# PATTERNS AND PROCESSES OF LAND COVER CHANGE: UNDERSTANDING TRADEOFFS AMONG ECOSYSTEM SERVICES

By

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To my family

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PATTERNS & PROCESSES OF LAND COVER CHANGE: UNDERSTANDING TRADEOFFS AMONG ECOSYSTEM SERVICES

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Patterns and processes of land use and land cover change are among the most critical causes and consequences of modern-day global environmental change. Anthropogenic land use simplifies ecosystem structure and function and so, understanding tradeoffs in ecosystem services begins with characterizing landscape dynamics. Using case studies from Mexico and Costa Rica I identify spatial typologies of land cover change, relate these patterns to driving forces, and identify institutional approaches that complement ecosystem service provision. In Chapter One, I first examine the legacy of a land-use system indigenous to southern Mexico on the vegetation structure of tropical dry forest and then characterize changing spatial landscape patterns. Chapter Two identifies the driving forces of wetland conversion in northwest Costa Rica. Through the development of an empirical, spatially-explicit conversion model, I classify important spatial typologies of wetland loss. Recognizing the spatial patterns of conversion within the broader economic and biophysical context is critical for evaluating tradeoffs and designing spatial-targeting criteria for environmental policies. In Chapter Three, I analyze PES policy and implementation in Costa Rica in order to suggest design nuances, including the incorporation of greater spatial nuance to enhance landscape-level ecosystem service provision. In Chapter Four, I synthesize landscape dynamics in a conceptual model that highlights the

tradeoffs in landscape-level ecosystem services and the role of conservation institutions in resulting land cover dynamics. In conclusion, land use patterns considered in both space and time represent an emergent expression of emphasized mix of ecosystem services, whether intended or unintended. Understanding landscape dynamics is essential in designing institutions to manage tradeoffs and encourage synergies necessary for society's well-being.

# CHAPTER 1
# LAND USE AND THE PROVISION OF ECOSYSTEM SERVICES

Land use and land cover change are among the most critical causes and consequences of modern-day global environmental change (Foley et al. 2005). This is illustrated by examples across many geographic scales. Land cover change accounts for roughly 20% of global greenhouse gas emissions (Baumert et al. 2005) and 35% of $CO_2$ emissions, the most important greenhouse gas (Houghton & Hackler 2001). Conversion of "natural" habitat to direct human land use is the greatest driver of species extirpation (Wilson 1988, Dirzo and Raven 2003). At the local scale, direct human land use is found to reduce ambient biodiversity levels below even what today's unprecedented extinction rates would predict (Polasky 2005). Interactions between these environmental-change processes may reinforce or mitigate their respective negative consequences. For example, climate change resulting from rising greenhouse gas concentrations further exacerbates net losses in biodiversity. In contrast, landscapes experiencing forest transitions (i.e., significant forest regrowth) mitigate climate change through carbon sequestration associated with forest regeneration. Secondary forests may also better facilitate the processes that generate and maintain biodiversity, relative to other land uses.

Land cover designations indicate generalized ecosystem properties and a particular range and nature of ecosystem functions. The mass ratio hypothesis offers a specific ecological mechanism linking generalized land cover and ecological function. The hypothesis holds that trait values (e.g., vegetation height, specific leaf area, etc.) of the dominant contributors to plant biomass in an ecosystem largely determine ecosystem functioning (Loreau et al. 2001). Diaz et al. (2007) found that combinations of trait values and abiotic factors effectively explain ecosystem functions in a subalpine grassland system. Ecosystem functioning, in turn, either

directly or indirectly characterizes ecosystem services, or the goods and services derived from ecosystem processes that benefit society.

Land use decision-making and resulting land cover changes entail important tradeoffs in the provision of ecosystem services. The dominant contemporary anthropogenic land use patterns tend to simplify ecosystem structure and function, emphasizing production-oriented services like crop cultivation (Foley et al. 2005). The efficiency achieved through simplification of ecosystem functions has clear advantages. Food production at the global scale, for example, has kept pace with population growth; indeed, surplus agricultural yields constitute the foundation upon which elaborate commerce has developed. Nonetheless, as society proceeds "domesticating" the biosphere (see Kareiva et al. 2007), tradeoffs are increasingly clear. At the local scale, a vast suite of ecosystem services—from infectious disease mediation to water quality regulation—are foregone or drastically diminished (Rodriguez et al. 2005) through industrial, monoculture agriculture or other land uses that appreciably simplify ecosystem processes.

## The role of landscape pattern

Land use patterns develop in relation to physical gradients in the landscape. Huston (2005) holds that spatiotemporal patterns of land use are linked to net primary productivity in different ways as economic development progresses through agricultural, industrial and then information phases. Each phase is characterized by a unique pattern of resource dependence. Thus land use may be an emergent expression of the value that society collectively places on particular ecosystem services at a given point in time. More than representing the result of land use tradeoffs, the spatial pattern of land use and land cover also plays a critical role in the provision of ecosystem services. The arrangement of land cover patches determines the nature

and degree of interaction among and between biological and physical elements of the landscape. For example, Ricketts et al. (2004) found that coffee plantations within roughly one kilometer of forest land cover benefited from wild pollinator species. Yields increased by 20% and the frequency of small, misshapen beans decreased by 27%.

The geographic pattern of land use also confers certain path dependencies that constrain future landscape dynamics while simultaneously generating an aperture of opportunities. For example, drainage of wetlands for agriculture eliminates the ecosystem service of storm-water runoff storage but enhances the food production service. Even after agricultural abandonment, the wetland ecosystem and services it provided may not return depending upon the level of hydrologic and geomorphic disruption involved in the drainage. If the land remains dry, afforestation may ensue, illustrating how previous land use redirects future ecosystem functioning and services.

The spatial pattern of habitat in a landscape may function as biological insurance in the sense that migration and dispersal across ecosystems plays a key role in maintaining local biodiversity in heterogenous, mixed-use landscapes (Loreau et al. 2003). Thoughtful landscape planning can maximize this insurance function, along with other ecosystem services, for a given sum area of ecosystem or land-cover type present in the landscape, even while ensuring economically-productive land uses. Polasky et al. (2005) found an L-shaped and relatively-flat efficiency frontier between aggregate economic returns (x-axis) and defined conservation goals (y-axis) when iteratively modeling simulated land-use patterns. These results indicate that many different landscape patterns achieve intermediate-to-high levels of both objectives but that maximizing either one results in precipitous losses of the other. Eliminating externalities—that

is, charging for ecosystem service degradation and rewarding ecosystem service provision—further diminishes the perceived tradeoffs between environmental and economic outcomes.

**The role of institutions**

In reality, theoretically-ideal landscape patterns are attainable through simulation, but are unlikely to arise spontaneously or even by design in complex landscapes with myriad ownership arrangements and, often, countervailing economic incentives. Institutions, through policies, management practices and social norms, may mediate land use systems, influence ecosystem dynamics and drive land cover trajectories (Harrison 1991, Chomitz and Gray 1996, Southworth and Tucker 2001, Foster et al. 2003). Diminishing tradeoffs between the provision of ecosystem services and classic economic production functions hinges on appropriate, dynamic institutions.

Protected area establishment comprises a cornerstone strategy of conservation institutions and parks have been established in many landscapes. Indeed, protected areas cover over 11% of the earth's surface (Chepe et al. 2005). The expense, both financially and socially, of wholesale land purchases associated with traditional protected area establishment limits the degree to which parks may influence overall landscape composition and pattern. Park locations are often chosen by narrow criteria, like species representation. Alternatively, parks are established by default on low-productivity lands such as rock and ice, places that are unlikely to be exploited even without protection (Hazen and Anthamatten 2004). The current global network of protected areas may contribute to uneven protection of critical ecosystem services if reserve selection proceeds according to current strategies (Pyke 2007).

Given the limited function of protected area establishment, along with the importance of landscape pattern in the provision of ecosystem services, institutions that influence private property land-use choices should prove useful for shaping landscape patterns in ways that

enhance ecosystem service provision. One such strategy is payments for ecosystem services (PES) which attempts to eliminate externalities associated with targeted ecosystem services. PES is a nimble instrument that may be configured to pay a determined amount directly to land owners who provide a particular ecosystem service or suite of services for a prescribed duration. PES contracts may be geographically-targeted at properties important for achieving particular ecosystem elements, meeting some land cover area threshold and/or which are integral components of some broad-scale "beneficial" landscape pattern. An important research theme is to determine in what contexts this shift in land tenure and economic incentive structures prescribed by PES is appropriate.

Understanding tradeoffs in ecosystem services begins with characterizing landscape dynamics, identifying spatial typologies of land cover change, relating these patterns to driving forces, and identifying institutional approaches that complement ecosystem service provision. I address all of these themes in the following chapters of this dissertation. In the first, I examine the legacy of a land-use system indigenous to Mexico on tropical dry forest structure and changing spatial landscape patterns. To what extent does past land use affect ecosystem properties directly related to the provision of forest ecosystem services? In the second chapter I identify the driving forces of wetland conversion and, through a spatially-explicit conversion model, I classify important spatial typologies of wetland loss. Recognizing the spatial patterns of conversion within the broader economic and biophysical context is a critical for evaluating tradeoffs and designing spatial-targeting criteria for environmental policies. In chapter three, I analyze PES policy and implementation in Costa Rica in order to suggest design enhancements, including the incorporation of greater spatial nuance. In the fourth chapter, I synthesize landscape dynamics in a conceptual model that highlights the tradeoffs in landscape-level

ecosystem services and the role of conservation institutions in emergent land cover dynamics. Finally, in the concluding chapter I synthesize what pattern analysis, models of landscape change, a decade of empirical PES trends and an integrated understanding of the role of conservation institutions in landscape dynamics has to say about tradeoffs in ecosystem services.

# CHAPTER 2*
# MILPA IMPRINT ON THE TROPICAL DRY FOREST LANDSCAPE IN YUCATAN, MEXICO: REMOTE SENSING & FIELD MEASUREMENTS OF EDGE VEGETATION


## Summary

The Yucatan Peninsula hosts part of Central America's largest remaining tract of tropical dry forest and has been identified as a region of critical landscape change. This study complements the extensive research on land cover conversion in the region by investigating a subtle but important aspect of forest modification. We examine changes in the spatial characteristics of *milpa* cultivation plots in the swidden landscape of Peto municipality in Yucatan state from 1988 to 2003 using remote sensing. We also test the hypothesis that *milpa* clearings create a discernible edge effect in terms of forest structure. Results indicate that spatial patterns of *milpas* have changed over time. The amount of *milpa*/forest linear interface increased over the study period. Both satellite-based vegetation indices and field-based canopy cover measurements indicated that forest buffering *milpa* clearings had significantly lower biomass than background forest, despite that the background forest is itself a mosaic of successional forest stages. In contrast, there was no difference in stand basal area for milpa edge forest and background forest. Multivariate models demonstrated that the *milpa* edge indicator was the most important variable in explaining differences of vegetation indices for milpa edges and background forest compared with other factors that create edges in the landscape. Models were relatively effective in explaining mean values of vegetation indices; but they performed poorly in terms of explaining measures of forest vegetation heterogeneity. Comparing model results from each date suggests that the importance of milpa edges decreases over time, possibly as a function


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of the accumulated land use history as *milpas* rotate through the forest matrix. Evidence supports the notion that the effects of milpa land use extend beyond the clearing itself and into adjacent forest.

## Introduction

Human land use shapes ecosystem structure and function at multiple scales of time and space (Turner et al.. 1995). Among the most significant global challenges in the next century relates to management of the transformation of the earth's surface occurring through changes in land use and land cover (Mustard et al. 2004). Much land change science research for tropical regions focuses on deforestation using discrete land cover classifications to study wholesale conversion, like deforestation (Defries et al. 2000). Land cover modifications, such as forest degradation without wholesale clearing, also merit attention. Such modifications may have drastic effects on ecosystem processes and services like species composition and richness (Ferguson et al. 2003), trophic pathways (Bunn et al. 1999), carbon sequestration (Lawrence and Foster 2004), and land-surface energy balance (Southworth 2004).

The tropical dry forest life zone supports much of the world's agriculture, having more productive soils than those of humid tropical forests (Murphy and Lugo 1986). The dry forest life zone covers 42% of all land area in tropical latitudes. Despite this geographic predominance, fewer studies have been conducted on this forest type relative to moist and wet tropical forests (Ramankutty et al. 2006, Pérez-Salicrup et al. 2004). The Yucatan peninsula of southeastern Mexico hosts part of the largest remaining expanse of seasonally-dry tropical forest in Mesoamerica and has been identified as a "hot spot" for tropical deforestation, in part related to government-sponsored agrarian settlement programs (Chowdhury and Turner 2006).

Agriculture is the major driver of land cover change in tropical regions (Lambin et al. 2001). Swidden agriculture, in particular, comprises a major land use and an important resource-

management system in many parts of the tropics (Coomes et al. 2000). An extensive ethnographic literature classifies and describes these cultivation systems throughout the world (Unruh 1990, Banerjee 1995, Teran and Rasmussen 1995). *Milpa* is the traditional form of recurrent swidden in Mesoamerica. It is based on rotation of maize fields and fallows, during which secondary forests are established to replenish organic matter and nutrients. *Milpa* cultivation in the Yucatan Peninsula generally occurs in conjunction with communal "*ejido*" tenure. This coupled land-use and land-tenure system has sustained cultivation on poor soils by regulating the number and timing of *milpa* fields; the amount of ejidal forest land (through deliberate set-asides); and the effects of population growth on land resources through generally non-divisible inheritance rights (Plaza 2000).

In 1992, however, an amendment to Article 27 of the Mexican constitution provided a multi-step process through which *ejido* members may elect to privatize their long inalienable *ejidal* land. This amendment occurred as part of a national agenda to create an institutional framework favoring private investment, the development of a land market and productivity gains in agriculture (Johnson 2001). Other efforts aimed at "modernizing" Mexican agriculture include extension programs like PROCAMPO, PRONASOL, and other economic incentives that promote sedentary, intensive cultivation (Chowdhury and Turner 2006).

Of all forested land in Mexico, 85% occurs in *ejidos*, making the country unique among both developed and developing nations (White and Martin 2002). In Yucatan state, most *ejidos* are dominated by *milpa* land use. At broad spatial scales in the Yucatan, the precipitation gradient is the most important variable driving vegetation patterns (Lawrence and Vester 2004). At local scales, like within an individual *ejido*, however, the structure and function of Yucatecan forests—from litter production to biomass and soil properties—are more strongly influenced by

forest age and history of cultivation than by environmental gradients (Turner et al. 2004). The process of clearing land for swidden cultivation and the subsequent regeneration of forest during fallow periods creates a landscape mosaic of active *milpas*, forest patches in various stages of succession, and edge forest at the linear border between the two.

Many biophysical, structural and floristic changes occur as a result of *milpa*/forest adjacency and due to the use of fire in creating the *milpa* clearing (Eaton and Lawrence 2006). These changes may reinforce, or be reinforced by, land use feedbacks. One such feedback is that after clearing vegetation to make a field, forest at the interface immediately experiences an increase in both photosynthetically active radiation (PAR) and wind penetration at the ground level; altered albedo; and changes in surface energy and water balances. Subsequent land use in the edge forest may include collection or harvesting of fuel wood and cultivation of fruit trees (Ochoa-Gaona 2001). Both these biophysical and land-use effects reinforce longer-term changes in soil properties, floristic composition, and plant and animal dispersal (Figure 2-1).

Remote sensing and geographic information science have become standard tools for addressing these complex human-environment interactions at the landscape-level. Remotely sensed vegetation indices have also proven useful for coarse-scale biodiversity assessment, complementing field-based surveys (Nagendra 2001). With such great emphasis on deforestation in the land change science arena, only recently have researchers begun to address more subtle issues of re-growth/succession and other qualitative forest changes and feedbacks (Chowdhury et al. 2004, Moran 2004, Rudel et al.. 2005). Most such studies still rely on the use of discrete land-cover classifications; yet a more effective approach is to use the full suite of continuous data available from satellite imagery. NDVI (normalized differenced vegetation index) has long been used as a proxy for biomass (Jensen 1996) and to study structural forest

attributes like canopy architecture (Eamus 2001). Similarly, thermal band data have been useful in discriminating successional stages of forest (Southworth 2004) given that surface energy balance is linked to the character of land cover and thus past land use.

With continuous data, both land-cover conversions *and* within-class modifications are detectable. Not only can conversion from 'agriculture' to 'successional forest' be seen after field abandonment, but also within-class changes. The latter includes processes such as changes in forest density, forest degradation or the ability to identify a greater number of successional stages as a forest matures (Southworth et al. 2004, Bonan et al. 2003, DeFries et al.. 2000). Illumination of within-class changes greatly enhances the ability to foresee potential wholesale conversions before they occur, a key issue for biodiversity conservation and policy monitoring.

The critical link between forest cover and communal land tenure in the region underscores the importance of analyzing landscape patterns and forest attributes since the amendment of the constitution. A spatially explicit, landscape-level analysis may facilitate a better understanding of the effects of broader neo-liberal policy shifts by addressing the spatial pattern of recurrent clearings, or the landscape context that influences forest/clearing adjacency within the mosaic of successional forest stages comprising swidden-dominated land use systems. In this research we examine whether the spatial characteristics of *milpa* clearings have changed over time (1988-2003) using remote sensing. We refer to the patterns or characteristics of forest vegetation as structure. With both image and field-based data we test the hypothesis that *milpa* clearings create detectable edge effects compared to the background forest matrix (itself a mosaic of successional stages) in terms of several measurements of forest structure. With multivariate models, we also examine how this swidden-induced land cover juxtaposition compares with other factors that create linear edges and affect forest vegetation characteristics.

## Study Region

The study site lies in the interior, central portion of the Yucatan Peninsula of southeastern Mexico, within the state of Yucatan (**Figure 2-2**). The landscape is composed of areas of mechanized, monoculture cultivation; large regions of forest within which swidden agriculture occurs (mostly in *ejidos*); and built areas of roads, communities, and other infrastructure. The region has a marked seasonality, with most precipitation occurring between May and October. A precipitation gradient exists across the peninsula, increasing toward the south. Because of the prominent karst topography of the region, the peninsula has virtually no surface hydrography. Most of the Yucatan Peninsula falls within the tropical dry forest life zone (Holdridge 1971). This includes subdeciduous tropical forest, deciduous tropical forest and scrub areas.

Within this region, we selected the municipality of Peto (3296 $km^2$), in the southern, central region of Yucatan state, as a subset for further analysis (Figure 2-2). Rural areas of the municipality are characterized by *milpa* agriculture within a forest matrix. *Milpa* farmers generally reside in communities and commute to their nearby *milpa*s daily.

## Methods

### Field-Based Vegetation Structure Measurements

We conducted fieldwork in March of 2004 and 2005, late in the dry season. The point-quarter method, a variable plot size technique (Bell and Dilworth 2002), was employed at randomly-selected *milpa* edge and forest transects to calculate stand basal area and estimate canopy cover (n = 19). Each 60 m transect consisted of three plots (n = 57), where we measured the diameter at breast height (dbh) of the nearest tree ($\geq$ 2.0 cm) to the center of the plot for four quadrants, along with each tree's distance from the plot center. We also estimated canopy cover at each plot. We conducted semi-structured interviews with *milpa* farmers in an opportunistic

fashion during field work to gain a better appreciation of the social dimensions of swidden cultivation and of land use history of key areas within the study site.

**Image Processing and Geographic Information Systems Procedures**

Three Landsat TM images covering the study area were obtained through the Working Forests in the Tropics (WFT) Program at the University of Florida (NSF-IGERT). All image processing was carried out using ERDAS Imagine 8.7. Dates selected were 1988, 1994, and 2003, with all imagery from the end of the dry season. The 1988 image represents the landscape prior to the ejido privatization amendment, whereas the 1994 and 2003 dates represent the landscape shortly after and roughly one decade after the amendment, respectively. The 2003 image was used as our base map after georeferencing it to regional topographic maps (1:50,000), obtaining a root mean squared error (RMSE) of less than half of a pixel (< 15 m). The remaining images were then geometrically registered to the base image through image-to-image registration (RMSE < 15 m). Final positional accuracy of geocorrected images was validated in the field with the aid of a handheld GPS (positional error < 5.5 m). Images were calibrated, to correct for sensor gain, atmospheric distortion, and differences due to non-anniversary image dates (Green et al.. 2005). Band 6, thermal emission (low gain) was also calibrated and converted to black body temperatures (BBTemp) in Kelvin.

We subset all images to Peto municipality, masking out urban areas, industrial agriculture, and commercially- developed roadside so that only forest cover and clearings remained. Unsupervised classifications were performed for all images to identify *milpa* clearings, pasture clearings and forest (Figure 2-3b). Sixty-four training samples from fieldwork in March of 2004 were used for accuracy assessment (overall accuracy 88% and Kappa 0.79). Two vegetation indices were calculated for each year for the area corresponding to the forest in the classifications. The first was the normalized difference vegetation index (NDVI),

$$\frac{\text{Band 4 (IR)} - \text{Band 3 (R)}}{\text{Band 4 (NIR)} + \text{Band 3 (R)}} \qquad (2\text{-}1)$$

and the second was a thermal-band ratio,

$$\frac{\text{Band 6 (thermal)}}{\text{Band 5 (MIR)} + \text{Band 4 (red)}} \qquad (2\text{-}2).$$

*Milpa* clearings that resulted from image classification did not yield sufficiently precise, linear edges for the purposes of this analysis. To surmount this challenge, texture analysis was employed as follows. Principal Components Analysis (PCA) was performed for each image date on all bands minus the thermal band[1]. Texture analysis was performed on the first three resulting PCs for each image date using a 3 x 3 pixel window. Texture was calculated as

$$\frac{\Sigma[\Sigma(x_{c\lambda} - x_{ij\lambda})^2]^{1/2}}{(n - 1)} \qquad (2\text{-}3)$$

where $x_{ij\lambda}$ is the PC score for pixel (i,j), $x_{c\lambda}$ is the PC score for center pixel of window (kernel), and $n$ is number of pixels in a window.

The texture images clearly and precisely delineated the edges of clearings in the forest (Figure 2-3c). Thresholding of high texture values, compared with the relatively low texture values of the background forest, was used to create an image of raster cells corresponding only to the edges of clearings (Figure 2-3d). These raster "edges" were vectorized to make polygons out of all clearings in the forest. These forest clearing polygons for each year were overlaid on their

[1] Due to their coarser spatial resolution, thermal bands were less useful in identifying *milpas*. For Landsat 7, resolution on the thermal band is 60 x 60 m, but on Landsat 5 it is 120 x 120 m.

respective unsupervised classifications and subsets of the polygons were created responding only to *milpa* clearings (i.e., excluding pasture clearings).

