing the dynamic nature of linked ecological-economic systems and the associated consequences for trying to predict changes into the future. Changes in markets, economic or environmental policy, and human preferences or culture all have the potential to significantly change incentives for various land uses. Modeled land values will reflect current realities only to the extent that current conditions are similar to those that were in effect when modeled habitat conversion was occurring. Significant changes in variables such as the road network, human population, crop prices, and productivity will necessitate a revision of land-value estimates by rerunning models based on up-to-date patterns of habitat conversion and drivers along with current agricultural economic conditions. On the other hand, the models are flexible enough to allow simulation of changes in land values as, for example, the road network increases or habitat continues to be lost.

Although we believe the model has broad applicability in situations in which farmers are maximizing their net economic welfare, we also recognize that it will not be appropriate in all contexts. When profit maximization is a secondary concern to factors such as minimizing year-to-year risk, adhering to cultural norms, or claiming territory through land conversion, our theoretical framework may not be suitable. In these situations, quantifying patterns and probabilities of land-use change through analysis of satellite imagery is still a very useful activity, but calculating economic rents associated with particular land uses becomes an extremely difficult task. Alternate approaches such as individual surveys of landowners that assess reasons for forest conversion and the associated benefits may be more useful under these circumstances.

Extensions and Implications for Conservation

Integrating economic costs within algorithms of the reserve design literature could lead to greater applicability to real-world problems. In contrast to many approaches that require extremely detailed biological information, basic cost estimates such as those we described here are relatively easy to obtain and have significant real-world value in that they translate the economic costs involved in potential conservation plans in a way that is universally understood (i.e., through their monetary value). Including such costs up front in conservation planning should streamline the design process, allowing decision makers the ability to immediately understand the costs of various proposals rather than having to estimate such values in a post hoc, separate process.

One of the most exciting applications of mapping opportunity costs is the potential for using it in tandem with maps of the benefits that unconverted natural habitat provide. Cost-benefit analyses are a familiar tool for assessing development projects, and recent papers in the environmental sciences have compared the economic costs and benefits of conservation in a variety of settings (e.g., Norton-Griffiths & Southey 1995; Balmford et al. 2002; Muriithi & Kenyon 2002; Sinden 2004; Naidoo & Adamowicz 2005). Spatial cost-benefit analyses of environmental conservation at a landscape or regional scale have yet to appear in the published literature, however (but see Bateman et al. 2003 for a similar example). Mapping costs and benefits of conservation and highlighting areas where economic benefits exceed costs could be a valuable priority-setting method and is a language that policy makers are familiar with and can readily assimilate into decision-making structures. It could also play a strong role in highlighting beneficial situations for both conservation and development, which is especially important given the current donor climate that favors the latter and not the former (Sanderson 2005).

Nonetheless, spatial cost-benefit analyses of ecosystem goods and services remain difficult for two reasons. First, although many nonmarket valuation exercises show that the economic values of ecosystem goods and services can be high, they are usually not "capturable" at a scale relevant to local agents of land-cover change, as in the above framework. In other words, although the global or regional value of ecosystem goods and services can be significant, often no mechanisms exist for local agents of land-cover change to benefit from this value, so it does not enter into their decision-making process. Second, the state of the art in nonmarket valuation and mapping of ecosystem goods and services is still developing (Kremen 2005) and needs further investigation to allow modeling of economic values at spatial scales that are fine enough to be used in spatially explicit landscape models of habitat conversion. These two issues point to important policy and research agendas for the conservation community.

We have developed a framework to model opportunity costs of conservation in transitional agricultural landscapes, yet in principle the method could be applied to any region where alternative land uses are well defined and their net rents are calculable (subject to the caveats mentioned above). For example, in a landscape dominated exclusively by forestry activities, the opportunity costs of conservation would be defined as the value of land for forestry, and their calculation would involve determining which spatial factors influence where logging occurs, along with the economic benefits that arise from such activities. Similar arguments hold for other anthropogenic activities such as oil and gas exploration, mining, or any combination of land-use activities. The key concepts are the factors that influence patterns of habitat conversion and the net rents of land uses to which habitat is converted.

We believe the approach we develop here is another step in the ongoing integration of disciplines toward the goals of conservation of biodiversity and sustainable development (di Castri 2000; Shogren et al. 2003). We and others (Moore et al. 2004) see important gains that can be obtained by incorporating economic information into the field of conservation planning. These benefits have already been demonstrated (Ando et al. 1998; Polasky et al. 2001), but their assimilation has not yet blossomed in the way that other aspects of reserve design evolution have. Although some of this is undoubtedly due to the general uneasiness of treading across scientific disciplines, we hope we have shown here that including economic costs into conservation planning can be done in a manner that is not overly demanding of transdisciplinary expertise.

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