From the *milpa* polygon vector files for each date, we filtered patches finer than the level of interest for this research ($\leq$ 0.09 ha) and calculated *milpa* area, perimeter and average density for each image date. A spatially-distributed random subset of 50% of the *milpa* polygons was selected for each year ($n_{milpa}$= 527, $n_{milpa}$= 796, and $n_{milpa}$ = 819 for 1988, 1994 and 2003 respectively). This ensured sufficient sample size for multivariate analysis while reducing the problems associated with spatial autocorrelation. The latter inflates the risk of type I error in statistical analyses (Hoeting et al.. 2006). Each *milpa* polygon was buffered, creating a zone corresponding to two pixels, equal to the length of the 60 m field transects. Overlapping buffer areas for closely-spaced *milpa*s were subtracted so that overlap was eliminated from the analysis. The mean and variance were calculated for the NDVI and thermal band ratio within the area corresponding to the *milpa* buffer polygons. Only forest pixels were included in the vegetation index computations so that if a given *milpa* was located along a road or other land cover type, only the adjacent forest pixels for that *milpa* were included. Further, any effect from shadow that may affect the dependent variables was averaged out by using ratios in the vegetation indices. These vegetation index summary statistics for *milpa* buffers were extracted from the GIS and exported to a database (illustrated in Figure 2-4).

To sample the remotely-sensed vegetation indices for the background forest matrix while controlling for the size and shape of the sample units (i.e., *n* uniquely-shaped *milpa* buffers), random points were generated within the background forest and the coordinates of those random points were assigned to *milpa* buffer polygons. This randomly re-positioned the *milpa* buffer sample units over the background forest matrix (illustrated in Figure 2-4). Buffer overlap was

again subtracted to exclude any such overlap and indices were extracted as done for the *milpa* buffer regions ($n_{forest}$= 519, $n_{forest}$= 776, and $n_{forest}$= 796 for 1988, 1994, and 2003 respectively). Finally, distance from the sample centroid to the nearest road and the nearest forest clearing (be it *milpa* or pasture) was calculated for all samples.

**Statistical Analyses**

To determine whether spatial landscape characteristics of the *milpa*/forest mosaic have changed over time, we compared combined *milpa* patch metrics (*milpa* area, *milpa* perimeter and the perimeter-to-area ratio) across three dates using MANOVA with follow-up ANOVAs for comparisons of individual metrics across the three dates. We tested for differences in means of stand basal area and canopy cover between *milpa*-edge sites and forest sites using Mann Whitney tests with Bonferoni corrections for multiple comparisons.

The means and variances of both vegetation indices were compared for *milpa* edges and the background/control forest for each image date using MANCOVAs. *Milpa*-edge indicator (*milpa* buffer zone or forest site) was the binary, independent variable of interest. Covariates in the model were distance to nearest road and distance to nearest forest clearing. The effect size for independent variables was assessed via partial eta-squared ($\eta_p^2$) values which indicate the percent of remaining variance explained by a given independent variable after accounting for the effects of other independent variables in the model (i.e.,, $SS_{effect}/[SS_{effect} + SS_{error}]$) (Norusis 1990). Finally, ANCOVAs were used to test for a difference of means for individual vegetation indices (*milpa*-edge versus control forest sites) for each time step. ANOVAs were used when covariates were not significant for a particular index.

Appropriate descriptive statistics were calculated for all variables prior to each analysis, including checking for correlations among dependent variables, between dependents and factor

levels (where applicable), and between covariates. Assumptions for each statistical test or model were tested appropriately. All analyses were performed on standardized variables using SPSS 11.5.

## Results

### Milpa Spatial Characteristics

MANOVA revealed that significant differences of means for the combined *milpa* patch metrics exist across the three dates of the analysis (Wilks' $\lambda$: 0.836; $F_{(6, 8562)} = 133.51$; P=0.000). The number of *milpa*s increased over time from 1054 in 1988 to 1639 in 2003 (Table 2-1). Comparing individual metrics in pair-wise fashion for the three dates showed no significant difference in mean *milpa* size ($p > 0.45$ for all cases), with the average *milpa* being slightly less than 1.5 ha in area. In contrast, *milpa* perimeter was different for each date ($p < 0.00$ for all cases), increasing from 583.95 m to 729.11 m from 1988 to 1994 but decreasing to 641.48 m by 2003. Mean perimeter to area ratio was the same for 1988 and 1994 at just less than 0.06 $m/m^2$ ($p = 0.633$), while the ratio was 0.061 $m/m^2$ in 2003 ($p<0.000$). Though statistically significant, this difference is miniscule on the ground.

### Milpa Edge Effects

Mann Whitney results (Table 2-2) comparing field measurements of vegetation structure for *milpa*-edge and control sites indicate no difference in stand basal area for the *milpa*-edge (x = 31.61 $m^2$/ha) than the control (28.3 $m^2$/ha)($p = 0.137$). Canopy cover is greater on average for the control (37.0%) compared with the *milpa*-edge sites (7.1%) ($p < 0.00$).

The three separate MANCOVAs (Table 2-3) reveal significant effects of all covariates and the *milpa* indicator for each date (for all independent variables, $p = 0.000$) and that significant differences exist among combined vegetation for *milpa* and control sites for each date (for all models, Wilks' $\lambda < 0.67$; $P = 0.000$). The partial eta-squared ($\eta_p^2$) values indicate that,

after accounting for the effects of the two covariates, the *milpa* indicator explains nearly half of the remaining variance in the combined vegetation indices in 1988 ($\eta_p^2 = 0.470$). For 1994 and 2003, the *milpa* indicator respectively explained roughly one-third ($\eta_p^2 = 0.300$) and one-fifth ($\eta_p^2 = 0.193$) of the variance remaining after accounting for the effects of proximity to the nearest road and nearest forest clearing.

Not all covariates were found to be significant for all vegetation indices across the three years of the study. ANCOVAs (or ANOVA where applicable), performed separately for each time step, revealed significant effects for the *milpa* indicator for all four vegetation indices, for all years ($p < 0.001$ for all cases) (Table 2-4). For 1988, 1994 and 2003, means for both NDVI and thermal band ratio indices were greater for control sites relative to forest at *milpa* edges ($p = 0.000$). The magnitude of the difference between *milpa* edge and control sites decreases over time, however (Table 2-4). In terms of the indices' variance for all years, *milpa* edge sites proved to be more heterogeneous than control sites ($p = 0.000$) with one exception. The exception is the thermal band ratio index variance for 1994 where the control site was significantly more heterogeneous than the *milpa* edge site ($p = 0.000$).

Across all years of the study, mean values of both NDVI and thermal band ratio indices are better predicted than the variance of the indices as evidenced by consistently higher $R^2$ values (Table 2-4). Except for the thermal band ratio index mean for 1994, the simple models with only three predictor variables explained nearly or over half of the variance in mean vegetation index values. In contrast, vegetation heterogeneity for both indices across all years was poorly explained with the highest adjusted $R^2$ value at 0.343 for 1988 (NDVI).

## Discussion

Swidden agriculture, or regions characterized by this land use, have primarily been examined in terms of the way that cultivation decreases the *amount* of forest cover (Myers 1993, Lawrence et al.. 1998, Vance and Geoghegan 2004). Of the research that has focused on the full, episodic swidden-fallow cycle (e.g., Walker 1999), none has examined edge effects of the agricultural clearings. Rather, focus has been on the space corresponding to the confines of cleared patches themselves. Ecological edge research has rested predominantly in the domain of conservation literature, generally examined as a result of longer-term, relatively "permanent" fragmentation processes (Bierregaard et al.. 1992), not as part of an episodic land use pattern (but see Ochoa-Gaona 2001). Forest age, cultivation history and management (land use, owner preferences) are the major drivers of the rates and dynamics of ecosystem functioning (Eaton and Lawrence 2006). Here we have complemented the growing literature on land cover modification with a subtle but important aspect of swidden land use effects and feedbacks. Results supported the hypothesis that milpa clearings create discernible ecological edge effects.

The *milpa* indicator variable is consistently the most important variable in explaining variance of the combined vegetation indices across all dates (Table 2-3). Coupled with the finding that forest in the *milpa* buffers was both consistently lower in biomass and more heterogeneous (as indicated by NDVI and thermal ratio indices), this confirms that at this scale of observation, *milpa* land use creates a discernible edge effect in the forest. Forest areas buffering *milpas* should favor edge species, managed species and habitat generalists at the expense of mature forest species and understory plant communities. Relative to other factors that create linear land cover juxtapositions (i.e., roads and the next-nearest clearing), local *milpa* edges clearly have the dominant impact on local vegetation structure, according to the relative partial eta-squared values (Table 2-3). That is, if the more distant edges created by the covariates

(other clearings in the forest or roads) affected broader spatial patterns of forest structure, the effect sizes for these variables would have been greater. This result conforms with other findings of scale-dependency in edge effects related to the ecological process in question (Baldi and Kisbenedek 1999, Huhta et al.. 1998). Nonetheless, when these local effects of *milpa* edges on forest structure are multiplied across the Yucatan landscape for thousands of *milpas*, this local land use/cover juxtaposition is clearly important in structuring broader regional vegetation characteristics. The spatially-radiating effects of *milpa* clearings should be considered in examining forest regeneration in swidden cycles, as opposed to focusing only on clearings themselves.

Examining the vegetation indices separately across *milpa* and forest sites, however, there is an appreciable discrepancy in the efficacy of the *milpa* indicator and covariates to predict mean vegetation heterogeneity relative to mean biomass, as indicated by the adjusted $R^2$ values (Table 2-4). Clearly, there are substantial drivers of spatial patterns of vegetation heterogeneity that were not considered here given the relatively low $R^2$ values (other than biophysical gradients which average out by virtue of the sampling scheme). Since all organisms and ecosystem processes respond to environmental variability, and *not* to the arbitrary concept of mean values, exploring which factors do effectively explain vegetation heterogeneity represents an important area for future research.

Lawrence and Foster (2004) found that after twenty-five years of forest regeneration in this region, woody basal area had recovered to 63% of total mature forest levels and that recovery of total live biomass took even longer. Comparing our stand basal area results with those of Read and Lawrence (2003) suggests that, on average, the successional forest matrix of ejidal land in Peto is at least ten years of age. Given that the cultivation history of their study

area is not as long as that of Peto and the fact that Peto is drier than the southern region of the Yucatan Peninsual, the forest may be on average even greater than ten years in age. Since we found no significant difference between stand basal area for *milpa* edge forest and control forest, this suggests either that none of the ejidal background ("control") forest was sufficiently mature to differ from *milpa* edge forest; or that in terms of this particular measure of vegetation structure, edges have no consequence on woody biomass in a matrix of secondary, successional forest. In contrast, vegetation biomass was greater for background forest than for forest at *milpa* edges as indicated by both canopy cover and the two remotely-sensed indices. The latter are particularly suited to indicate the photosynthetic capacity of vegetation. Thus, taking all three forms of forest structure measurement together, results may indicate that edge effects have greater impact in terms of green, leafy biomass than woody biomass. Leaf litter contributes to the accumulation of organic matter in the soil, as well as nitrogen concentrations (Lawrence and Foster 2002). Thus, perhaps after some lag in time, even within a matrix of successional forest as in this landscape, edges may feedback to influence woody basal area through changes in nutrient cycling.

The total number of *milpa*s has increased within Peto municipality over the three dates supporting the notion that efforts to "modernize" the swidden landscape may potentially weaken the communal land tenure institution (Klepeis and Vance 2003), which has traditionally limited the number of fields cultivated through non-divisible inheritance rights to ejidal land. While none of the farmers interviewed had privatized their land, several alluded to an existing informal land market and expressed interest in eventually pursuing the multi-step privatization process if their *ejido* elects; further south on the pensinsula, Abizaid and Coomes (2004) found evidence of changes in land use made in anticipation of privitization. Increasing numbers of land users

and/or decreased fallow times appear to "unintentionally" accompany efforts to liberalize the region's agriculture (Klepeis and Vance 2003). This trend suggests a loss of agronomic services that may constrain future agricultural productivity without substantial chemical inputs, given that key aspects of forest productivity decrease significantly after two clearing cycles (Lawrence and Foster 2004), let alone promotion of permanent cultivation in order to meet the government-set goal of a surplus maize (Chowdhury and Turner 2006).

Compared with reported values in the literature for mean *milpa* area (Klepeis et al.. 2004, Vogeler 1976, Vogeler 1970), field size in Peto is smaller than average (roughly 1.5 ha in Peto compared with 4 ha). Field size in Peto is, on average, smaller than those further south on the peninsula because there, agro-forestry ejidos created in the 1960s were quite large by comparison. This smaller size and the static mean *milpa* area across dates for our Peto study region also may reflect both biophysical and social constraints to the expansion of average field size. Firstly, the "cockpit karst" topography, particularly in the southwest region of the study area (Puuc Hills), affects site selection for agriculture, as it has in this landscape for millennia (Killion et al.. 1989). Field size is necessarily limited by the area of "*planos*" or flat land suitable for cultivation. Secondly, farmers indicated that they generally work their fields alone, such that labor may constrain the possible extent of *milpas*, a condition confounded by out-migration of youth toward the peninsula's urban areas and coastal fringes (Lutz et al.. 2000).

Results from the analysis of *milpa* patch metrics suggest subtle but important changes in the spatial characteristics of the *milpa*/forest landscape mosaic over the course of the study. The static mean of the ratio of perimeter to area (i.e., $[\Sigma\ {}^{milpa}{}_{P}/\ {}^{milpa}{}_{A}]/n$, a simple, patch-based measure of shape complexity) across the three dates contrasts with the landscape-level trend where the average amount of edge forest created per average field size (i.e.,, $[\Sigma\ {}^{milpa}{}_{P}]/n$ : [Σ

$^{milpa}A$]/n) has increased. Since mean field size is constant across dates, this indicates an increased variability of *milpa* perimeter. This underscores the importance of considering both patch-level and landscape-level means when analyzing spatial land-use patterns. Further research is needed to determine if this trend mirrors natural variability driven by biophysical contours (e.g. topography) in the landscape or whether it represents an increase in forest-edge juxtaposition driven by policy-related land-use choices and land-use path dependency, or another trend such as increased *milpa* activity along linear features like roads.

Lawrence and Foster (2004) found past land use to be the most important factor determining forest processes and characteristics, which may provide a reason why the importance of *milpa* edges in explaining vegetation structure has decreased over time (Table 2-3; decrease in $\eta_p^2$ across dates). That is, edge effects from *milpa* clearings may be more significant when land use intensity is relatively low at the landscape level when edges are created in a matrix of mature forest such that the contrast between edges and the forest matrix is great. Over time, as *milpa*s rotate through the landscape and the matrix is transformed to a mosaic of successional forest stages, ecological edges may not be as important in driving patterns of vegetation structure relative to past *milpa*-fallow cycles that occurred in a particular location (Eaton and Lawrence 2006). Future research which maps *milpa* land use cycles as they occur (in contrast to this retrospective remote-sensing approach) will help discern the dynamic relative influence of *milpa* edges in determining forest vegetation characteristics.

## Conclusions

In this region, past land use is a critical determinant of the rate and nature of forest regeneration, yet to date, most research has examined only forest clearings themselves. With this study, we have complemented existing research with a subtle but important aspect of *milpa* land

use effects and feedbacks by examining the impact of *milpa* clearings on the structure of surrounding forest. Results supported our hypothesis that the "footprint" of *milpa* land use is broader in geographic extent than the patch of cleared land itself with associated constraints in agricultural productivity potentially transferring to the forest region bordering each *milpa* site. The *milpa* edge indicator variable was the most important factor in explaining variance in the examined measures of forest structure, though its influence decreased over time and appears to also be spatially scale-dependent. While mean values of various measures of forest structure were well explained by the models, factors important for explaining vegetation heterogeneity were clearly omitted and likely include past land use. We found that the amount of edges between the forest matrix and *milpas* has increased over time. Thus, understanding the role of ecological edges within a swidden landscape will contribute to sensible policy formulation and management within the matrix of remaining successional forest.

**Acknowledgements**

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Table 2-1. Mean milpa patch metrics across three years of analysis.

| Patch Metric | 1988 | 1994 | 2003 | Significance |
|---|---|---|---|---|
| **N** | 1054 | 1593 | 1639 | n/a |
| **Area** (ha) | 1.38[a] (±1.21) | 1.41[a] (±1.02) | 1.41[a] (±1.27) | All $p \geq 0.45$ |
| **Perimenter** (m) | 583.95[a] (±323.20) | 729.11[b] (±400.17) | 641.48[c] (±398.77) | All $p < 0.00$ |
| **P/A** ($m/m^2$) | 0.057[a] (±0.02) | 0.058[a] (±0.01) | 0.061[b] (±0.02) | For a p=0.63<br>For b p<0.00 |

Table 2-2. Mean forest structure measurements for *milpa*-edge and control sites.

| Vegetation Attribute | | Mean | Std. Dev. | *p* |
|---|---|---|---|---|
| Stand Basal Area | *Milpa* edge | 25.66 | 28.25 | |
| | Control | 30.67 | 16.09 | 0.137 |
| Canopy Cover | *Milpa* edge | 7.1 | 4.49 | |
| | Control | 37.0 | 16.05 | 0.00 |

Table 2-3. Mancova results for three dates indicating the effect size ($\eta_p^2$) of each covariate and the *milpa* indicator, along with their significance.

| Year | Variable | Wilks' $\lambda$ | $\eta_p^2$ | ($df_1$, $df_2$) | F | p |
|---|---|---|---|---|---|---|
| 1988 | d. road | 0.924 | 0.077 | (4, 1039) | 21.4 | 0.000 |
| | d. clearing | 0.916 | 0.084 | (4, 1039) | 23.7 | 0.000 |
| | *milpa* indicator | 0.297 | 0.470 | (4, 1039) | 615.3 | 0.000 |
| 1994 | d. road | .949 | 0.069 | (4, 1565) | 20.90 | 0.000 |
| | d. clearing | .964 | 0.077 | (4, 1565) | 14.55 | 0.000 |
| | *milpa* indicator | .663 | 0.300 | (4, 1565) | 199.06 | 0.000 |
| 2003 | d. road | 0.927 | 0.042 | (4, 1608) | 31.87 | 0.000 |
| | d. clearing | 0.914 | 0.030 | (4, 1608) | 37.84 | 0.000 |
| | *milpa* indicator | 0.513 | 0.193 | (4, 1608) | 380.95 | 0.000 |

$\eta_p^2 = SS_{effect}/[SS_{effect} + SS_{error}]$ (Norusis 1990)

Table 2-4. ANCOVA/ANOVA results for the three dates in the analysis. Bonferroni-adjusted significance is indicated for the *milpa* indicator for each vegetation index.

| Year | Veg. Index | *Milpa* Edge | Control | df | F | P* | Adjusted $R^2$ |
|---|---|---|---|---|---|---|---|
| 1988 | NDVI x | 0.535 (±0.591) | 2.124 (±0.446) | 1 | 1569.91 | 0.000 | 0.711 |
| | BBT/R+MIRx | 0.983 (±0.329) | 2.109 (±0.485) | 1 | 1183.32 | 0.000 | 0.697 |
| | NDVI $s^2$ | 0.424 (±0.882) | -0.512 (±0.351) | 1 | 337.36 | 0.000 | 0.343 |
| | BBT/R+MIR $s^2$ | 1.204 (±1.386) | 0.052 (±1.211) | 1 | 156.10 | 0.000 | 0.167 |
| 1994 | NDVI x | -0.250 (±0.394) | 0.409 (±0.338) | 1 | 769.28 | 0.000 | 0.482 |
| | BBT/R+MIR x | -1.099 (±0.069) | -0.920 (±0.284) | 1 | 111.47 | 0.000 | 0.217 |
| | NDVI $s^2$ | -0.270 (±0.451) | -0.534 (±0.188) | 1 | 149.33 | 0.000 | 0.130 |
| | BBT/R+MIR $s^2$ | -0.391 (±0.849) | 0.318 (±1.130) | 1 | 86.22 | 0.000 | 0.139 |
| 2003 | NDVI x | -0.913 (±0.751) | 0.312 (±0.480) | 1 | 853.05 | 0.000 | 0.530 |
| | BBT/R+MIR x | 0.053 (0.192) | 0.469 (0.180) | 1 | 1127.58 | 0.000 | 0.632 |
| | NDVI $s^2$ | 0.635 (1.425) | -0.408 (0.563) | 1 | 207.81 | 0.000 | 0.200 |
| | BBT/R+MIR $s^2$ | -0.123 (0.346) | -0.502 (0.269) | 1 | 340.27 | 0.000 | 0.289 |

* Bonferroni adjustments for multiple comparisons made to significance interpretations (p must be < 0.006 for significance)

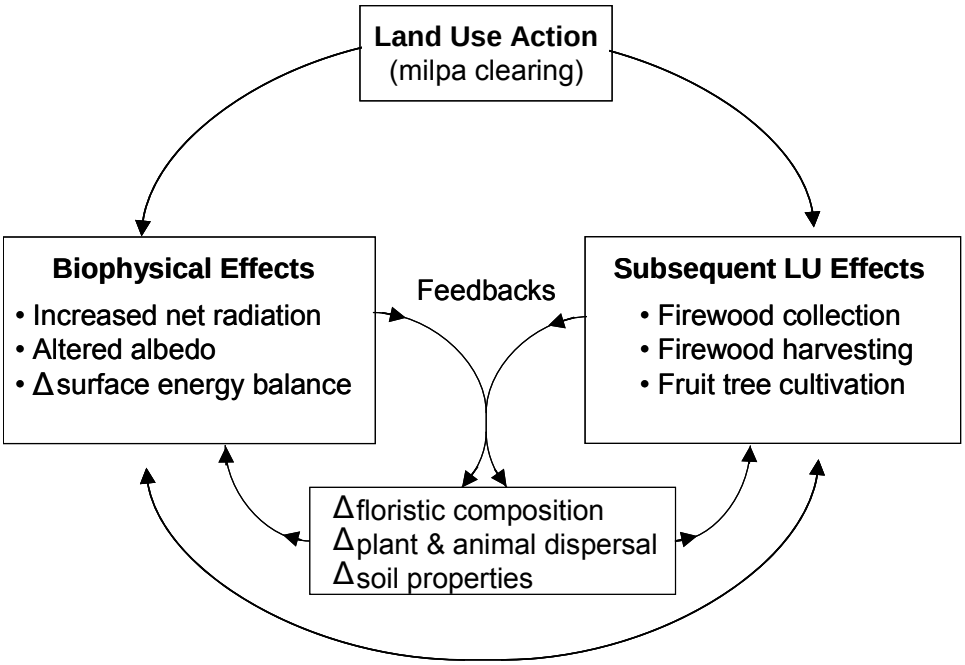


Figure 2-1. Interactions and feedbacks between land use (*milpa* agriculture) and land cover properties.

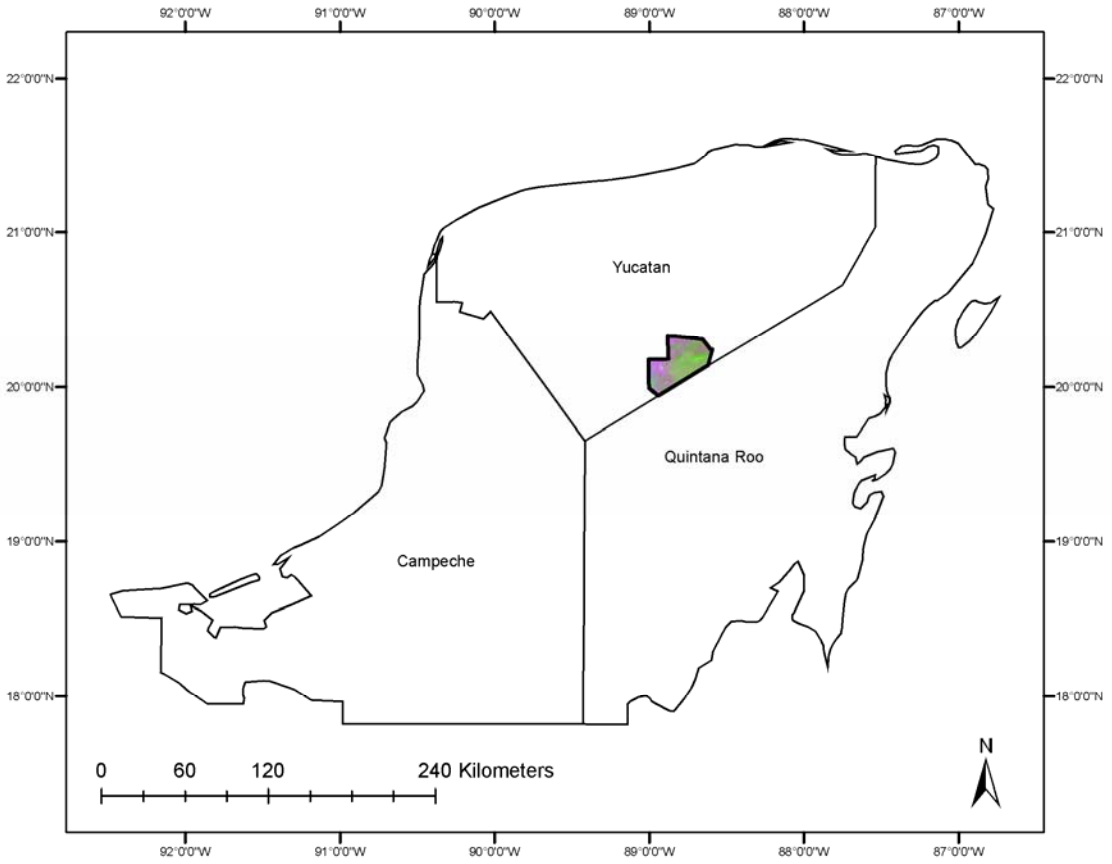


Figure 2-2. Map of Peto municipality study region in the southern, central region of Yucatan, Mexico.Peto study area is shown as a 5,4,3 (as R,G, B) composite subset from an April 1988 Landsat 5 Image.

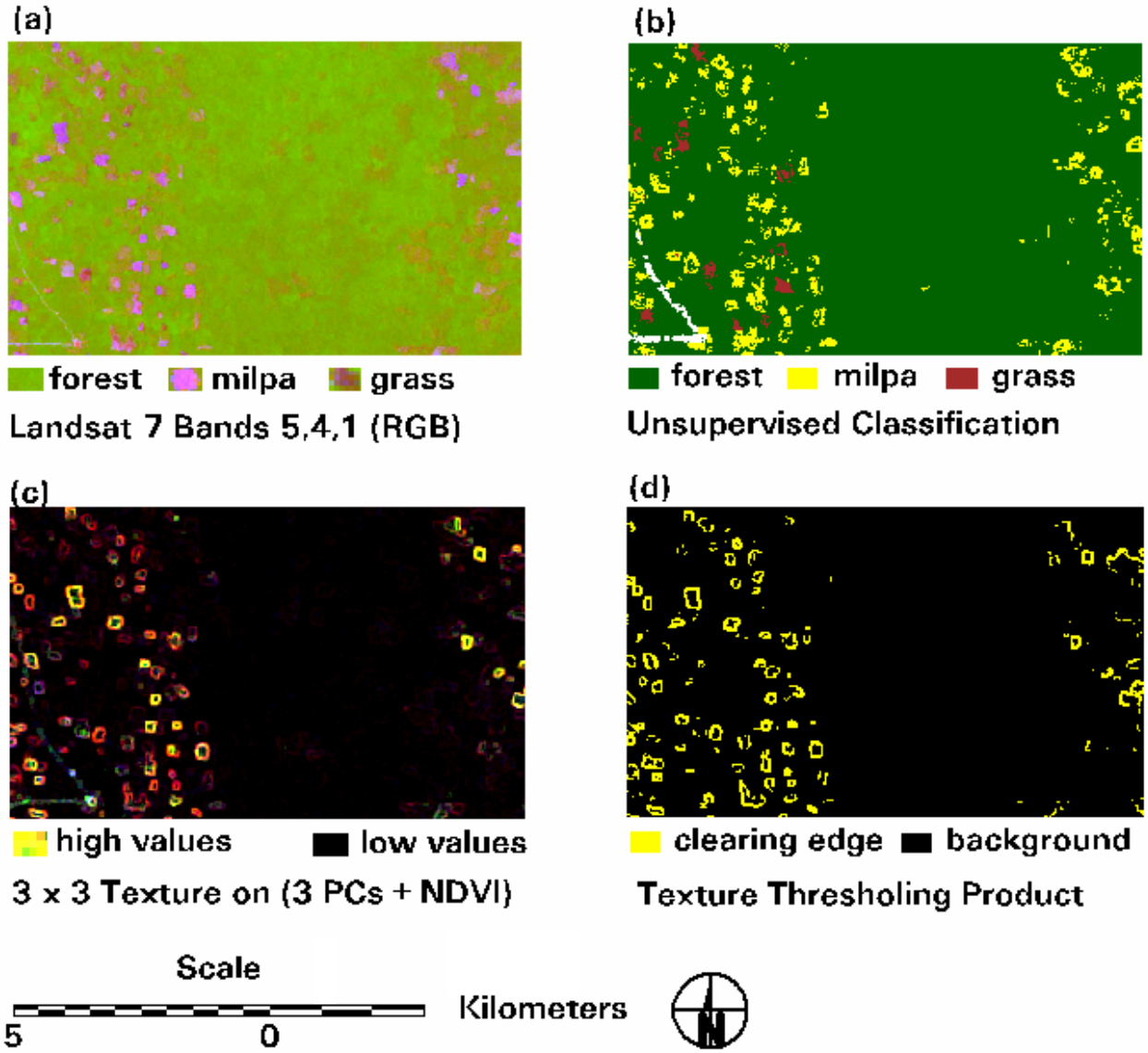


Figure 2-3. Examples of intermediate remote sensing products used to isolate clearing edges from 2003.

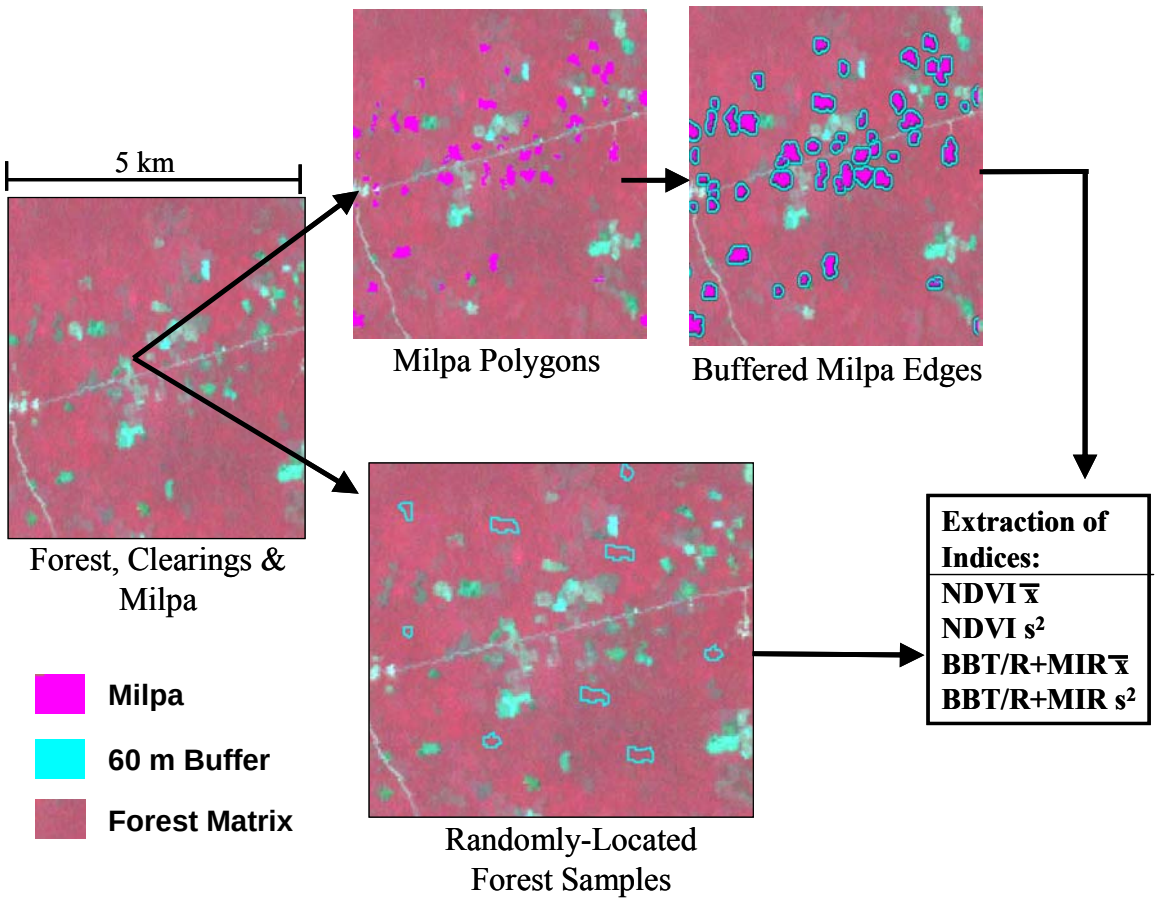


Figure 2-4. Flowchart illustrating the protocol for extracting vegetation indices for forest region buffering milpas (top) and background forest (bottom) for a small subset of Peto study region (RGB:4,3,2).

# CHAPTER 3*
# CONVERSION OR CONSERVATION? UNDERSTANDING WETLAND CHANGE IN NORTHWEST COSTA RICA


## Summary

Wetlands are more threatened than any other ecosystem type, with losses exceeding 50% of their original extent worldwide. Despite the small portion of the Earth's surface that they comprise, wetlands contribute significantly to global ecosystem services. In this study we tested the hypothesis that the location and rate of change in wetland amount in the Tempisque Basin of northwest Costa Rica is predictable from landscape setting. Our results demonstrate that a strong potential exists for developing predictive models of wetland conversion based on an understanding of wetland location and surrounding trends of land use. We found that topography was the single most important predictor of wetland conversion in this area, entraining other conversion processes, and that spatial patterns of wetland loss could consistently be predicted from landscape-level variables. Areas with highest probabilities of conversion were found in the most accessible, non-protected regions of the landscape. While Palo Verde National Park made a substantial contribution to wetland conservation, our results highlight the dependence of lower-lying protected areas on upland processes, adding a little-addressed dimension of complexity to the dialogue about protected area management. Conservation strategies aimed at reducing wetland loss in tropical habitats will benefit from careful analysis of the dominant land use system(s) at a relatively broad scale, and the subsequent development of management and policy responses that take into account dynamic opportunities and constraints in the landscape.


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## Introduction

Wetlands are ecologically diverse habitats that are integral to a range of ecosystem processes and provide many important ecosystem services. They are also under threat in many parts of the world from conversion to agricultural and residential uses. Some of the main scientific and practical challenges for wetland conservation include understanding the processes that lead to wetland conversion to other land uses; predicting where and when wetlands are most likely to be lost; and developing ways of reducing or mitigating negative anthropogenic impacts on wetlands. In this paper we test the hypothesis that wetland conversion is predictable from landscape location and develop a quantitative approach to predicting wetland losses. Our results suggest a range of management and policy actions that could reduce rates of wetland conversion in our study site.

A disproportionate percentage of threatened plants and animals are wetland obligates (Boylan and McLean 1997); although wetlands comprise less than 3% of the Earth's surface, they may contribute up to 40% of the globe's ecosystem services on an annual basis (Zedler and Kercher 2005). The many benefits provided by wetlands include flood protection (Hey and Philippi 1995), water quality enhancement (Jeng and Hong 2005), carbon storage—if managed appropriately (Mitra et al. 2005), and nutrient cycling, both internally and at the upland interface (Bunn et al. 1999). Wetlands also play a critical role in the land-surface energy balance (Meijerink et al. 2005), provide habitat for many species (Trebitz et al. 2005), and are important for hydrologic connectivity at the watershed scale and beyond (Pringle 2003).

The conversion of land with relatively natural vegetation patterns to more intensive, direct human use (e.g. monocrop agriculture) is a major cause of global declines in biodiversity in recent decades (Forester and Machlis 1996, White et al. 1997, Cincotta et al. 2000). Wetlands are more threatened than any other ecosystem type (Millennium Ecosystem Assessment 2005),

with wetland losses exceeding 50% of the original modern-era global extent (Mitsch 2005). Both the ecological contributions of wetlands and the many threats to their persistence are best understood from a landscape perspective (Mitsch and Gosselink 2000). Wetland processes (e.g., floodwater retention and mercury methylation) and character (e.g., riparian, or coastal) vary across space and time (Haig et al. 1998, Ogawa and Male 1986). Spatial and temporal context also influence anthropogenic activities in the landscape. Direct wetland loss often occurs in flatter areas through conversion of wetlands to other land uses, primarily agriculture and urban areas (Zedler and Kercher 2005). Indirect losses occur as a consequence of such things as climate change (Klein et al. 2005); water extraction (Lemly et al. 2000) and other forms of hydrologic alteration (Liu and Cameron 2001); adjacent and upstream changes in land use (Pringle 2003); and changes in populations of wetland-dependent wildlife (Haig et al. 1998, Semlitsch 2002). Tropical wetlands are particularly at risk from the increasing human demand for fresh water and land, dependence upon hydroelectric energy for development, inadequate wastewater treatment, and frequent inability to craft or apply appropriate protection measures when wetland sites have been neither documented nor inventoried (Junk 2002). All Central American nations are signatories of the 1971 Ramsar convention, which created a framework for national and international collaboration in conserving wetlands and promoting the sensible use of wetland resources. However, anthropogenic pressure on wetlands in the region is only expected to increase in the coming years (Ellison 2004, Junk 2002).

Land cover changes in many tropical areas have been characterized and quantified extensively over the last several decades (Lambin and Geist 2001), and spatially-explicit models of deforestation processes have been developed for many regions (Irwin and Geoghegan 2001). Changes in spatial patterns of wetland mosaics resulting from wetland loss have been quantified

for the northeastern U.S. (Gibbs 2000). The amount of wetland loss has been predicted for coastal Louisiana (Cowan and Turner 1988) and the U.S. Atlantic coastal plain (Koneff and Royle 2004). Yet there are few generalized wetland conversion prediction models akin to those for deforestation (but see Reyes et al. 2000).. Landscape conservation and planning efforts are increasingly recognizing the importance of geographic setting in defining and prioritizing wetland functions and in understanding how wetlands are influenced, both directly and indirectly, by the socio-economic drivers of landscape change (Haig et al. 1998). This is particularly true for tropical regions that lack long-term, spatially-explicit data and have been dominated by conservation efforts that focus predominantly on tropical forests.

Our primary hypothesis in undertaking this study was that the processes that drive wetland conversion are broad-scale and socio-economic in nature, meaning that wetland conversion should therefore be predictable from the location of wetlands along relevant gradients in the broader landscape. Alternative hypotheses include the suggestions that wetland conversion occurs randomly; that wetland conversion is driven primarily by local (small-scale) processes, and hence is not tightly aligned with broader gradients in the landscape; and that wetland conversion is driven by such a complex, multi-layered set of processes that the prediction of wetland loss from a landscape setting is essentially impossible. Our objectives in this study also included the development of a potentially generalizable modeling approach for predicting wetland conversion, and the formulation of management and policy recommendations for our study area, the Tempisque Basin of northwestern Costa Rica. The results suggest that although a diversity of processes contribute to wetland conversion, the single most important driver of wetland conversion in our study site is topography, which entrains a range of other landscape-level processes. Our results support the hypothesis that wetland conversion can be understood

through analysis of landscape context; provide a detailed example of one way in which the likelihood of wetland conversion in tropical landscapes can be predicted; and shed light on relevant management and policy concerns in our study region.

## Methods

### Study Region

The Tempisque River Watershed is a 5,404 $km^2$ basin in the province of Guanacaste, in northwestern Costa Rica (Figure 3-1). The Tempisque River originates in the northeastern area of the watershed in the Guanacaste Mountain Range, which runs from northwest to southeast and defines the eastern border of the Tempisque basin. The region experiences one wet and one dry season per year and has a mean temperature of 27.5$^{o}$C. The mean annual precipitation of 1817 mm falls mainly between May and November (Mateo-Varga 2001), though a break in the rainy season (*el veranillo* or little summer) corresponds roughly to the month of July. Annual and inter-annual variation in rainfall affects the character and extent of active wetlands at any given time. The Tempisque Basin's intricate mosaic of diverse wetlands and adjacent uplands, including wet meadows, open-water lagoons, flooded forests and riparian mangroves, provides critical habitat for resident and migratory water birds traveling along or over-wintering on the Central American isthmus (McCoy and Rodriguez 1994). Counts of more than 50,000 waterfowl have been made in the wetlands of Palo Verde National Park, including the endangered Jabiru stork (*Jabiru mycteria*).

The economy of the Tempisque region has long been based on extensive cattle ranching, and many of the formerly forested upland areas have been converted into cattle pasture (Peters 2001). Traditionally, the wetland areas of the lower basin were used for grazing during the dry season since they remained green much longer into the dry months than upland pastures. The crash of the beef market in the 1980s, the rise of tourism, international lender-forced structural

adjustments, and the implementation of a region-wide hydroelectric/irrigation project have contributed to the transformation of land use in the Tempisque Watershed from an extensive, cattle-ranching system to a more diverse and more intensive agrarian system (Daniels 2004). With the implementation of irrigation (the Arenal-Tempisque Irrigation Project or PRAT as it is known by its Spanish acronym), commercial, crop-based production has become more important than cattle ranching in the lower basin. Depending on the season, PRAT injects a volume of 50-85 $m^3/s$ of water into the terrestrial landscape year-round. Flooded rice, sugar cane, and melon cultivation are increasingly prominent features of the lower watershed and often occur at the expense of wetlands. Ensuing alterations in hydrologic connectivity, although not measured directly in this study, also contribute to changes in wetland locations and configurations.

Roughly a third of the 18,760 ha Palo Verde National Park (situated in the lower Tempisque watershed) is wetland. The park and adjacent, non-protected wetlands were declared as Ramsar sites in 1991 (Mateo-Varga 2001). Wetlands and all surface waters are held in the public domain by the Costa Rican government and fall under a special category of protection. In practice, however, defining and delineating wetlands has long challenged the Ministry of Environment since most wetland definitions are suitable for temperate latitudes but not necessarily applicable in tropical systems (Ellison 2004). For this research, "protected" wetlands were considered to be only those that fell within the bounds of the national park.

**Overview of Methods and Modeling**

We derived binary land cover maps from three Landsat images by identifiying wetland and non-wetland classes at the pixel level. The land cover history of each pixel was traced over the three dates and compiled into a single data layer of wetland "trajectories." This dichotomous trajectory map represented whether each pixel had been "conserved" or "converted" by the final date in the image series. We then modeled this binary response variable with multiple logistic

regression models, using orthogonal predictors obtained via Principlal Components Analysis (PCA), to explore the relevance of different variables in driving landscape change.

**Land Cover Data and Predictor Variables**

We used the Ramsar definition of wetlands[1] in this research, with the interpretation that flooded rice fields were *not* considered wetlands for the purposes of this analysis. This is also the interpretation used by the Costa Rican government. Binary land cover maps (wetland/non-wetland) were produced from a three-date, early dry-season Landsat image classification series (1975, 1987 and 2000) using a rule-based classification technique (path 16, row 53 for all dates; also path 17, row 53 for 1975 since watershed was split across multiple scenes by WRS1). The dates of the images capture "baseline" land cover (1975) prior to the establishment of the National Park or the implementation of the irrigation program, an image date after their implementation (2000), and one mid-point image (1987).

Radiometric data were converted from radiance to surface reflectance (Jensen 2000) by correcting for sensor gain, atmospheric distortion, and differences caused by non-anniversary dates. Geometric correction for the 2000 image was performed with sixty ground control points taken with a handheld GPS. The root mean squared error (RMSE) error achieved from a first order geometric transformation was 0.4880 pixels, or less than 15 m. The other images were then co-registered to the 2000, also achieving an RMSE less than 0.5. Pixel size for the 1975 MSS image mosaic was resampled to 30 x 30 m to match spatial resolution from later image dates.

[1] Wetlands are areas of marsh, fen, peatland or water whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters.

Since statistical clustering of spectral data alone has often proved ineffective for accurately classifying wetlands (Ozesmi and Bauer 2002), we employed a rule-based procedure that incorporated domain knowledge, spatial relationships, and two distinct algorithms on spectral data (for normal and non-normal distributions of land cover classes) to improve the accuracy of the resulting land cover maps (Figure 3-2). A descriptive example of a rule used is as follows:

> Classify as wetland all pixels identified as wetland by the parallelepiped algorithm if they occur at $\leq$ 60m elevation and do not fall within ground-mapped polygons of agricultural fields where crops are cultivated.

This particular rule helped compensate for the spectral confusion of wetlands and rice fields by exploiting location information mapped during fieldwork. Rules were applied systematically across dates so as not to bias classification or the modeling of trajectories, adjusting for differences in spatial resolution and inter-annual climate variation accordingly. Accuracy was assessed separately for each of the three dates using over 90 independent reference points for each. For 2000, reference data were collected via field training samples, while for 1987 and 1975, they were derived from higher resolution aerial photo-mosaics (1:35,000 and 1:20,000, respectively). While the data-collection and processing time for this classification method were costly compared with traditional remote sensing on spectral reflectance data alone, the results justify the expenditure. Accuracy of wetland classification exceeded 90% across all dates (class-wise Kappa statistics were also > 0.90). For more details on this rule-based classification, see Daniels (2006).

We traced the wetland status (wetland or non-wetland) of each pixel across the three classification dates and compiled the result into a single data layer of dichotomous wetland "trajectories" representing whether each pixel was "conserved" or "converted" by 2000 (Table 3-1). To simplify the modeling process, we considered only one-way conversions *from* wetland

land cover *to* other land cover types, excluding the potential for the reverse trend. The same simplification is typically employed for spatially-explicit deforestation modeling (Mertens and Lambin 2000). The potential for misclassification of dry versus converted wetlands was reduced by a number of steps and checks, specifically (1) careful standardization and recalibration of images to compensate for differences in prior rainfall (Daniels 2006); (2) inclusion of a mid-point image to provide another reference point, while increasing the probability of unidirectional change and reducing error propagation across trajectories; (3) testing for a relationship between wetland size and conversion probability (this was weak); and (4) checking for spatial pattern in pixels that were apparently converted back to wetland (this was lacking).

In a basin-wide descriptive analysis of land cover change, Daniels (2004) found that the major drivers of wetland conversion in the Tempisque Basin were related to agricultural intensification in the lower watershed and alterations in hydrologic connectivity and hydroperiods. To assess the relative importance of drivers of wetland trajectories related to these processes, a number of predictor variables and interaction terms were considered for inclusion in the predictive model (Table 3-2). These data were compiled in a multi-layer, raster geographic information system (GIS) for the Tempisque Watershed with a spatial resolution of 30 $m^2$. Predictors included x/y coordinates, elevation, slope, a protected area indicator surface, wetland patch area, distance to nearest road, distance to the Panamerican Highway, distance to the nearest population center, and distance to the Tempisque River (features are mapped in Figure 3-3). The use of distance surfaces as predictors has several distinct advantages. First, it allows for a spatially explicit modeling framework and enhances our ability to understand the role of landscape setting in wetland conversion. Also, this approach allows for accurate representation of important but temporally dynamic variables (e.g. distance to nearest population center remains

the same while the population itself may change). The rationale for testing each variable and a host of interaction terms (Table 3-2) was based on a qualitative understanding of changing land use systems in the Tempique Basin.

**Statistical Model Building**

For each raster cell (i.e., sample unit) in the GIS, the dependent variable (wetland trajectory) and its corresponding values for each of the ten proposed predictor variables were extracted from the GIS to a database (N = 223,689). Fourteen interaction terms among predictors were computed and added to the suite of independent variables. We then generated random subsets for model building and independent model validation using the formula

$$n_{test}/n_{train} = [1 + (p\text{-}1)^{1/2}]^{-1},$$

where *p*, the number of predictor variables (Schaafsma and van Vark 1979). With *p* equal to four for the final model, this determined the ratio of training (n = 141,813) to testing (n = 81,876) cases in our split sample approach to model validation (Ozesmi et al. 2006). A rigorous exploratory analysis was performed to better discern relationships among potential predictor variables and complement our ground-based understanding of landscape dynamics (see Daniels 2004). Potential predictors with non-significant ($p>0.05$) or weak correlations ($rho<0.25$) were eliminated from further consideration. A series of models was developed by combining conceptual models of wetland conversion processes, findings from the exploratory analysis, and the principle of parsimony. We compared the Akaike Information Criterion (AIC) across models to determine the most effective suite of predictor variables. This is computed using each model's log likelihood ratio:

$$AIC = -2(\text{log likelihood ratio}) + 2k$$

where *k* represents the number of model parameters estimated. AIC indicates the loss of information incurred for incorrect model specification (Hoeting et al. 2006)..

For the best models, we transformed the suite of landscape (predictor) variables along orthogonal axes via principal components analysis (PCA) to eliminate multicollinearity, which can substantially confuse interpretation of model results (Legendre & Legendre 1998). Varimax rotation was employed to increase interpretability of resulting components by enhancing the distinction between variables that significantly load on each component and those that do not, while maintaining the cumulative variance of the structure matrix (McGarigal et al. 2000). PCA has been widely used with multiple regressions in spatial ecological analyses for many years. For recent examples from several different contexts, see Chang et al. (2006); Summerville et al. (2006); Krueger et al. (2006); and Corbett & Anderson (2006).

In ecological modeling there are well-documented tradeoffs between prediction accuracy, reliable coefficient estimation, understanding driving mechanisms and ecological interpretation (Graham 2003). One of the central weaknesses of PCA is that components are often uninterpretable. The more variables that are included, the less interpretable the model becomes and the more likely it is to produce statistically significant results that are ecologically meaningless. For this reason, we performed an initial round of variable selection for model building as described above prior to PCA. Our goal was to not only make accurate predictions, but to understand dynamics in this wetland landscape through interpretation of the model.

Two predictors were withheld from the PCAs and added separately to the models: protected area status (falling within PVNP or not) and wetland patch size. The other variables group naturally along major biophysical and social landscape gradients; and given the importance of protected areas in the conservation of natural habitat (Sanchez-Azofeifa et al. 2003) and the role of patch size in landscape-level ecological processes (Flather and Bevers 2002), we wanted to estimate model coefficients for these critical predictors directly. Finally,

variance inflation factors (VIF) indicated that multicollinearity among these two variables and the principal components used as predictors was not an issue (VIF < 1.2 and $p < 0.001$). Three prediction models with slightly different combinations of predictor variables were compared using the AIC. Jackknifing (n = 141,813 leave-one-out iterations; see Efron and Tibshirani 1993 for discussion of jacknifing approach) indicated that the model was stable and all independent variables were unbiased to less than 0.001 for all variables.

**Model Validation and Performance Assessment**

Model validation was conducted with independent dataset (n = 81,876) that were set aside prior to building the logistic regression model, as described above. The logit, predicted probability of conversion and predicted trajectory (conservation or conversion) were obtained by evaluating the final prediction equation for each independent observation. Nagelkerke's $R^2$ values along with area under the curve (AUC) for receiver operating characteristic (ROC) plots were used to assess model prediction performance. Nagelkerke's $R^2$ ranges from zero to one; it is considered a "pseudo- $R^2$," however, in that the percent of variance explained for a dichotomous dependent variable depends on its frequency distribution (Nagelkerke 1991). The AUC provides a threshold-independent measure of model performance (Cumming 2000) by assessing model function independent of the probability cut-off used for defining the conversion trajectory. This is done by balancing the opposing goals of model sensitivity (the probability of predicting conversion when, in fact, conversion has occurred), with model specificity (the probability of predicting conservation when, in fact, conservation has occurred). AUC is calculated by a calculus-based trapezoidal rule and ranges between 0 and 1. An effective model would demonstrate a rapid rise in sensitivity (y-axis) with little increase in the probability of producing a false positive (x-axis), having an AUC that approaches 1. By contrast, a random model would display a curve of a diagonal line between the origin and (1,1), with an AUC of 0.5.

When evaluating the model to display its prediction results in geographic space, we selected the threshold probability corresponding to the highest sensitivity and lowest complement of specificity (i.e., one minus specificity, the probability of falsely predicting conversion). This was 0.6, meaning that any pixel with a probability of conversion greater than 60% was mapped as a conversion.

Model coefficients from the logistic regression equation, based on orthogonal variables, were interpreted to determine the relative importance of independent variables and landscape gradients in predicting wetland conversion. We evaluated the model on a pixel-by-pixel basis in order to create a spatially-explicit map of wetland conversion probability. A thorough exploration of mapped results was performed digitally to identify salient spatial trends of wetland conversion probabilities. We also mapped residuals and predicted trajectories from the model, along with error rates, to better discern any spatial typologies of wetland conversion and to assess the validity of the model in geographic space (Rogerson 2001).

## Results

### Model Selection

The predictor variables for the best models included PC1, PC2 and PC3, tenure status and, for one of the models, patch size (see **Appendix A** for details). AIC and $R^2$ values for the model that included patch size were better (AIC = 119,819 and $R^2$= 0.48) than for the model where patch size was excluded (AIC = 125,403 and $R^2$= 0.44) (Table 3-5). Residuals became less variable and model fit improved as patch size increased. In contrast, however, AUC was slightly higher when patch size was excluded (0.81 versus 0.79); and the model coefficient for patch size (-0.00) demonstrated that virtually no relationship existed with the dependent variable (Table 3-5). The observed trend for the residuals appeared related to the fact that the number of

observations decreased as patch size increased (Cumming 2000). After taking these issues into account, wetland patch size was excluded as a predictor in the final model.

**Principal Components Analysis**

The final composite predictor variables were comprised of these variables: y coordinate, elevation, slope, distance to river, distance to nearest road, distance to population center and distance to the Panamerican Highway. PCA with varimax rotation reduced the dimensionality of these seven predictors into three orthogonal components with eigenvalues greater than one, which explained over 73% of the original variance (Table 3-3). Communalities were relatively high, except for distance-to-nearest-road (0.53), confirming that PCA effectively reduced dimensionality while explaining most of the variance in the original variables (Table 3-4).

The rotated structure matrix (Table 3-4) shows that distance to nearest road (0.70) and distance to population center (0.90) loaded highly onto the first component. This component represented the socioeconomic infrastructure gradient of the Tempisque landscape (PC1). Positive signs for both of these loadings indicated that large distances from towns were associated with the same end of the component as large distances from roads, since population centers in the Tempisque Basin are connected through a transportation network. The y-coordinate also loaded substantially, but negatively, on this component (-0.52) meaning that wetland sites become increasingly isolated (less socio-economically connected) moving downstream (i.e., further south) in the basin. On the second component (PC2), the y coordinate, distance to river, and distance to the Panamerican Highway loaded highly. The loading for y coordinate was negative (-0.78) while loadings for distance to the Panamerican (0.89) and river (0.57) were positive. The distance from the highway and river increases moving south in the basin, reflecting the watershed's triangular, geomorphically-defined shape. This second component was called the north-south gradient in the landscape (PC2). Slope (0.79) and

elevation (0.86) loaded highly on the third component, representing the topographic gradient of the landscape (PC3).

**Spatial Predictions**

The probability of wetland conversion is mapped in Figure 3-4**a**. In general, the probability of conversion increased moving away from the central, lower watershed. Wetlands near the river's discharge point, within the protected area, had the lowest probabilities of conversion ($p < 0.20$) in the entire basin. Areas along the river channel in the lower watershed also had low probabilities of wetland conversion (around 0.25), even on the non-protected west bank of the river. At the outer reaches of the basin's wetland network, among the highest and driest wetlands in the landscape, pixels had consistently high probabilities of conversion ($p > 0.60$). For these sites, highest conversion probabilities tended to occur at the edges of larger wetland patches ($p > 0.85$), with probability decreasing toward the interior of such patches ($p < .60$). Localized non-protected wetland patch networks with high probabilities of conversion ($p > 0.80$) occured immediately adjacent to the national park within large, corporation farms that cultivate sugar cane, cantaloupe and rice. Areas within the park had significantly lower probabilities of conversion on average, yet there were three distinct regions in the park where wetlands had very high probabilities of conversion ($p > 0.80$).

**Model Performance**

The final model had an AUC of 0.81. This indicates that there was an 81% chance that the conversion probability for a randomly-selected observation belonging to the conversion trajectory group, would be greater than that of a randomly selected observation belonging to the conservation group (Fielding and Bell 1997). Figure 3-4**b** illustrates where sites of both conversion and conservation were correctly predicted, along with errors. False negative cells (i.e., prediction of wetland conservation when conversion actually occurred) were concentrated

in the lower, central watershed in and around PVNP. False positives (i.e., prediction of wetland conversion when the site was conserved) were clustered.

**Model Coefficients**

All predictor variables were statistically significant at $p<0.001$ (Table 3-5). The order of influence of landscape drivers on wetland loss demonstrated that the topographic gradient (PC3) was the most important correlate of wetland conversion, followed by socioeconomic connectivity (PC1), land tenure, and finally, north-south gradient (PC2). Model coefficients ($B_i$) are in log-odds units and so we interpreted the exponentiation of the coefficients ($e^B$). The magnitude of the model coefficient for PC3 was 3.10 (Table 3-5). As PC3 scores increased, degree of landscape relief—elevation and/or slope—increased. Every unit increase in the topographic gradient raised the likelihood of wetland conversion more than twenty-two times ($e^B$ =22.11). The coefficient of PC1 was -1.03 (Table 3-5). As PC1 scores increased, distance from built landscape features like population centers and roads also increased and y-coordinate values decreased (i.e., move southward). The negative sign on the PC1 coefficient indicated that the likelihood of conversion decreased as PC1 scores increased. For every unit increase of the socioeconomic infrastructure gradient (i.e., increased isolation), the likelihood of conversion was about one-third ($e^B$ = 0.36) as great.

Results also demonstrated that tenure status affected the probability of wetland conversion ($B$ = -0.91). The likelihood of conversion decreased in the protected area; in fact conversion was less than half as likely ($e^B$ = 0.40) inside the park as in the non-protected sites, controlling for other relevant drivers of conversion and variation in the landscape. The final predictor in the model was PC2 ($B$ = -0.82). As PC2 scores increased, y-coordinate values decreased. Since the north-south gradient's coefficient was negative, the likelihood of wetland conversion decreased moving south in the basin.

## Discussion

### Drivers of Wetland Change

The results provide clear support for the hypothesis that wetland conversion is predictable from landscape setting in the Tempisque Watershed. The importance of landscape setting in this instance provides valuable insights into the process of wetland conversion in Costa Rica. Landscape variables proved to be strong and significant correlates of changes in wetland amount and location. The relative order of influence of the drivers of conversion from highest to lowest was topography, socioeconomic connectivity, land tenure, and north-south gradient respectively. This order suggests a hierarchical structure of influences in which the physical landscape (topography) may provide the ultimate context for landscape processes that are more proximate in nature, driving conversion at finer spatial and temporal scales. In other words, topography defines higher, dryer wetland areas that are more likely to be settled and converted to human land use relative to lower-lying, boggy areas. The physical landscape consequently provides a constraining influence—in a hierarchical context—on socioeconomic drivers of conversion (PC1). This result makes sense both for economic efficiency and security of investment in agricultural production. It may also indicate increased likelihood of *indirect* conversion related to urbanization such as altered hydroperiods, changes in surface runoff patterns, and changes in water quality. As might be expected, isolated areas in the lower basin had a lower probability of conversion than sites with greater economic connectedness or proximity to infrastructure like towns and roads.

Although tenure affected wetland conversion probability, given that there are no occupants or direct agricultural threats inside the park, the fact that conversion was no less than half as likely to occur inside the park underscores the critical, indirect role of intra- and inter-annual hydrologic alterations and cumulative watershed effects. While converted wetlands inside the

park are not precluded from being wetlands again in the future, the model illustrates sites that are presumably most sensitive to changes in hydrologic connectivity. The north-south gradient (PC2), may reflect the underlying influence of geomorphic configuration, thereby supporting the idea that shape and directional gradients in a watershed influence the spatial organization of natural and anthropogenic processes. These, in turn, affect location of wetlands and threats of their conversion (Huston 2005).

**Assessment of Model**

Our model of wetland conversion provides a relatively strong fit to the data (AUC = 0.81) by comparison to other published models. Pontius and Schneider (2001) used two distinct methods for predicting deforestation in a Massachusetts watershed and obtained AUC values of 0.65 and 0.70, respectively. Other spatially-explicit models of land cover conversion report accuracies ranging from sixty-five to 90% (Brown et al. 2002, Pijanowski et al. 2002, Mas et al. 2004). The nature of focal land cover transitions and differences between studies in modeling techniques and performance measures make the utility of cross-model comparisons questionable. Nonetheless, our results fall within the upper range of conversion predictions for upland habitats. Given that ours may be the first spatially-explicit attempt to model wetland conversion at the landscape scale in the tropics, future models should be able to improve upon this performance.

The spatial pattern of prediction errors points out the weaknesses of the modeling approach that we employed (Figure 3-4**b**). False negatives (i.e., failure to predict conversion when it actually happened) were concentrated in the lower, central watershed in and around PVNP. These conversions were largely due to alterations in the hydrologic regime as little mechanized agriculture or infrastructure development has taken place in these locations of the basin, indicating the model's weakness in predicting indirect wetland conversion related to

hydrologic alterations. This shortcoming was not surprising since predictions were made using landscape variables rather than spatially-explicit hydrologic data.

The clustered spatial pattern for false positives (i.e., prediction of wetland conversion when the site was actually conserved) suggests that the watershed-scale model failed to predict correctly in places where finer-scale processes might have controlled the pattern of conversion. For example, in the central region of the northern border of the park, flooding has been associated with land use conversion to agriculture and poor irrigation drainage. In response, officials identified a buffer zone for which special management provisions are hoped to prevent land use conversion and maintain the quality of wetlands *inside* the park. Urban (2005) points out that ecologists tend to model fine-scale processes that have somewhat limited utility for studies that seek to scale their results up to entire landscapes. In contrast, our prediction model performed well for the overall landscape-level process of conversion but clearly missed some of the finer scale processes occurring locally within the watershed. Perhaps this is related to a spatial filtering or averaging effect of using a predefined grain of analysis (i.e., the 30 m pixel of Landsat imagery).

The low probability of conversion of both protected and non-protected wetlands near the basin's discharge point is probably related to the fact that these areas are increasingly prone to flooding, as a consequence of upstream canalization and channel-widening activities. Much of the discharge region is largely inaccessible by land, suggesting that the significant investment required for conversion to direct human land use has facilitated persistence of even non-protected wetlands in these isolated, seasonally-boggy and flood-prone areas. Though conversion probabilities inside the park were low on average, conversion of upstream patches has the potential to further alter hydrologic connectivity, hydroperiod, and water quality for

wetlands inside the park. Of the three distinct regions inside the park with high probabilities of conversion (>0.80), one occured adjacent to a portion of the river that was channelized in the mid-1980s. This is consistent with findings in many other watershed studies where riparian areas are degraded or converted by diversion and channelization (e.g., Toner and Keddy 1997, Tiner 2005). A second such region occured around an historically-maintained canal used to bring freshwater from the river into the lagoons of the park when it was a cattle ranch. In recent history, the connectivity of lagoons in PVNP and the river decreased as the channel was filled by sediment deposition (Jimenez et al. 2003).

**Conservation Implications**

The spatially-explicit map of conversion probabilities (Figure 3-4a) suggests several patterns that have relevance to conservation planning. First, remaining wetlands in proximity to agriculture and urban land uses are at extremely high risk of conversion. Prevention of direct conversion will require active protection and management. Even then, such wetlands may be at risk of degradation and/or indirect conversion from cumulative watershed impacts (Vorosmarty and Sahagian 2000). Secondly, active maintenance of a buffer region around PVNP may serve to protect wetlands within the park from indirect conversion. As emphasized in Pringle (2001), the park's location at the bottom of the watershed implies that successful conservation of its wetlands will entail an integrated, watershed-level approach to managing land-use, nutrient runoff and hydrologic flows.

Finally, the typology of wetland conversion in the non-protected areas of the Tempisque landscape appears to be one of "sweeping" conversion from higher/dryer, economically-connected regions to more isolated, boggy sites. The park provides a defensible and successful border against direct human land use, meaning that only indirect wetland conversion is possible within its bounds. The pattern of conversion inside PVNP is therefore more patchy and less

predictable in relation to landscape-level predictor variables than for surrounding areas. This difference in typologies is similar to results from deforestation analyses, where different spatial patterns of conversion develop depending upon the driving forces and their spatial organization in the landscape (Husson et al. 1995). Mertens and Lambin (1997) found that dividing landscapes into regions with uniform deforestation typologies enhanced model specificity. Developing a typology of wetland conversion processes may help to better predict wetland loss in the future. The development of such a typology, however, is confounded by the inherently patchy nature of wetland occurrence in the landscape. Whereas deforestation entails any of various patterns cutting into large blocks of forest, even "sweeping" wetland loss may appear patchy given that wetlands are dispersed and comprise a small proportion of the surface area of any region.

The spatial nature of this model aids in pointing out the need to carefully consider dominant land use systems in relation to opportunities and constraints presented by the physical landscape, which may change over time. Extensive cattle-ranching, the dominant economic land use in the Tempisque Basin up to the late 1970s, resulted in deforestation of uplands, occurring at a wide range of elevations and even on sloped lands. Agricultural intensification, in contrast, has in part facilitated reforestation of uplands at the expense of land use conversion in flat, low-lying areas (i.e., wetlands). The conservation mechanisms and policies that are most appropriate for the respective land use regimes are complementary, but would obviously differ in their strategy and budgetary foci (e.g. upland forest reserves and riparian buffer restoration versus wetland network preservation and monitoring of in-stream nutrient load). Our results support Huston's (2005) proposed theory of land use change, where spatio-temporal patterns of land use

are linked to net primary productivity in different ways depending upon the phase of economic development and its constituent patterns of resource dependence.

Though threats of direct human land use within PVNP are not an issue in this landscape, indirect conversion of wetlands (i.e., through altered hydrology) was found to represent a threat to protected wetlands; and one that was not as readily predicted by landscape variables. Our results agree with those of Cowan and Turner (1988) for Louisiana's coastal region in that indirect impacts on wetlands are often as important as direct conversions. This finding adds a dimension of complexity to the dialogue about protected area management that is little addressed (but see Pringle 2001). Most discussions on the efficacy of protected areas center around the degree to which park boundaries are defensible given natural resource dependence among people living in and around parks, particularly in the tropics (Brandon et al. 1998, Bruner et al. 2001, Terborgh 1999). The degree of "land-surface connectivity" between parks and their surrounding landscapes has been quantified and described in many case studies (Mas 2005, Ravan et al. 2005). Defries et al. (2005) analyzed 198 protected areas around the world and found a substantial increase in park isolation over the last twenty years.

Model results confirmed that while surface connectivity appears to be critical for landscape management and planning considerations, it is only one of many elements of ecological connectivity for which protected areas should be monitored and managed. Assessing and monitoring hydrologic connectivity and changes therein will require an understanding of how all land cover and resource use at the watershed level are connected on the surface (two dimensional), through hydrologic flows (three dimensional), and how connectivity varies over time (four dimensional). Clearly these interlinked processes, and the landscape position of protected areas in relation to them (Harris et al. 2005), must be considered carefully in the

context of integrated watershed management, non-linear pattern-process relationships, and critical thresholds in drivers and responses (Bedford and Preston 1988). While this approach considers only wetland conversion (as opposed to degradation), the results may be used to focus finer analyses of wetland conditions. Future work may also explore temporal aspects of wetland conversion in greater detail using a multinomial spatiotemporal model.

## Conclusions

Model results supported our hypothesis that wetland conversion is driven by broad-scale processes related to socio-economic trends of land use. Conversion was effectively predicted from wetland location along relevant social and biophysical landscape gradients. This study demonstrates that a strong potential exists for developing a generalizable approach to predicting wetland conversion. By quantitatively exploring and mapping wetland conversion probabilities, areas of the landscape in which different mechanisms of landscape change are operating can be identified and brought to the attention of managers and policy makers. While it is encouraging that Palo Verde National Park makes a substantial contribution to wetland conservation, this study also highlights the importance of landscape context and landscape influences on hydrologic connectivity, along with the dependence of lower-lying areas on their watersheds. Ultimately, wetland conservation and the maintenance of the many goods and services wetlands provide will have to encompass a regional perspective on ecosystem management and appropriate spatial planning within the mosaic of different land uses that influence wetland persistence.

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Table 3-1. Construction of wetland trajectories using three binary land cover maps (wetland/non-wetland). Each raster cell in the landscape fell into one of these two trajectories of conversion or conservation. MSS = Multispectral Sensor, TM = Thematic Mapper, and ETM = Enhanced Thematic Mapper.

| | Date (Landsat platform) | | | |
|---|---|---|---|---|
| | 1975 (MSS) | 1987 (TM) | 2000 (ETM) | Trajectory |
| Land Cover | Wetland | Wetland | Wetland | Conservation |
| | Wetland | Wetland | Non-wetland | Conversion |
| | Wetland | Non-wetland | Non-wetland | Conversion |

Table 3-2. Response variable (wetland trajectory) and proposed predictor variables and interaction terms, along with their theoretical justification, for wetland conversion model.

| Raster Layer | Data Source | Description | Justification |
|---|---|---|---|
| Wetland Trajectory | This analysis | Cells classified as wetlands in 1975 were extracted from land cover classifications for 1975, 1987 and 2000. These cells were coded for either the conservation or conversion trajectory depending on whether they remained wetlands or converted to another land cover/use. | The dependent variable. |
| Tenure Indicator Surface | OTS[1,2] | A raster dummy variable coding for the park (PVNP) and the non-protected parts of the Tempisque Watershed. | Protected area establishment is a key strategy in biodiversity conservation. While many studies have assessed the efficacy of protected areas in preventing forest conversion, little analogous work exists for wetland protection. |
| Wetland patch size | This analysis | Cells from the wetland trajectory surface were clumped into patches according to an 8-neighbor rule. Area was calculated for each patch. | Patch size suggests something about accessibility and degree to which other landscape variables (e.g. distance to nearest road) may affect wetland habitat. Wetland size is related to cost of conversion for agricultural land uses. |
| X Coordinate | This analysis | The X coordinate for the center point of each $30m^2$ sample cell. | There are several rivers with a primary directional gradient along east-west axis (e.g. Rio Canas and Bebedero) whose flows, alterations, seasonal fluctuations affect wetland areas; rivers have been increasingly channelized resulting in the desiccation of many wetland areas in the floodplain |
| Y Coordinate | This analysis | The Y coordinate for the center point of each $30m^2$ sample cell.. | The Tempisque River has a primary directional gradient running north-south whose flow has been significantly reduced in the last 20 years due to extraction for agricultural uses. The stream bed has also been incised and channelized, cutting off dendritic flow-like patterns in one key wetland area |
| Digital Elevation Model (DEM) | OTS[1,3] | 15 m 'x,y' resolution (re-sampled via nearest neighbor algorithm to 30 m resolution) and 1 m for 'z' resolution | Elevation partly controls accessibility and distance to river (important for irrigation). It interacts with slope. Occurrence of wetlands in this basin is related to seasonal surface flows and intersection of land surface with seasonally saturated water table. The lower the elevation, the greater the area of "up-stream" land cover change and hydrological changes must be integrated and the more likely wetland conversion should be |
| Slope Model | Calculated | 30 m resolution, calculated from DEM and expressed as percent | Percent slope partly controls accessibility since steep areas are more expensive to construct roads on. It interacts with elevation and also controls utility of land for agricultural purposes (many crops' cultivation requires 0% slope for mechanized harvest to function efficiently). Wetlands occurring at no slope should be more likely to follow the conversion trajectory. |

Table 3-2. Continued

| Raster Layer | Data Source | Description | Justification |
|---|---|---|---|
| Distance to Road | OTS[1,4] | Raster surface of 30m spatial resolution where value of each cell is the shortest distance to a public road. Note that private roads on large, commercial farms are not included here. | Roads imply changes to hydrological period and connectivity by altering overland water flow. Also, accessibility may play an important role in determining whether a wetland area will be drained and used for agriculture. |
| Distance to Panamerican | OTS[1,4] | Raster surface of 30m spatial resolution where value of each cell is the shortest distance to the Panamerican Highway | This is Costa Rica's major highway controlling access to national and international markets. Areas closer to the highway would seemingly be more likely to be converted directly (for agriculture) or indirectly through changes in water quality or hydroperiod. |
| Distance to River | OTS[1,4] | Raster surface of 30m spatial resolution where value of each cell is the shortest distance to the Tempisque and Bebedero (or confluence thereof) Rivers | The likelihood of switching to 12 month, intense cultivation system depends on the availability of water with the river being one of the two major water sources. Low lying, wetland areas adjacent to the river are probably more likely to be converted to irrigated agriculture than such areas that are farther from a water source. |
| Distance to Population Center | INEC[5] | Raster surface of 30m spatial resolution where value of each cell is the shortest distance to one of six population centers in the watershed | Proximity to towns is related to the likelihood of lowland areas being used for agriculture or probability of incurring hydrologic alterations related to urbanization. Urban runoff contributes to eutrophication of fresh water lagoons that may result in a state change and proximity to population centers implies sufficient labor pool to work the land. This variable is somewhat correlated with d_rds (+0.401). |
| Coordinate Interactions | Calculated | $x^2$, $y^2$, $xy^2$, $x^2y$, $xy^3$, and $x^3y$ | These terms control for possible interactions or nonlinearities) between observation coordinates. |
| Biophysical Interactions | Calculated | slo.elev = slope * elevation<br>$slo^2$.elev = $slope^2$ * elevation<br>slo.$elev^2$ = slope * $elevation^2$<br>riv.slo = distance to river * slope<br>riv.elev = distance to river * elevation<br>rv.elslo = distance to river * elevation * slope | These terms control for possible interactions between slope, elevation and distance to river. |
| Built Structure Interactions (Accessibility) | Calculated | Acs_index = distance to Panam * distance to rd<br>Acs_indx2 = distance to Panam* distance to rd * distance to population center | These terms control for possible interactions between distance to the Panamerican Highway, distance to road, and distance to population center. |

[1]OTS = Organization for Tropical Studies. [2]Vector layer of PVNP obtained from OTS and a raster indicator surface was created indicating cells within park and cells in remainder of watershed. [3]Produced by the Instituto Geografico Nacional (IGN) through interpolation of point data on topographic maps. [4]Produced by digitizing data from topographic maps [5]Population statistics were obtained from the Instituto Nacional de Estadisticas y Censos in order to identify population centers comprising 90% of the basin's population; coordinates of these six cities were taken at central square using a handheld GPS (~5m accuracy).

Table 3-3. Eigenvalues and percent variance accounted for by each of the three principal components extracted through PCA.

| Component | Eigenvalues | % of Variance | Cumulative % |
|---|---|---|---|
| 1 | 1.80 | 25.65 | 28.11 |
| 2 | 1.78 | 25.41 | 53.31 |
| 3 | 1.55 | 22.16 | 73.22 |

Table 3-4. Rotated structure matrix for the seven original predictor variables. Three principal components extracted through PCA are shown with their respective loadings. "Distance to population center" has been abbreviated, for example, as "d. pop. ctr," etc.

| | Component | | | |
|---|---|---|---|---|
| | PC1 | PC2 | PC33 | |
| Predictor | socioeconomic | north-south | topographic | Communalities |
| y | -.52 | -.78 | .05 | .88 |
| elevation | -.19 | -.11 | .86 | .78 |
| slope | .11 | .06 | .79 | .64 |
| d. river | -.38 | .57 | .39 | .63 |
| d. roads | .70 | -.15 | -.14 | .53 |
| d. pop. ctr. | .90 | .048 | .09 | .82 |
| d. panamerican | -.20 | .89 | -.11 | .85 |

Table 3-5. Model performance statistics computed with independent observations and model coefficients estimated with (Model 1) and without (Model 2) wetland patch size. AIC = Akaike information criterion, $R^2$ = Nagelkerke's pseudo- $R^2$ , AUC = Area Under the Curve for Receiver Operation Characteristic (ROC) plots, $e^B$ = exponentiation of the coefficient.

| Model | AIC | $R^2$ | AUC | Socio-economic ($e^B$) | North-south ($e^B$) | Topo-Graphic ($e^B$) | Tenure ($e^B$) | Patch Size ($e^B$) | Inter-cept ($e^B$) |
|---|---|---|---|---|---|---|---|---|---|
| 1 | 119,820 | 0.48 | 0.79 | -.81 | -.85 | 2.68 | -1.00 | -0.00 | 2.147 |
| | | | | (0.45) | (0.43) | (14.57) | (0.37) | (1.00) | (8.56) |
| 2 | 125,404 | 0.44 | 0.81 | -1.03 | -.82 | 3.10 | -.91 | excluded | 1.99 |
| | | | | (0.36) | (0.44) | (22.11) | (0.40) | | (7.32) |

*All coefficients for models 1 and 2 were significant at $p<0.001$ level.

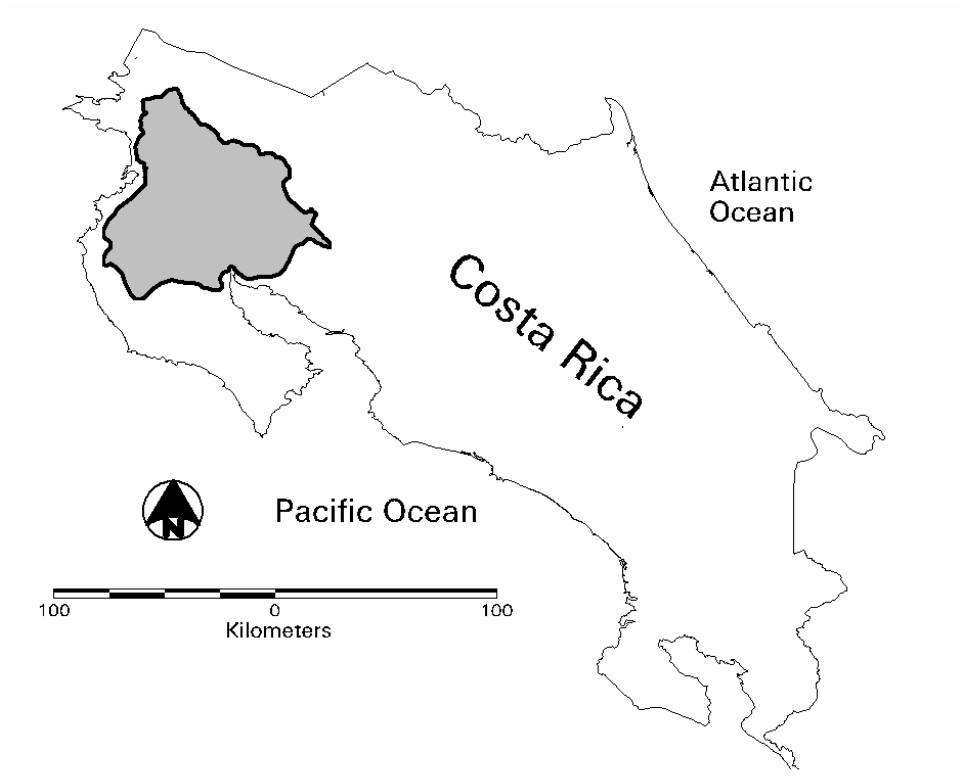


Figure 3-1. Map of the Temspique River Watershed in northwestern Costa Rica.

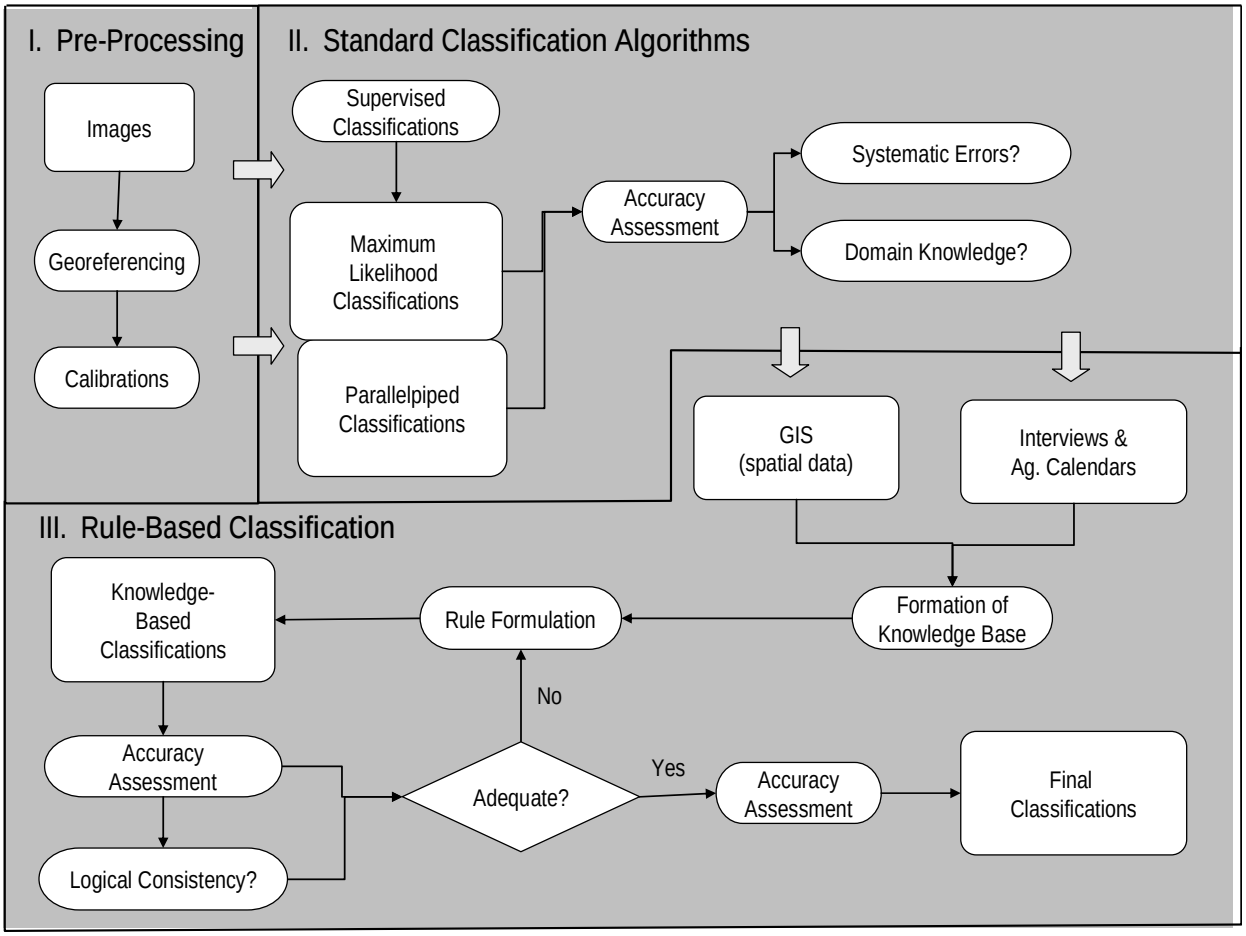


Figure 3-2. Flowchart of image rule-based land cover classification procedure used to derive binary "wetland" and "non-wetland" land cover maps for 1975, 1987 and 2000 Landsat images (Reprinted with permission from Daniels 2006. Incorporating domain knowledge and spatial relationships into land cover classifications: a rule-based approach. International Journal of Remote Sensing 27(14): 2949-2975). "Ag. Calendars" refers to an organized matrix of data compiled from interviews with local farmers and officials at the Ministry of Agriculture (MAG) detailing crop phenology and the timing of specific land use actions (field preparation; planting; pesticide, herbicide and fertilization application; harvest; post-harvest field maintenance, etc.) for all major crops.

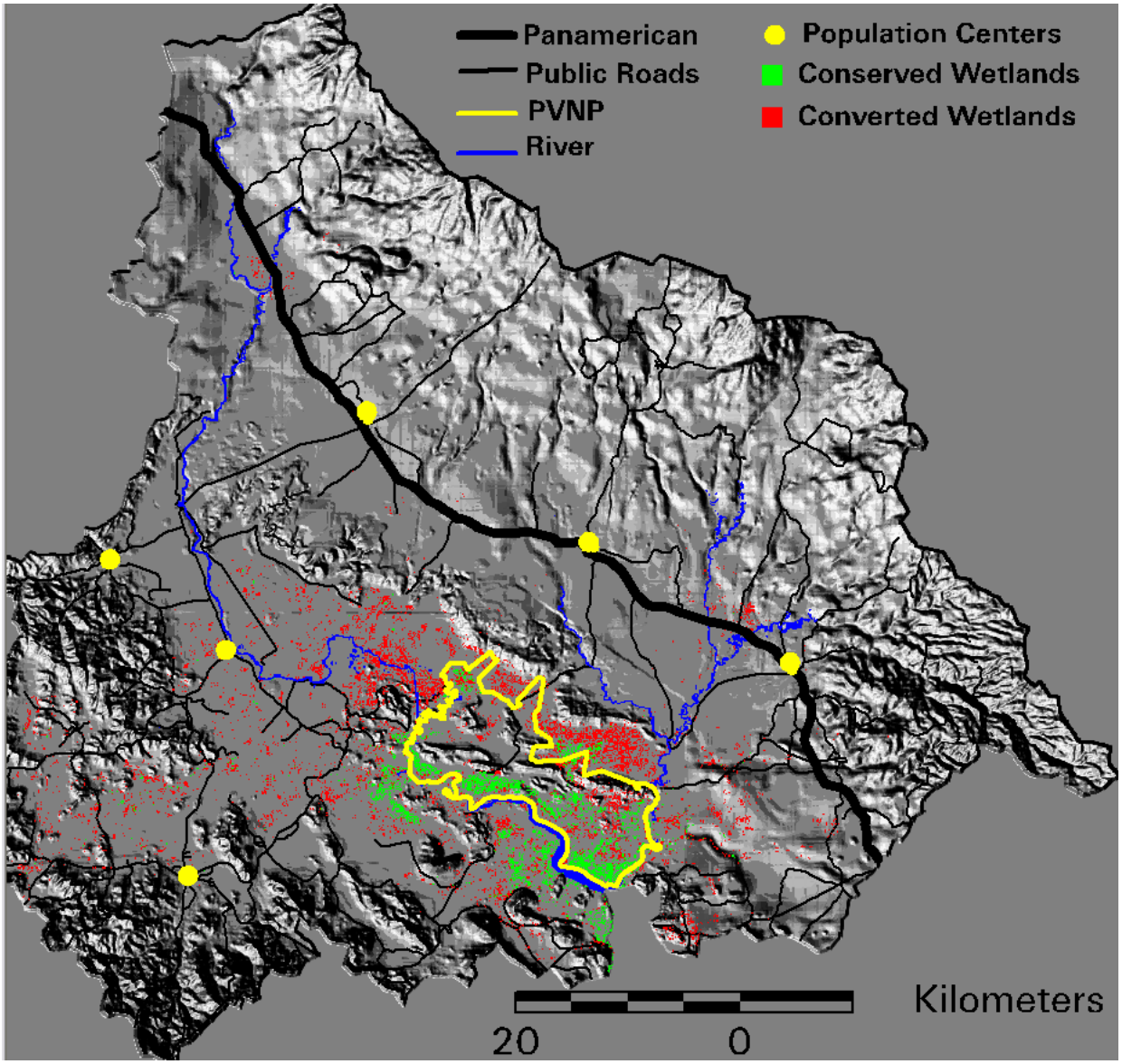


Figure 3-3. Map of landscape features in the Tempisque Basin that influence wetland conversion.

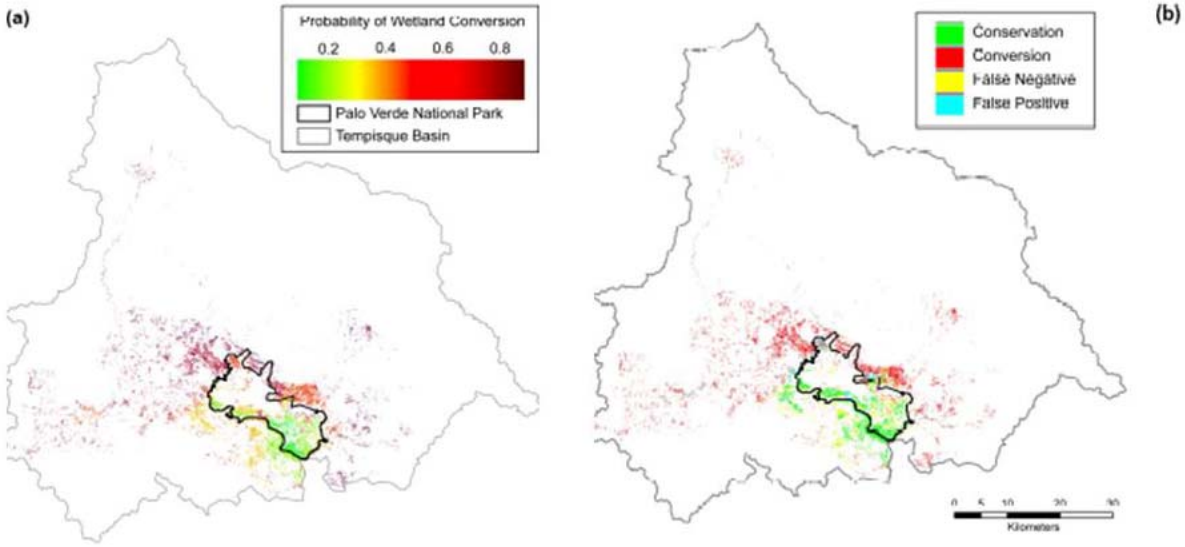


Figure 3-4**.** (a) Map of probability of wetland conversion. In the non-protected watershed, probability generally increases as distance from central, lower watershed increases, suggesting a "sweeping" typology of conversion. Within PVNP, the typology is much more patchy and the probability of conversion is generally much lower, though some areas still may be at risk for conversion (b) Map of actual wetland trajectories and model prediction errors. Correctly predicted conserved sites (green) are located predominantly in the central, lower watershed. Correctly predicted converted sites (red) generally decrease in frequency from the outer extremities of wetland occurrence in the watershed, relative to the discharge point of the river, in toward the central, lower watershed. False negatives, failure to predict conversion when it actually happened (yellow), are concentrated in the lower, central watershed both inside and outside of PVNP. Errors of commission, false prediction of wetland conversion when the site was conserved (cyan), occur in small, cluster-like areas within the central, lower watershed, in proximity to PVNP.

# CHAPTER 4
# A DECADE OF PAYMENTS FOR ENVIRONMENTAL SERVICES (PES): BUILDING ON COSTA RICA'S MODEL AND APPLYING LESSONS LEARNED


## Summary

Costa Rica has pioneered a nation-wide payments for environmental services (PES) scheme that addresses the critical role of private property land use in the provision of ecosystem services. The scheme complements the country's lauded national park system, effectively matching it in area. We describe the origin and functioning of Costa Rica's PES. We then explore a decade of national-level empirical trends (1997-2006) which demonstrate both achievements and challenges. Costa Rica's experience highlights the real-world hurdles of PES implementation and may prove instructive to emerging and future PES schemes. Institutional-design tradeoffs entail striking a balance between efficiency versus equity in participation, production versus conservation modalities, and optimal provisioning of ecosystem services versus achievement of socioeconomic objectives. We suggest several design-enhancements for Costa Rica's scheme. These include decoupling the finance of PES monitoring from the monitoring itself; strategically targeting PES land for both ecological and social objectives; reverse auctioning PES contracts to enhance efficiency and laddering contracts over different time spans to enhance ecosystem service continuity. The long term viability and credibility of PES as a policy tool hinges on learning from the experience of existing programs and on continual innovation. Costa Rica is well-positioned to begin pilot testing some of these nuanced PES design elements.


## Introduction

While certainly not the only approach to conserving and managing ecosystem services, payments for ecosystem services (PES) is the first conservation mechanism explicitly designed to address these positive externalities. Variants of PES have existed since at least 1985 when the U.S.

Conservation Reserve Program began purchasing long-term cropland retirement on U.S. farms (Szentandrasi et al., 1995). This voluntary program retires agricultural production in exchange for several ecosystem services including soil erosion reduction, habitat provision, and improved water quality. In the developing world, Costa Rica is not only a PES pioneer, but has successfully implemented the only nationwide program to-date.

Over the last decade, PES in Costa Rica and elsewhere has evolved into a more-formalized approach to manage and sustain ecosystem services. PES-based conservation efforts have proliferated in the developing world, and are being actively promoted by international aid and conservation organizations. PES goals may include both ecological objectives, like biodiversity conservation (Pagiola et al., 2005a), and social benefits like poverty alleviation (Pagiola et al., 2005b) and enhanced land tenure security (Grieg-Gran et al., 2005). As with any conservation mechanism, Costa Rica's experience illustrates that PES entails navigating a complex array of program-design tradeoffs. As PES institutions continue developing, it is important to clearly define and evaluate PES in light of specific program goals to ensure they achieve their intended objectives (Mulder and Coppolillo, 2005). Indeed, the long-term viability and credibility of PES as a policy tool hinges on learning from Costa Rica's experience and leadership in the field.

PES schemes present many complex institutional and political design challenges due to the broad array of issues that must be addressed and the logistics of dealing with many stakeholders. An extensive literature exists on Costa Rica's PES (Chomitz et al., 1999; Landell-Mills and Porras, 2002; Rojas and Aylward, 2003; Zbinden and Lee, 2005; Miranda et al., 2006; Pagiola, 2006; Sierra and Russman, 2006; Wunder, 2005; Wunder, 2007). Our goal is to complement this body of literature by reflecting on empirical trends from 1997 to 2006. Costa

Rica's PES system is currently gearing up to implement a suite of innovations and enhancements after reflecting on the first World Bank/GEF-affiliated project, Ecomarkets (World Bank 2000). This new phase represents a second round of collaboration between Costa Rica and World Bank/GEF with the goal of mainstreaming and scaling up PES through focusing on identifying and refining sustainable funding mechanisms. Our review dovetails nicely with this initiative.

Details of Costa Rica's PES scheme have not always been consistent or well-documented in the literature, likely due to the evolving legal structure of the program along with divergence between the written laws and their effective regional implementation. Our objectives are to accurately describe PES design and implementation and discuss themes that are critical to the enhancement and continued evolution of the system. In section 2, we describe the origin and operations of Costa Rica's PES, and present national-level data to illuminate trends, achievements and tradeoffs. In section 3, we analyze several themes critical to PES systems. Costanza and Farley (this issue) further discuss the importance of these themes to successful PES programs: institutional design based on program administration and opportunity costs; ecosystem service bundling and payment levels; program financing and equity; spatial considerations for

PES implementation; and finally, tradeoffs in PES systems relevant to socioeconomic objectives. Some of the challenges we identify are unique to Costa Rica; others apply to PES programs more broadly. Both theory and experiences from elsewhere offer meaningful insight for enhancing Costa Rica's PES design, while programs around the globe stand to learn much from Costa Rica's experience.

## History and Trends in Costa Rica's PES Program

### PES Evolution and Scheme Design

Though currently well-known for its conservation programs, in the recent past Costa Rica had one of the highest deforestation rates in the world; between 1986 and 1991 Costa Rica was losing

4.2% of remaining forest cover per year (Sanchez-Azofeifa et al., 2001). To address this and other environmental issues, Costa Rica began building a system of national parks and private reserves in the 1970s, which today encompasses over one quarter of the national territory. Yet deforestation in non-protected areas continues to occur, threatening to isolate protected areas as forest islands (Sanchez-Azofeifa et al., 2003). Further expansion of non-extractive protected areas is impractical, if not inappropriate, given Costa Rica's population growth rate of 1.7% (World Bank, 2007) and lingering concerns over lack of just compensation for private property incorporated into the current park system (Steed, 2003). PES emerged in Costa Rica partly in response to the need for addressing land use choices on private property.

In much of Latin America, the forestry sector has a long history of government subsidies through interest-free loans, tax exemptions, provision of seedlings, extension services and even direct payments (CIFOR, 1999). In recent decades Costa Rica has been no exception (Brockett and Gottfried, 2002). Evolution of forestry incentives began in the late 1970s with tax credits aimed at offsetting the costs involved in establishing and managing forest plantations (Figure4-1). From remarkably favorable credit conditions, to tradable tax vouchers, Costa Rica used subsidies to promote growth in the forestry sector. Over time, however, international pressure mounted to eliminate such subsidies. An acute financial crisis in the early 1980s saw the country become the first in a series of Latin American nations to default on international loans (Lara, 1995) at a time when their per capita debt load was among the highest in the developing world (Biesanz et al., 1982). Subsidies to the forestry sector were politically unsustainable since Costa Ricans failed to see much contribution from forestry to the local economy. The third World Bank loan negotiated during the ensuing structural adjustments abolished subsidies to the forestry sector (Watson et al., 1998). Yet Costa Rica cleverly turned the subsidy concept on its

head by articulating the broader social cost of deforestation and the need to compensate private land owners for the ecosystem services their forest stewardship provides. Thus, Costa Rica's archetype PES program evolved seamlessly from the existing trajectory of forestry incentives (Figure4-1), shifting the nominal focus from timber to conservation. Capacity-building and ecological awareness played an important role in affording this policy evolution.

The authorizing legislation for PES in Costa Rica was the fourth national forestry law passed in 1996 (Ley 7575, 4-16-96, Gaceta 72, Alcance 21). Ley 7575 recognizes four environmental services provided by forest ecosystems: biodiversity, watershed function, scenic beauty, and greenhouse gas mitigation through the storage and sequestration of atmospheric carbon. Land owners may sell their environmental services through one of several modalities which currently include (a) reforestation through plantations, (b) protection of existing forest, (c) natural forest regeneration, and (d) agroforestry systems (Gaceta 51, 3-13-07). Table 4-1 reviews the criteria and implementation history for Costa Rica's PES approaches. The payment per hectare is the same for all land owners within each modality (Table 4-1). Payments occur for five years, during which the PES-related land-use restriction is supposed to be noted on the property title to ensure that the service provision continues even if a property is deeded to another party.

Each year a program budget and PES procedures manual are published by the Ministry of Environment and Energy (MINAE) and the PES administrative agency, the National Forestry Financing Fund (FONAFIFO), respectively. MINAE determines the distribution of funds across modalities and also provides some direction with regard to priority zones for each method. From the publication date of these executive decrees each year, interested land owners meeting the requirements have fifty days to submit the necessary paperwork to the appropriate regional

FONAFIFO offices. Generally, the program can only accommodate about a quarter of the annual applicants into the scheme. By design, FONAFIFO should prioritize contracts within biodiversity conservation corridors identified by the GRUAS reports (García, 1996; Castillo, 2006) and through annual consultation with the national system of protected areas (SINAC) within MINAE (Rojas and Aylward, 2003). In practice, however, prioritization of PES contracts varies regionally. Regions that were not targeted by the World Bank-funded Mesoamerican Biological Corridor initiative, and/or that lack a strong civil society presence to conduct outreach, may operate on a first-come, first-served model of prioritization out of logistical necessity (Daniels, personal observation).

Each contracted environmental service provider must have a formal forest management plan designed by a professional forester, regente, according to the specifications of the modality in which they are participating (Article 20, Ley 7575). The fixed cost of this activity is taken off of the top of the program payment and is thus proportionately higher for small holders. Other responsibilities include posting signage on the land declaring that it is protected from hunting, fire and logging (Article 12, Gaceta 51, 3-13-07). The same regentes that write management plans are charged with monitoring compliance with PES regulations (Article 21, Ley 7575). Regentes are required to perform a site visit every twelve months for the life of the contract (Article 10.2, Gaceta 51, 3-13-07).

**Empirical Trends for Costa Rican PES**

The mean annual PES budget over the last decade exceeds $13.3 million USD or 0.43 % of Costa Rica's 2006 national budget.[1] To put this in perspective, the entire EPA budget for 2006 comprised 0.0003% of the U.S. federal budget[1]or three orders of magnitude difference relative to

[1] Data Sources: PES budget: FONAFIFO, Costa Rica 2006 National Budget: CIA World Fact Book, EPA 2006 Budget: EPA website, US 2006 National Budget: CIA World Fact Book

this single program within Costa Rica's portfolio of conservation initiatives. The extent of Costa Rica's investment in PES underscores their commitment to conserving environmental services by addressing land use management on private property. It also highlights the importance of iteratively reviewing the institutional design of PES and its implementation in the name of enhancing efficiency and efficacy.

Overall budgetary efficiency, plotted as cumulative area enrolled in PES versus cumulative PES budget, corresponds roughly to the five year payment cycles (Figure4-2**a**). The slope between data points for individual years represents the gain in PES area per unit of FONAFIFO's annual budget. Recruitment of area into the PES scheme diminished per unit of the budget over the first years of the program, up through 2002. This is a function of having to spend an increasing portion of each successive annual budget servicing contracts from past years. By 2002, the 1997 cohort—the largest in the program's history with 102,784 ha—had finished receiving payments and program efficiency increased markedly. Conceptualizing PES as a cumulative forest protection scheme for the provision of environmental services as in Figure4-2**a** assumes that land owners will abide by Article 19 of Ley 7575 once the payment period has expired. That is, land owners will continue to protect and maintain forest cover as mandated under the law so that PES investments have cumulative and lasting effects for environmental service provision. In practice, however, Article 19 is somewhat unrealistic and is weakly enforced.[2] For example, within the forestry modalities of PES, land owners may choose not to re-plant a plantation site after timber harvest and the expiration of the PES contract. In the forest

[2] Article 19 of Ley 7575 was instrumental in establishing a favorable context for PES, but is problematic in its implementation. With regard to opportunity cost, the program payment (no matter how low) technically always exceeds the land rent from the next best land use, given that it is illegal to change from forest land use to another activity.

protection modality of PES, the common practice of forest thinning and/or clearing of the understory (socolando) may ensue after the payments end, making the gradual land use change difficult to detect (Daniels, personal observations). Figure4-2**b** illustrates the difference between the best and worst case PES implementation scenarios respectively. The upper curve represents conservation of all PES forest, even after payments end—likely an unrealistic scenario. The lower curve represents conservation of only the forest areas receiving contract payments (i.e., forest area for expired PES contracts is subtracted off the running cumulative area). Institutional design and supporting forest conservation policies are critical in determining where the empirical curve falls between these two extremes.

Another aspect of budgetary efficiency relates to hectares per contract for individual land owners. Figure4-3 illustrates that across some time steps (e.g., 1998 to 1999), the total PES area recruited may increase while the number of contracts stays roughly the same. This means the area per contract is greater and the relative administrative cost per hectare recruited is lower. In contrast, from 2004 to 2005, the number of contracts is constant while the recruited area drops precipitously (i.e., area per contract is much smaller). This indicates a tradeoff between program efficiency and equitable distribution of environmental service contracts across the range of property holdings.

The overwhelming majority (89%) of recruited PES area throughout the history of the program has been for the forest protection modality, with only five and 6% falling in the reforestation and management modalities, respectively. The budgetary breakdown, however, is somewhat different given that payments per hectare of the timber-related modalities is over twice the payment level for forest protection in order to cover the higher costs of planting and technical assistance (Table 4-1). A decrease in recruited PES area from 2004 to the present reflects the

implementation of the agroforestry modality which is based on payments per tree rather than area of forest contracted for environmental service provision (Figure4-3).

Over the last decade, Costa Rica's PES program has purchased ecosystem services from over half a million hectares of land in "forest use" (5,314 $km^2$), including regenerating forest and plantations at various stages of the timber production cycle (Figure4-3). As such PES has provided a significant private-property complement to the country's network of national parks, which comprises only slightly more area (5,415 $km^2$). As Costa Rica begins implementing a new phase of PES (corresponding to a second World Bank/GEF-sponsored project), reflecting on institutional design at this point should enhance existing arrangements and facilitate innovations that further improve PES performance.

## Evaluating Costa Rica's PES Program

### PES Administration

FONAFIFO, the semi-autonomous arm of MINAE that administers PES, has considerable freedom and flexibility with regard to how the program is implemented. A 1990 budgetary law (Article 32, Ley 7216, Gaceta 245, Alcance 48, 12-26-90) created the agency and charged it with financing forestry initiatives among small and medium-sized producers. As such, the institutional strengths of FONAFIFO arguably lie in its forestry-related capacities. The agency was charged with managing the PES scheme only since 1996 (Article 46, Ley 7575). FONAFIFO's Board of Directors (Article 48) is comprised of two representatives from the private forestry sector, one industrial and one small to medium-sized producer group (e.g., JUNAFORCA); one representative from the Ministry of Agriculture; one from the national banking system; and a single representative from the Ministry of Environment. The Board essentially writes the executive decrees defining explicit participation criteria, modalities and payment details in the annual PES Procedures Manual. This leadership structure and the

historical role of FONAFIFO prior to PES may have set forth some degree of institutional path-dependency, restricting PES design and implementation innovations to a degree. Political pressure from the forestry lobby has further reinforced this structure.

FONAFIFO's particular institutional structure has both positive and negative consequences regarding PES objectives. Benefits of the forestry-bias to date include the development of progressive, technically-sound small forestry operations that have at least nominally contributed to rural development. By facilitating the establishment of such forestry plantations, the scheme design may reduce legal and illegal logging pressure on natural forests.[3] Plantations also generate carbon credits with potential for sales on the international market, thereby creating a positive feedback for PES funding (e.g., a current proposal for the World Bank's BioCarbon Fund). The negative consequence of the institutional forestry bias from a conservation perspective is that ecosystem services provided by plantation land use are production-biased relative to those provided by natural forest cover. To date, the scheme has identified generalized categories of environmental services provided by land uses (i.e., modalities) already employed in pre-PES forestry incentives (Figure4-1), as opposed to identifying ecosystem functions and services, and <u>then</u> defining with greater nuance what land cover, land use and management practices best provide these services. New modalities are currently being proposed, however, and will be regionalized according to local needs (World Bank 2006).

A holistic approach to forest ecosystem service provision and management requires that production, consumption and conservation issues be addressed in lockstep to enhance <u>net levels</u> of service provision. The tradeoff between production and conservation modalities, however,

[3] A decade ago, 50% of local timber came from natural forests compared with only 5% today (MINAE/National Forest Office, 2004).

has been highly politicized since the beginning of PES in Costa Rica. Sound planning and rational discourse sometimes get lost in the propaganda from the two artificially-distant extremes. For example, the forest management modality was eliminated entirely in 2002, arguably on principle alone, reflecting the delicate balancing-act that FONAFIFO and policymakers face in sustaining support for PES in Costa Rican society. Unfortunately minimal rigorous peer-reviewed research exists to objectively provide insights regarding the optimum distribution of PES area and funding across modalities for a range of different economic and ecological scenarios.

From 1990 through 2003, FONAFIFO's role was largely that of a bank. In essence, its mandate is still financial in nature—collecting, managing and dispersing funds through payments and loans (Article 46, Ley 7575). Yet, PES implementation entails a host of administrative, information-systems, and monitoring/reporting considerations which the agency accomplishes using less than 10% of its given annual budget. In 2003, FONAFIFO took PES field administration from SINAC through the staffing of eight regional offices (housed within regional SINAC offices). Over time, the agency has become savvier in managing the challenging ground-based logistics of PES implementation. Decentralization has enhanced both efficiency and accessibility for interested landholders. The eight administrative zones are divided into geographic regions that do not correspond to natural landscape units like watersheds, however.

Monitoring for Costa Rica's PES scheme is weak and leaves room for improvement. The duty of all field verification, management plan drafting and monitoring falls, by design, to third party agronomists and foresters (regentes) compensated by PES participants out of the program payment (Article 21 of Ley 7575). Contracted foresters may have a disincentive to report non-compliance with PES contracts since they may fail to receive compensation if a non-compliant

PES contract is disqualified. Further, regentes may lose the non-compliant contract from their portfolio of managed contracts. Since regentes have public faith (fe publica), there is little oversight of their work. FONAFIFO's Board of Directors has been slow to develop explicit criteria and procedures that regentes should follow during the initial and follow-up site visits. For example, only in 2004 did the Procedures Manual specify how, where and in what units GPS points should be taken on-site by the regente to identify the property being contracted for PES (Gaceta 46, 3-5-04). Data collected prior to 2004 were often recorded in a variety of incompatible map datums and projections. Only in 2006 did the manual require regentes to begin mapping the actual contracted forest area within the larger landholding. This marks a dramatic improvement as officials, researchers and conservation groups may now use remote sensing methods to complement field-based monitoring and begin to systematically quantify the impacts of PES on forest cover.

FONAFIFO has demonstrated its capacity to effectively incorporate lessons-learned by adapting its administrative design. Nevertheless, the PES monitoring mechanism still merits considerable re-thinking. First, foresters may not always be the most appropriately trained for evaluating ecosystem services or monitoring their provision, particularly as new modalities are added in the future. A more robust approach incorporating ecologists, hydrologists, geographers, ecological economists and landscape planners may be beneficial. Greater monitoring oversight, including penalties for hasty technical work, is also needed. The program should move toward completely decoupling the financing of monitoring from the act of monitoring itself. For example, fees now paid directly to regentes could be deposited into a general fund for each region. Then payments could be made out of the fund to regentes randomly assigned to perform

follow-up visits, without regard to which regente had written the original management plan. In this way, regentes could better self-police in executing technical and monitoring duties.

**Opportunity Costs**

The payment amount for Costa Rica's PES program has long been a topic of debate. In theory, the payment should exceed the land rent earned for the next-best land use option (i.e., the opportunity cost). Payments were derived from calculating an average opportunity cost for the most immediate land use option prior to PES initiation over a decade ago, which was assumed to be cattle ranching. Since that time, FONAFIFO has annually adjusted payments upwards, to minimally match inflation (with a marked increase in 2005). There are several problems with this approach. Land rent for cattle ranching varies greatly depending on location and specialty (breeding, dairy or meat). Cattle ranching was relatively less profitable due to low beef prices at the time (Arroyo-Mora et al., 2005). And finally, low-intensity cattle ranching is no longer necessarily the most immediate land use alternative as some regions of Costa Rica have been moving away from this extensive production model toward higher-intensity land uses (Daniels, in prep).

Intensive agriculture and development/urbanization are increasingly prevalent land use options. Sites suitable for cultivating export-grade pineapple, for example, can be rented for about $390/ha per year or sold for around $5800/ha (Oviedo, 2006). Such high land rent is possible by externalizing the costs of environmental degradation like water pollution. As long as local to international laws and institutions fail to internalize social costs, PES may be less competitive, highlighting the importance of policy coherence in effective PES implementation (Costanza and Farley, this issue). The PES payment of $41/ha per year for natural forest regeneration or $64/ha for forest protection is trivial for those interested in profits alone, if their land is suited for intensive agriculture. PES is thus generally more attractive on marginal lands

which may or may not provide ample levels of ecosystem services for a particular landscape or region (see section 3.6). Rapid development in some regions increases the need for environmental services that reduce peak stream flows and prevent flooding (Marsik and Waylen, 2006). Nearly three million square meters of new construction were permitted within Costa Rica in 2004 alone (Estado de la Nacion, 2006). Yet the very process of urbanization often precludes even the consideration of PES because of the comparatively immense one-time profit that a landowner can earn by selling their property. Land speculation and real estate development are particularly prevalent in coastal regions and in the urban Central Valley. Zbinden and Lee (2005) point out the need for more research on opportunity cost dynamics in Costa Rica. The long-term viability of PES depends upon addressing these difficult issues of modern-day land use competition openly without being perceived as a threat to PES validity and utility.

Consideration of how Costa Rican land use economics have changed in recent years underscores the importance of a PES design that incorporates a feedback loop for changing economic contexts. Periodic updates regarding opportunity costs could be used in conjunction with PES contract "laddering" to ensure provision of ecosystem services over appropriate time scales despite economic change. That is, rather than having a fixed term (currently five years) for PES contracts, laddering over different term lengths with higher payment rates for longer contracts would help ensure at least some critical level of environmental service provision even when market conditions make PES a less-attractive land use. Furthermore, a vast literature on adoption of conservation-friendly management practices and land use decision-making suggests that the process is far more complex than accounting for farm profit levels alone (Godoy, 1992; Ayuk, 1997; Neupane et al., 2002; Berentsen et al., 2007). This suggests that there may be room for outreach and education to enhance consideration of the non-monetary factors involved in the

decision to participate in PES (e.g., the long-held Costa Rican ideal of maintaining the small family farm appears to play an important role). Ecosystem services valuation must be tied to overall quality-of-life considerations. Improved understanding of the dynamics between natural, social, built and human capital can help better inform appropriate land use decisions (Costanza. et al., 1997; Costanza, 2001).

**Ecosystem Service Bundling**

The natural functioning of ecosystems delivers inseparable "bundles" of ecosystem services (Brennan, 1995). Often, service delivery occurs in synergistic fashion, especially between adjacent ecosystems. Certain management strategies, however, can enhance some services relative to others, or even result in their total loss. Prudent ecosystem service management requires considering complementarity (e.g., riparian forest habitat and enhanced water quality) or competition (e.g., forest habitat versus food production from a cleared agricultural field) among services . Figure4-4 conceptually illustrates a multi-dimensional production possibility frontier for several ecosystem services. One reasonable management objective could be to increase the volume defined by the provision level of interacting services. Given a target level for a focal service, another goal could be to achieve the corresponding maximum provision level for other bundled services as illustrated. Unfortunately, however, many ecosystem service tradeoffs are still either unknown or poorly understood (Rodriguez et al., 2006).

Costa Rica's PES program bundles the sale of ecosystem services. The assumption is that the prescribed land use of a given modality will result in the provision of at least one or more of the four environmental services specified by the program. Yet the different modalities leave room for various levels of service provision, with a somewhat nebulous link between the modality, level of service provision and flat payment rate. Differentiated payments would better

reflect the degree of ecosystem service bundling provided by a given contract and have been proposed for future implementation in Costa Rica (World Bank 2006). For example, land that contains old growth forest cover would certainly store more carbon, while simultaneously providing greater biodiversity, than equal area of early successional forest within the same life zone. The PES program might consider paying more for the old growth forest within its forest protection modality. Differentiated payments could provide part of the missing link in the current institutional design toward maximizing the service provision volume depicted in Figure4-4. Furthermore, allowing graduated payments through multiple tiers of ecosystem services within a modality may increase PES retention and reenlistment.

**Sustainable Financing**

A successful PES program must have the appropriate mechanisms and political will to capture funding from a wide range of ecosystem service beneficiaries. Costa Rica's scheme successfully exemplifies this – a monopsony that captures ecosystem service "sales" across multiple scales. The scheme indirectly connects local, regional and international buyers of ecosystem services to individual land owners (Figure4-5). Locally, for example, a new decree is being phased in over the next seven years (Decreto 32868, Gaceta 21, 1-30-06) where water concessionaires pay a fixed tariff that gets invested in watershed protection. Internationally, Costa Rica has marketed discrete carbon storage/sequestration services such as the $2 million certified offset sold to Norway in 1997. FONAFIFO successfully bundles ecosystem services, while simultaneously exploiting markets for the sale of discrete services.

Sustainable financing mechanisms improve the likelihood that resources will be available to continue funding PES programs into the indefinite future. Figure4-6 illustrates the continuum of relative financial sustainability among funding sources both within-country and internationally. Reliance on external loans and grants is the least secure PES financing source.

Costa Rica has benefited from being a pioneer in the field of PES. Yet as other nations begin or expand PES initiatives, Costa Rica may face greater competition for such funding, stressing the need to begin pilot-testing designs that enhance efficiency. Stronger international treaties on biodiversity and climate change that require payments for international public goods would provide a more secure external funding source (Farley et al., this issue). Such arrangements would reduce free riding by developed nations and contribute to PES success by enhancing the demand-side of environmental markets.

Within-country, financing may entail voluntary payments, funding from the general budget, or funding from a specific activity. Voluntary purchases, like FONAFIFO's innovative Environmental Service Certificates (CSAs) sold to local utilities like Energía Global, provide funding but allow free riding by non-purchasing firms. CSAs are certificates of bundled ecosystem services that any entity may purchase. In practice, CSAs function like a donation to the PES scheme, but the concept is radically different. Great potential exists for enhancing CSA sales through an eco-friendly certification process for tourism-related businesses. Criteria for the certification might include carbon-neutrality, biodiversity conservation, and enhancing hydrologic functions (e.g., offsetting diminished aquifer recharge for each square meter of constructed surface area). Firms could provide these environmental services on their property or through purchasing CSAs. This would generate revenues for PES while internalizing some of the environmental and social costs of the largest foreign currency-earner in the Costa Rican economy, tourism (Brockett and Gottfried, 2002). Currently, however, few tourism businesses in highly-visited areas appear to be aware of CSAs, underscoring the need for an outreach mechanism to capture such sales.

PES funding from a nation's general treasury risks competition from numerous other budgetary needs. Therefore taxes or fees on goods or services related to provisioning of ecosystem services are more sustainable revenue sources for PES. Costa Rica has a fuel tax, currently about 28 cents/liter of gasoline (Decree 33570, 1-8-07). By law 3.5% of the revenues should be channeled to FONAFIFO to fund PES (Article 5, Ley 8114, 7-9-01), significantly less than the originally-intended one-third of revenues. This design is conceptually sound since it requires polluters to pay for the atmospheric waste absorption capacity for $CO_2$. However, revenues first pass through the Ministry of Finance where competition for other legitimate uses, e.g., the Costa Rican social security system, is understandably great. Fuel tax revenues actually dedicated to FONAFIFO do not always meet their intended level. Three and 0.5% of fuel tax revenues would be about $8.6 million per year (Miranda et al. 2006), but FONAFIFO's budget has been as low as $3.1 million in 2005. Government estimates of income tax evasion in Costa Rica are high (Lutz and Daly, 1991; O'Grady, 2006), though recent reforms have shown promise in turning this trend around (Umaña, personal observation). The fuel tax gap may arguably be making up for much-needed revenues. The PES funding shortfall creates a gap between the supply of landowners interested in PES and the demand FONAFIFO can generate with its given budget. Both the water tariff and fuel taxes apply to goods with inelastic demand. Such taxes should be more sustainable under changing economic conditions than those on goods or services with more elastic demand, like tourism.

Biodiversity services have proven especially challenging for developing targeted financing instruments at the local level. Costa Rica has devised a particularly innovative and sound strategy to capitalize a trust fund (The Trust Fund for Sustainable Biodiversity Conservation) that will serve as the financier of last resorts. The fund will target zones of

globally-significant biodiversity that do not overlap with existing carbon and water-related PES funding mechanisms (World Bank 2006).

**Equity in Funding PES**

Sustaining funding for PES involves iterations of internalizing ecosystem services at the local, regional and global levels. An important component of sustainable funding mechanisms is rooted in effectively dealing with free riding - the act of benefiting from a service without paying for it (Olson, 1965). Globally, Costa Rica provides biodiversity conservation and carbon sequestration, services that yield global benefits yet most recipients at the global scale do not pay for these benefits (Farley et al, this issue). Both of these services are considered to be non-rival and non-excludable, placing them in the public goods realm (Samuelson, 1954; Randall, 1993). Who is responsible for managing and financing these services? Theory advises that governments should have a significant role in managing and directing general funds for non-excludable services (Randall, 1993; Daly and Farley, 2004). However, drawing from a country's internal general funds does not reflect the larger benefits to global society.

On a regional scale, scenic beauty is considered a non-rival but excludable and congestible service, because overuse of the landscape could diminish or potentially eradicate aesthetic qualities (Randall, 1993). One logical way to preserve the quality of a congestible service is to consider it a public good, which can then be subject to user fees or other methods of management (Randall, 1993; Bengston and Youn, 2006). Obvious users include tourists; past negotiations have occurred between actors such as hotels, rafting companies and tour industries (Pagiola, 2006). Yet because there is such a vast range of users in Costa Rica, identifying and maintaining a collective base from which to acquire funding is difficult. One way might be to implement a tourist fee, which would more evenly distribute the cost among the beneficiaries. Currently, visitors and citizen air travelers alike pay an exit fee of US $26; an additional fee may

be a simple way of using an established channel for funding scenic beauty provision. This would widely spread the burden of payments while increasing the funding pool. Reasonable thresholds could be established through willingness-to-pay surveys and by examining similar payment schemes. This mechanism does not burden the poor, since no such exit fee exists for land-based border crossings.

On a local level, the free-riding issue for water conservation stands to be controlled by the new water tariff, once implemented nationwide. The tariff will effectively equalize costs across all concession holders. Yet, if the tariff is passed on to users, it may disproportionately burden the poor since, albeit a negligible $\$0.003/m^3$ (Article 5, Decreto 32868), it constitutes a higher percentage of their total income. The fuel tax charges citizens for carbon sequestration by forests, standardizing the funding of this service across local beneficiaries; but international beneficiaries continue to free-ride (Farley et al, this issue). Free-riding also occurs for biodiversity protection, as similar such measures do not exist to explicitly charge local beneficiaries.

**Spatial Variability and PES Targeting**

Ecosystem processes, climate, disturbance, and characteristics of human user populations clearly vary across Costa Rica's diverse geography, and interact to influence ecosystem service provision. Yet the flat payments in Costa Rica's PES scheme to-date fail to account for this variability. Carbon storage and sequestration vary greatly by forest type and successional stage (Rojas and Aylward, 2003). Landscape beauty is likely greatest in places of high visibility – (e.g., along roads, mountaintops). Areas of high biodiversity value are identified through the GRUAS reports (García, 1996; Castillo, 2006). Hydrologic services present scientific uncertainty as well as spatial dependence on human user populations. Forest type, climate, and landscape setting are all key factors influencing hydrologic services (Chomitz et al., 1999). De

Camino et al. (2002) developed a qualitative ranking system for ecosystem service provisioning by forest type, which could provide a basis for more empirical measurement of service provision differences, as proposed in the next phase of Costa Rican PES. Benefits from diverse services can be aggregated using indices (e.g., the U.S. Conservation Reserve Program "environmental benefits index" or Australia's BushTender "biodiversity quality index" (Chomitz et al., 2006).

Popular wisdom suggests that forests regulate high and low flow events, increase total water supply, and reduce erosion and sedimentation. Scientific evidence, though, presents a more complex and site-dependent view (Bruijnzeel, 2004; Bruijnzeel, 2006; Kosoy et al., 2007). Key findings of studies relevant to Costa Rica's PES program include: 1) Runoff is less in forests, except for cloud forests. 2) Dry season flow and groundwater recharge contributions from forests are site-specific, and largely depend on local geology, tree species composition, and successional stage. 3) Peak flows are mitigated in newly regrowing forests, but full benefits are achieved once complete vegetative cover becomes established. This effect is most prominent in small watersheds, and less important with increasing watershed area. 4) Forests encourage more rainfall only in cloud forests or over large geographic areas (e.g., the Amazon). 5) There is greater scientific consensus about the water quality and sediment reduction benefits provided by forests. Despite uncertainty about hydrologic services, utilities in Costa Rica have renewed their CSA contracts for the purchase of environmental services, indicating their satisfaction.

On an annual basis and at the nationwide scale, Costa Rica receives far more rainfall (170 $km^3$/yr) than its water use (6 $km^3$/yr, Pagiola, 2006). Despite this abundance of moisture, spatial and seasonal variability can cause serious water shortages with nationwide consequences. Since about 80% of Costa Rica's electricity is generated in hydroelectric plants, the variability of rainfall relative to plant locations is a critical concern. This is particularly true during ENSO

events since water levels in the Arenal reservoir, located in the driest Costa Rican province of Guanacaste, can be significantly diminished during El Niño (Amador et al., 2000). In fact, President Arias declared a national energy crisis in March 2007 due to insufficient electric production as a function of record-low water levels in concert with other malfunctions. Economic losses in the industrial sector alone summed to $20 million in a single week within a longer period of rolling blackouts (Avalos, 2007). This underscores the critical nature of considering spatial and temporal variability of hydrological functions. Appropriate spatial targeting of watershed services could offer greater resilience in times of climate anomalies and technical failures that affect national electricity supply.

Just as ecosystem service provisioning varies across landscapes, opportunity costs of their protection vary as well. In a general sense, environmental markets can improve conservation efficiency over command and control regulation by identifying specific locations or firms offering the lowest costs and greatest benefits (Tietenberg, 1989; Salzman and Ruhl, 2002; Pagiola et al., 2005a). Careful arrangement of PES payments may similarly achieve the same environmental benefits at lower costs. In a system of uniform payments, however, landowners with low opportunity costs receive rent from PES programs, reducing money available to spend elsewhere, while those with higher opportunity costs are unwilling to participate even if they could provide socially valuable ecosystem services. Through spatial targeting, payments can be matched to levels of service provision, eliminating the blunt subsidy nature of uniform payments across diverse landscapes (Salzman, 2005).

Tools for spatial targeting of ecosystem services have been developed and used in numerous geographic contexts and policy settings (Babcock et al., 1996; Babcock et al., 1997; Ando et al., 1998; Polasky et al., 2001; Stoms et al., 2004; Chomitz et al., 2006; Naidoo and

Ricketts, 2006; Beier and Patterson, in review). Costa Rica's PES scheme might gain from implementing a spatially-nuanced approach that employs these kinds of tools (Chomitz et al., 1999; Ferraro, 2001). Wünscher et al. (2007) highlight the efficiency gains in targeting payments to landowners based on both service provision and opportunity cost. They demonstrate that conservation gains for the Nicoya Peninsula in northwest Costa Rica would be 58-88% greater using a targeted PES system that ranked each parcel's total ecosystem services score and opportunity cost of service provision. The surveys that Wünscher et al. used to estimate opportunity costs are expensive and time-intensive, however, meaning they could undermine the efficiency gains of a spatial targeting. This is particularly true since both ecosystem functioning and opportunity costs are dynamic, requiring periodic updates that exceed the capacity of FONAFIFO's field staff. Thus, while a good theoretical concept, Costa Rica needs a more straightforward method to estimate landowner costs.

One solution might be reverse auctioning where landowners self-identify through a confidential bid. Potential service providers discretely submit to the buyer—in this case FONAFIFO—the price they would accept to enroll in the PES program (Stoneham et al., 2003; Latacz-Lohmann and Schilizzi, 2005; Salzman, 2005; Sierra and Russman, 2006). Reverse auctioning provides several clear benefits: it can prevent collusion and bidding up of prices among landowners; it is well-suited to monopsonies; and it can reduce, though not eliminate rent seeking (Chomitz et al., 2006) by reducing information asymmetry between the ecosystem service provider and buyer. Reverse auctions have been used in Australia's BushTender program (Stoneham et al., 2003; Salzman, 2005) and the U.S. Conservation Reserve Program. When the coordinating agency matches areas of greatest benefit and lowest cost, efficiency in

ecosystem service provision is maximized. The buyer then accepts bids up to a budget threshold, service provision level, or cost-benefit ratio.

Latacz-Lohmann and Schilizzi (2005) show evidence from experiments, models, and real-world PES data demonstrating substantial gains in total ecosystem services provision on a fixed budget by targeting services and auctioning versus paying a fixed price. Efficiency gains may not be universal or measurably positive from auctioning, and can shrink over time as landowners learn how to strategically bid (Latacz-Lohmann and van der Hamsvoort,1997; Latacz-Lohmann and Schilizzi, 2005). Careful auction design and selective information disclosure by the buyer are necessary to maintain efficiency when auctions are repeated over time. Adaptively testing spatial targeting and auctioning methods in different parts of Costa Rica could help determine their feasibility and utility while advancing the state of knowledge about efficient, fair, and sustainable PES systems. Currently, the GRUAS reports and other efforts provide for basic spatial targeting. Yet once superimposed, target areas cover 70% of the country, confounding spatial prioritization and meaningful clustering of PES properties (Sills et al. 2005). For many services, protecting adjacent land offers synergistic benefits. Designing proper incentives for multiple-landowner coordination is an important challenge for the Costa Rican PES systems (Latacz-Lohmann and Schilizzi, 2005) and may include allowing groups of adjacent landowners to bid together on conservation contracts in an auction format.

**Socioeconomic Objectives**

The potential exists for synergy between rural development and conservation goals, yet the relationship between PES and socioeconomic objectives is still largely uncertain. Costa Rica's PES program aims "to benefit and augment the quality of life for rural populations whose lands possess forest or the potential for forest cover through silviculture" (Article 1 of Ley 7575). Though the law itself targets rural populations, evolution of the political discourse has re-framed

the issue to center on whether PES positively benefits "the rural poor." At the national level, the majority of PES participants are small and medium landholders (Sills et al., 2005). Nonetheless, regional studies found that PES participants tend to have higher off-farm incomes, larger properties and higher levels of education than otherwise equivalent non-participants (Zbinden and Lee, 2005). Furthermore, a recent qualitative study in the central cordillera region found that the income generated from PES is used by the majority of poor participants for routine household expenses, precluding its application to longer-term savings or sustained quality-of-life investments (Esposito, in prep).

Thus, PES is probably contributing very little to enhancing the economic well-being of the poorest of Costa Ricans, since participants are on-average not among the poorest landholders; and even poor landholders are likely better off than the landless poor. Yet economic well-being is not equal to quality-of-life. Whether earning income from PES or not, the poorest and richest landholders alike benefit from greater landscape levels of environmental services afforded by the program. To-date, no indicators have been designed or measured for the latter in Costa Rica with regard to enhanced provision of ecosystem services through PES. Initiatives are underway to implement a PES-impact-monitoring system to better understand the degree to which socioeconomic objectives are being met (World Bank 2006). Perhaps a germane question at this juncture is, in precisely what ways does Costa Rica hope PES benefits the rural population and what are the relative priorities (income, capacity-building or ensuring a healthy, safe environment)?

Several modifications over the last decade have attempted to facilitate smallholder participation. In key impoverished regions, FONAFIFO makes exceptions to the need for legal title when submitting PES applications provided that landholders meet certain requirements (see

Decree 31633, La Gaceta 29, 2-11-04). Transaction fees have been reduced, though not eliminated, by allowing smallholders to form associations and enter PES "in bulk." FONAFIFO has streamlined their information system with other government databases to facilitate verification of requirements (Pagiola, 2006). However, the degree to which these measures have facilitated access remains un-quantified. That there is no shortage of willing participants demonstrates that PES is clearly attractive to a sufficient number of landholders. Yet this confounds the ability to better understand achievement of PES-related socioeconomic objectives, or even how such objectives should be defined.

Despite debate in development literature about appropriate tactics for ameliorating poverty, there is consensus that financial assistance alone will not yield success. Rather, a combination of investments in health services, education and infrastructure is essential (e.g., U.N. Millennium Village projects). Costa Rica has long-been recognized for its extensive social services and emphasis on education. FONAFIFO is gearing up to increase collaboration with civil society to enhance outreach and capacity-building for marginalized groups (World Bank 2006). If PES is to better the quality-of-life for the rural poor, perhaps an explicit, formal design linking PES participation with these broader well-being institutions and mechanisms is needed in the next phase of implementation.

## Conclusions

Land use change has significant ecological impacts, and is second only to electricity/heat generation as a source of global greenhouse gases (Baumert et al., 2005). To address forest-related land use change, Costa Rica implemented a novel, market-based conservation strategy. PES—coupled with a long-standing commitment to address deforestation and biodiversity erosion—has substantially transformed the externalized values of forests. Costa Rica designed a conceptually-sound PES finance mechanism and set an example for other countries to follow. In

fact, PES land now rivals the much-lauded Costa Rican national park system in area, illustrating how significantly this strategy affects private-property land use.

We reviewed Costa Rica's experience with PES; the lessons learned range from logistical to scientific. Even when scheme design is sound (e.g. Costa Rica's reliance on a polluter pays-inspired fuel tax to fund PES), implementation can fall short of the intended policy (e.g., co-optation for other uses). This case study illustrates that many challenges arise for PES schemes due to the complexity of working with large numbers of diverse stakeholders in an ever-changing economic context. We have made suggestions regarding how Costa Rica's scheme design might be enhanced (Table 4.2) as it embarks on a new phase of PES implementation through the "Mainstreaming and Scaling Up PES" project with World Bank support. These include decoupling the financing of monitoring from the monitoring itself, strategically targeting PES land for both ecological and social objectives, and laddering contracts over different time spans to enhance the continuity of ecosystem service provision. While each of these changes offer benefits and drawbacks, their careful consideration and use can promote future PES-based conservation in Costa Rica while providing valuable lessons for emerging programs.

Rodriguez et al. (2006) stress the need to critically consider tradeoffs resulting from competing ecosystem services. Ecosystem management institutions like PES schemes are influential in tipping this balance. Our review of Costa Rica's PES program highlights institutional design tradeoffs affecting the nature, amount and geographic arrangement of ecosystem service provision. For example, larger PES contracts are advantageous for institutional efficiency and for meeting ecological scale-dependency in ecosystem service provision. Yet this translates into fewer PES contracts and diminished program equity. Another example entails allocation of PES contracts across production and conservation modalities of

PES. Forestry initiatives may contribute to rural development and relieve timber pressure on natural forests, while protection of natural forest generally yields a greater bundle of ecosystem services. FONAFIFO's institutional forestry bias is arguably appropriate for the current mix of modalities, but may be inadequate to administer modalities added in future iterations of the PES program. This underscores the importance of feedback mechanisms in PES design so that the institutional arrangements may evolve appropriately.

One critical tradeoff in Costa Rica's current PES scheme design occurs between maximizing ecosystem services and achievement of socioeconomic objectives. Providing socially-optimal levels of ecosystem services and raising the quality-of-life for the rural poor are both components of the PES program. To manage this tradeoff consciously, Costa Rica might explicitly define quality-of-life indicators and implement ecologically-rigorous spatial targeting criteria, as we have suggested here. We have also identified steps to improve program efficiency—conserving more land through tiered payments that spatially target areas of high ecosystem services, combined with reverse auctioning to conserve land at a rate consistent with landowners' opportunity cost. Reverse auctioning can reduce rents to landowners, yet for the poor, such rents could constitute valuable supplemental income. Alternatively, if benefiting the poor is the program's primary goal, PES would likely achieve lower overall levels of ecosystem service provisioning. Ecological economics seeks a sustainable economic scale, fair distribution of resources, and efficient allocation (Daly and Farley, 2004). These goals are typically ranked in that order with the understanding that maintaining justice and efficiency is impossible in the absence of sufficient natural capital to support the human economy. While some level of "win-win" may be possible between the numerous tradeoffs that PES entails, a more decisive PES design is required.

The tradeoffs we highlight do not represent design flaws per se. Rather, they are inherent elements of any PES system and serve as junctures for critical decision-making on the part of the implementing agencies and supporting constituencies. We have pointed out the achievements and challenges in Costa Rica's present PES scheme, providing insight useful to other programs in evaluating PES design choices. As this case study illustrates, PES is a pliable conservation tool that can be molded to fit specific contexts and meet certain objectives; but tradeoffs should be anticipated and dealt with both a priori and iteratively for the long term success of environmental service provision. Viable PES schemes hinge on innovation and Costa Rica is well-positioned to begin pilot testing some of the more-nuanced design elements we have proposed here.

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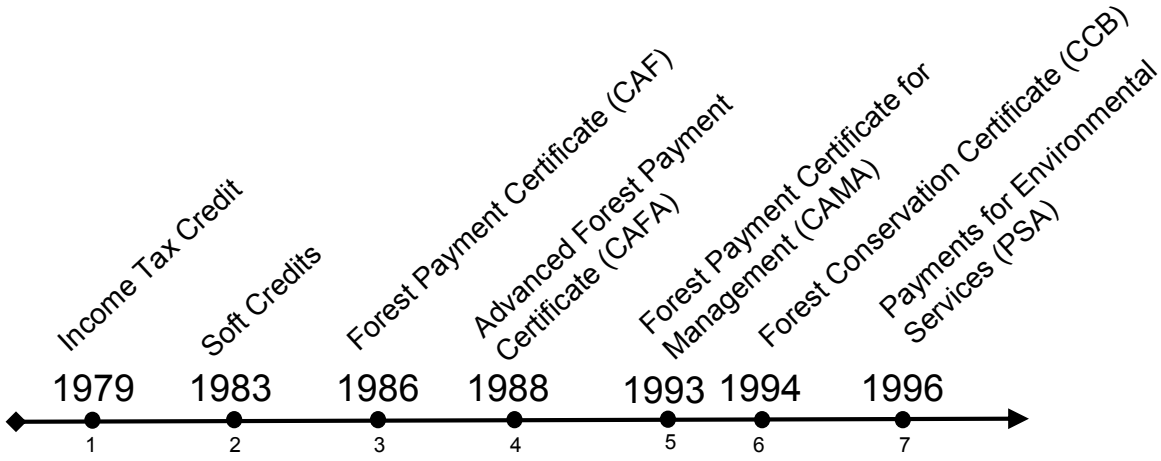
Table 4-1. Legal status of PES modalities over the last decade. The legal citations listed in parentheses are referenced in the citation column on the right.

| Modality | Status | Criteria | Payment | Priority | Citation |
|---|---|---|---|---|---|
| Forest Protection | Dates from adoption of Forest Law 7575 (1996) to present (1) | (2) 3.4. Between 2 and 300 hectares enrolled (2) 3.6. Maximum 600 hectares (within indigenous areas) | (3) 2(a) $64 per hectare per year (provided over five year period and renewable) | (2) 2.2.1. SINAC biological corridors; 2.2.2. Existing biological corridors; 2.2.3. Protection of AyA hydrologic resources; 2.2.4. Unpurchased protected areas; 2.2.5. Locations in cantons with MIDEPLAN Social Development indexes lower than 40% | (1) Ley Forestal Nº 7575, publicado en *La Gaceta* 72 del 16 de Abril del 1996.<br><br>(2) Reglamento Nº 9, Manual de Procedimientos para el pago de Servicios Ambientales, FONOFIFO, publicado en *La Gaceta* 51 del 13 Marzo del 2007. |
| Reforestation | Dates from adoption of Forest Law 7575 (1996) to present (1) | (2) 3.1. Between 1 and 300 hectares enrolled; 3.2. Maximum 50 hectares enrolled; 3.3. Minimum 50 hectares enrolled. | (3) 2(b) $816 per hectare over ten-year period | (2) 2.1.1. "High potential" forest plantations; 2.1.2. Areas with threatened species; 2.1.3. Pastures defined as Kyoto lands; 2.1.4. Projects under natural regeneration for at least one year | (3) Decreto Ejecutivo Nº 33226, MINAE, publicado en *La Gaceta* 141 del 21 de Julio del 2006.<br><br>(4) Reglamento Nº 0, FONOFIFO, publicado en *La Gaceta 1*51 del 8 Agosto del 2006. |
| Natural Forest Regeneration | Dates from first mention in 2005 to present (5,6) | Minimum 2 hectares enrolled (2) 3.5. | (3) 2(c) $41 per hectare per year (provided over five year period and renewable) | None specified | (5) FONOFIFO, http://www.fonafifo.com/text_files/servicios_ambientales/montos.pdf, (2007) |
| Agro-forestry Systems | Dates from 2003 to present (5) | (2) 3.7. Min. 350 trees, max. 3500 trees per participant; 3.8. Maximum 336,000 trees per joint project, cooperative or indigenous reserve; 3.9. Specific requirements per hectare and square km. | (3) 2(d) $1.30 per tree (provided over three year period) | (2) 2.3.1. Projects with organizations with FONFOFIFO agreements; 2.3.2. Land as described in *(1)Ministerio de Agricultura y Ganadería. 1995. Metodología para la Determinación de la Capacidad de Uso de las Tierras de Costa Rica. San José, Costa Rica. 60p.* 2.3.3. Areas with specific agreements with FONOFIFO | (6) Reglamento Nº 2, FONOFIFO, publicado en *La Gaceta* 26 del 7 Febrero del 2005.<br><br>(7) Decreto Ejecutivo Nº 30478 MINAE, publicado en La Gaceta 112 del 12 Junio del 2002. |
| Forest Management | Dates from adoption of Forest Law 7575 (1996) until 2002 (1, 7). | Criteria determined by conservation area (8) 5.1-5.10 | (8) ¢123,540 (or about $343) per hectare (provided over five year period) | (8) 5.1-5.10 Priority determined by conservation area (SINAC) | (8) Decreto Ejecutivo Nº 30090, MINAE, publicado en *La Gaceta* 32 del 14 de Febrero del 2002. |

Table 4.2. Summary of issues and recommendations for Costa Rica's PES program.

| Identified Issue | Category | Action | Benefits | Challenges |
|---|---|---|---|---|
| Institutional forestry bias | Program administration | Change internal structure of FONAFIFO's board; involve other environment-related professionals in PES design & monitoring | "Forestry-sector bias" has helped develop small plantations, reducing pressure on forests | "Forestry-sector bias" provides equal funding for plantations, which do not provide the ecosystem services of natural forests |
| Monitoring too closely tied to compensation | Program administration | Have multiple regentes monitor a contract over its lifetime (e.g. random assignment each year) | More transparent reporting of success & failure | Need to develop feasible way to re-structure regente payments from the proposed "pooled fund" |
| Amount of payment | Opportunity cost | Redefine payment amounts to be more sensitive to economic fluctuations; consider "contract laddering" | Possibly greater perception of fairness to landowners, and ensure longer-term service provision | Difficulty of PES competing with high-value land uses; higher payment levels could mean less total land and ecosystem services enrolled |
| Free riding and identifying long-term funding sources | Equitable financing; Sustainable financing | Eliminate free riding by international & local beneficiaries; move toward more sustainable financing sources | Fairer & more sustainable program financing; being implemented at national level | Lack of political will to require fair international payments for global ecosystem services |
| Level of service provision variable and loosely linked to modality | Bundling, Spatial targeting | Target payments to areas of high ecosystem service values, differentiate payments based on services delivered, modalities, regions of the country | Could improve program efficiency and total ecosystem service provision | Could reduce participation by the poor; fair measurement and payment systems must be developed |
| Level of service provision variable and loosely linked to modality | Spatial targeting | Reverse auctioning to identify opportunity costs for landowners to participate | Could improve program efficiency and total ES provision | Could reduce participation by the poor; auctions must be carefully designed to avoid strategic bidding |
| Poverty alleviation debatable goal of PES | Socioeconomic objectives | Reduce transaction costs and other barriers to entry for poor households | Greater participation for poor | May reduce total delivery of ecosystem service benefits |
| Poverty alleviation debatable goal of PES | Socioeconomic objectives | Further research into why poorer households do not participate in PES, and if PES is appropriate for poverty alleviation | Better identify potential barriers to entry for poor households | Requires rigorous and political will to act on recommendations |




[1]Decree No. 10521-AH, Sept. 1979. Income tax credit given to land owners involved in reforestation activities to offset the cost of plantations. The concept was to promote plantations as a way of alleviating deforestation pressure on natural forests. This tax credit targeted large landholders since small holders generally did not pay income tax.

[2]COREMA-AID project. International funding helped to finance low-interest reforestation loans with long grace periods and extended repayment windows. This initiative was the first of several soft credit incentives, some of which still continue in the present (e.g. FONAFIFO-brokered loans for reforestation).

[3]Article 82 of the Second Forestry Law (No. 7032, La Gaceta 13; Circulo 84 – May 6, 1986) creates the Certificado de Abono Forestal (CAF). Reforestation investments in plantations are made up front by land owner and compensation is given later through a tradable tax voucher.

[4]Decree No. 18691-MIRENEM-H, Dec. 1988. Like CAF but compensation is given prior to reforestation investment so that land owners with less capital could participate.

[5]Decree No. 22452-MIRENEM-H, 1993 (La Gaceta 170, Alcance 6). Established that scientifically-managed timber extraction from natural forests would be eligible for tax vouchers.

[6]Decree No. 23101-MIRENEM-H, April 1994 (La Gaceta 74). Established that tax vouchers could be paid for natural forest protection (equal to the CAF vouchers paid for reforestation).

[7] Fourth Forestry Law (No. 7575, Gaceta 72, Alcance 21 – April 16, 1996). Article 22 affirms continuation of tax vouchers for protecting natural forest, along with other tax benefits. Article 24 provides that land owners voluntarily allowing forest regeneration are eligible for the same benefits. Article 29 details tax benefits for plantation owners.

Figure 4-1. Timeline detailing the evolution of PES in Costa Rica.

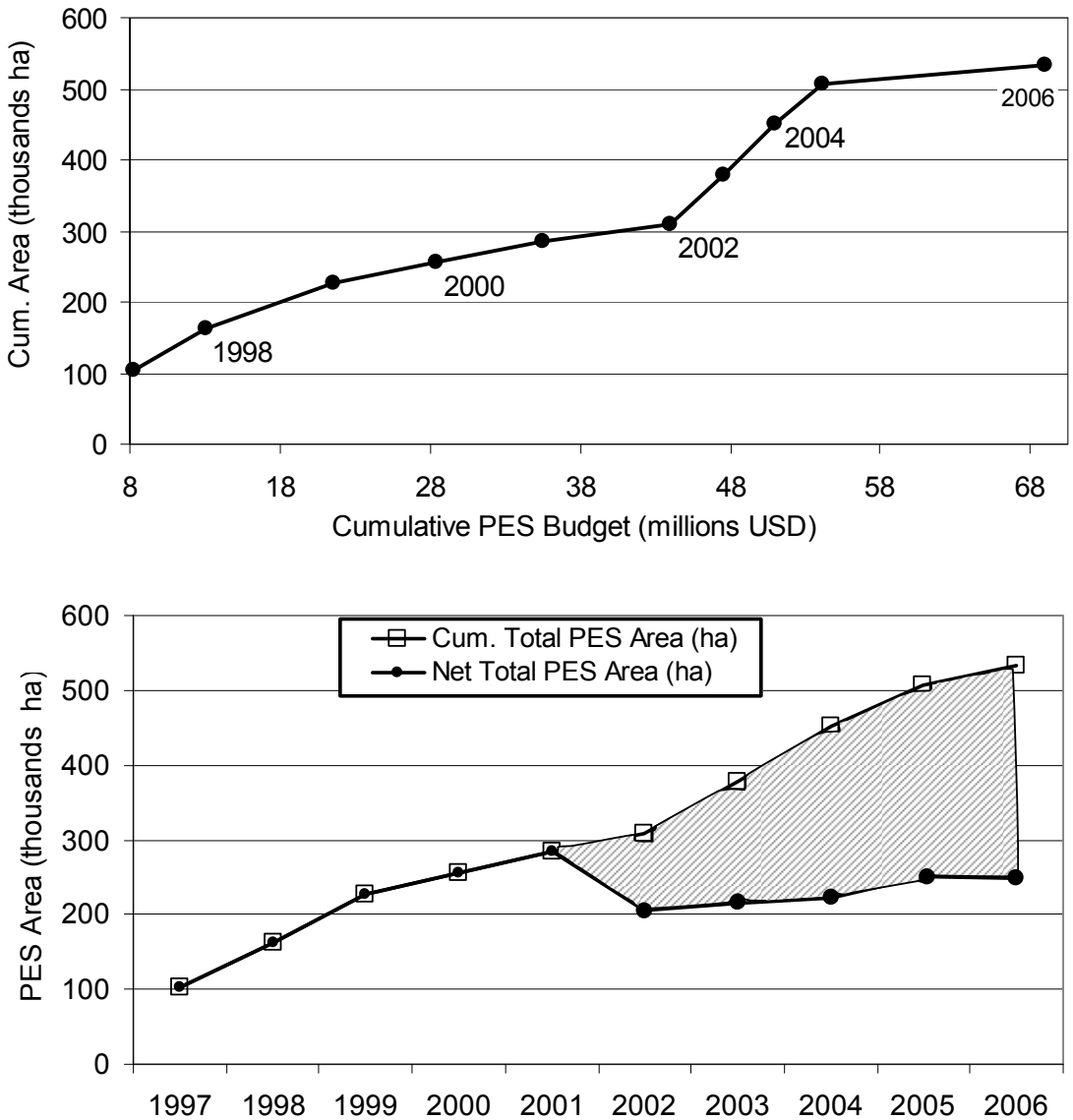



Figure 4-2. (a) Cumulative area in PES (thousands of ha) as a function of the cumulative budget (millions USD) FONAFIFO receives to implement PES (b) Time series of cumulative PES area (thousands of ha) recruited across all modalities (square) in contrast with net area where expired contracts are subtracted off of the running sum (circle). The shaded region between the two curves represents the difference between the best and worst case PES implementation scenarios respectively where 100% of PES forest is conserved even after payment period ends (defining upper limit of region) or where none of the contracted forest is conserved after payments end (defining the lower limit of region). PES scheme design is critical in determining where the empirical curve falls between these two extremes.

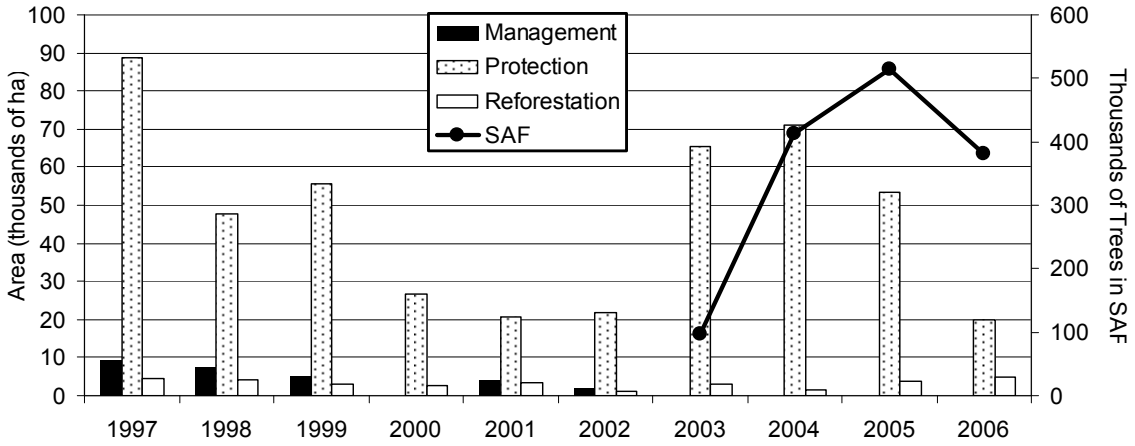



Figure 4-3. Time series of the recruited area per modality and number of trees in the agroforestry modality (SAF).

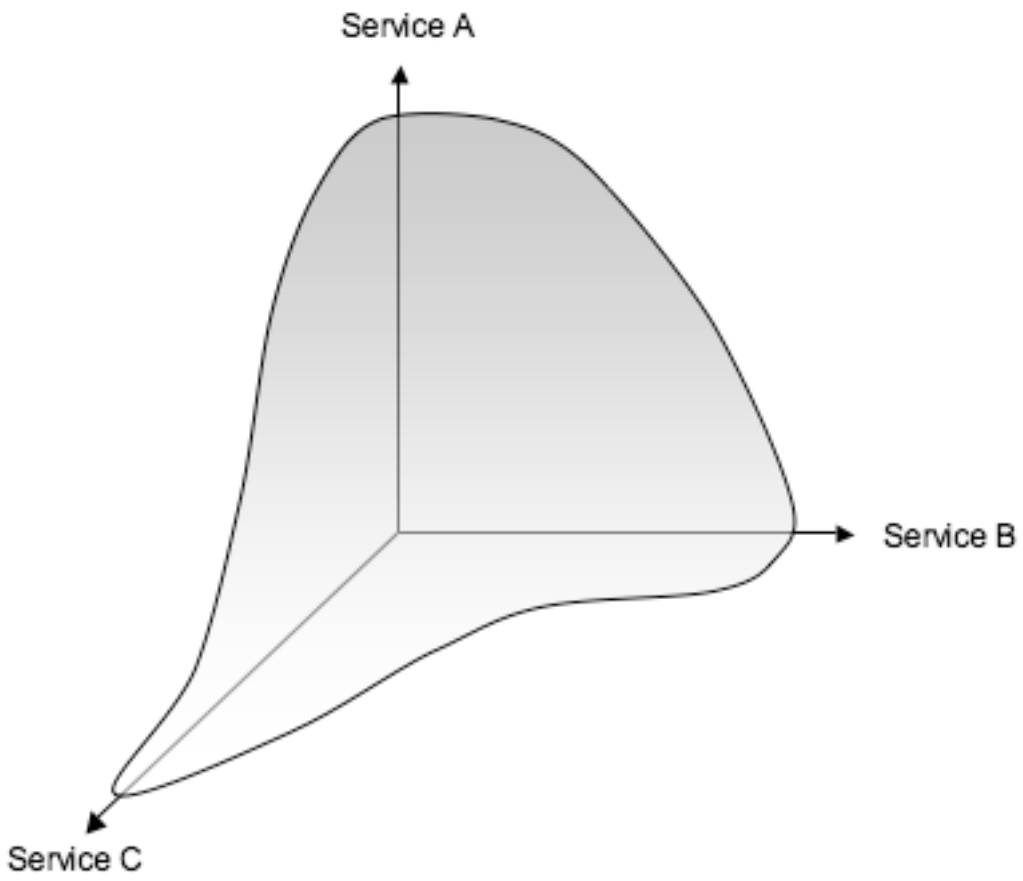



Figure 4-4**.** A conceptual multi-dimensional production possibility frontier for ecosystem services. Bundling ecosystem services entails maximizing the volume defined by provision levels of interacting services.

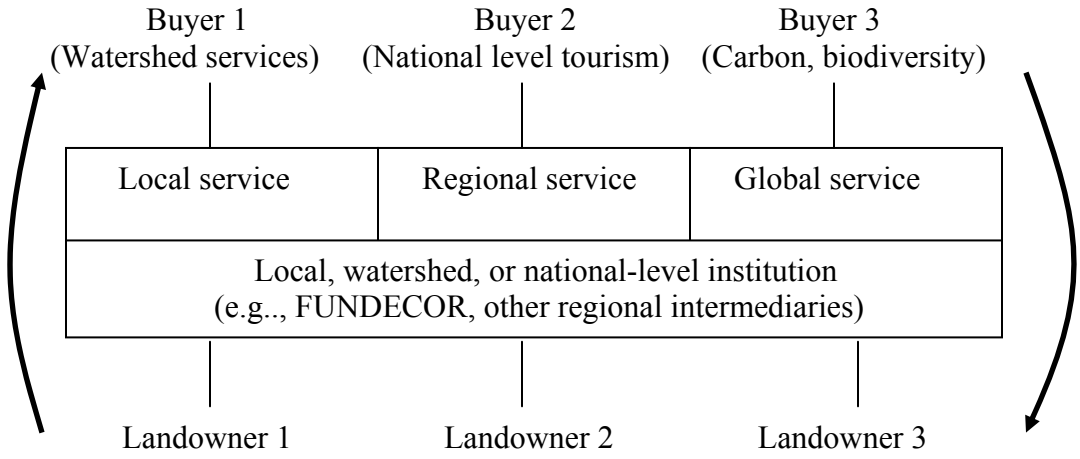



Figure 4-5. Costa Rica's model for using institutions to bundle services linking buyers and sellers across different spatial scales.

| Funding from | Internal | Internal | External | Internal | Internal | External | External |
|---|---|---|---|---|---|---|---|
| Revenue source | Tax/fee on good or service with inelastic demand | Tax/fee on good or service with elastic demand | Compulsory payments under international treaty | Voluntary payments | Funding from general budget | Grants or voluntary payments | Loans |
| Example | Tax or fees on energy or water users | Tax or fees on luxury goods or tourism | Carbon credit or biodiversity purchases under strong international treaties | Payments from firms (e.g., Cerveceria Costa Rica) | General appropriation subject to renewal | GEF grants; Norway's pre-Kyoto carbon credit purchase; U.S. voluntary carbon markets | World Bank loans |

Higher — Moderate — Lower

Degree of sustainability/security

Figure 4-6. Relative sustainability/security of financing mechanisms for PES

# CHAPTER 5*
# FOREST EXPANSION IN NORTHWEST COSTA RICA: CONJUNCTURE OF THE GLOBAL MARKET, LAND-USE INTENSIFICATION AND FOREST PROTECTION


## Summary

Though widely documented in developed countries, understanding of how and whether forest transitions occur in the developing world is limited. This research entails a spatially-explicit, remote-sensing-based analysis of land cover conversion trends in northwest Costa Rica to explore the mechanisms driving the observed forest expansion. I assess the physical and socioeconomic landscape setting for dominant conversion patterns in light of broader conservation and development initiatives, along with relevant economic trends, in order to propose a conceptual model accounting for forest expansion. Results demonstrate that conversion processes are interrelated and characterized by unique landscape niches. Agricultural intensification occurred in lowland areas, facilitating reforestation of marginal grasslands in upland areas. Protected area establishment facilitated forest recovery. Yet forest conservation and regeneration were significant on private property as well due to the conjuncture of declining beef prices, agricultural intensification and revised forestry policies. Elements of this study provide support for aspects of the classic explanations for forest transitions while highlighting the limitations of forest transition theory to account for observed reforestation trends.


## Introduction

Changes in forest cover have consequences for biodiversity, climate change, and water resources. For many years, the focus of land-cover change research in the tropics centered on deforestation processes (e.g., Skole et al. 1994, Lambin 1997). Approaches to studying and quantifying deforestation, however, have sometimes belied the complexity of forest cover

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changes. Forest cover dynamics may be nuanced, multi-directional and exhibit varying degrees of reversibility. Forest degradation may precede outright deforestation (World Bank 1995) or mask long-term, qualitative forest change where net forest cover remains stable (de Jong et al. 2001); secondary re-growth may keep pace with concomitant deforestation (Ramankutty et al. 2007), making "rates" of forest-cover change misleading oversimplifications. Soil degradation that ensues in the wake of intensive non-forest land use may preclude near-term reforestation (Carpenter et al. 2004). Alternatively, forest transitions may occur in which long-term declines in forest cover halt, followed by an enduring expansion of forested area (Mather 1992). The drivers of these respective processes of forest change are often distinct (Grainger 1995), but sometimes overlap or interact (Stern et al. 1992).

Proximate and ultimate drivers of deforestation have been identified through historical-comparative studies and synthetic meta-analyses (Geist and Lambin 2002). Similarly, the concept of forest transitions has emerged to explain how and why forest expansion through secondary regrowth occurs (Mather 1992). The concept holds that economic development drives off-farm migration and urbanization, contributing to spontaneous forest regeneration on agricultural lands due to land abandonment and a lack of labor (Rudel et al. 2005; Rudel, this volume). An alternative, though not mutually exclusive, pathway to the forest transition posits that scarcity of forests and forest-derived products, along with changing social norms promoting "improved" land management, may motivate reforestation through tree planting (Rudel et al. 2005) and/or the establishment of forest reserves (Perz 2007).

Studies from developed, industrial countries yield evidence to support these mechanisms of forest transitions (e.g., Foster 1992, Andre 1998, Staaland et al. 1998, Mather, Fairbairn and Needle 1999). Questions abound, however, regarding how relevant developed-world

experiences and observations are in developing countries (Koop and Tole 1999). Perz (2007) highlights the need to incorporate other contextual biophysical and institutional explanations of forest expansion if the forest transition concept is to prove meaningful in the developing-world forest management and conservation dialogue. National studies with aggregated data are unlikely to address important questions about regional developing-country forest change. Sub-national case studies like this are critical to understanding the relationship between development and forest cover change (Klooster 2003).

In the region of Mexico and Central America, forest cover dynamics are complex with evidence of forest recovery in some places (Bray, this volume). Costa Rica is a good case in point since it has undergone net forest expansion in recent years. The country experienced centuries of forest loss, most pronounced from 1950 to the mid-1980s, and had one of the highest deforestation rates in the world at 3.9% per year during this period (Leonard 1986). The inflection point in Costa Rica's forest-area curve occurred in the late 1980s (Kleinn et al. 2002), representing an opportunity to study the process of reversing the country's net forest loss to a trend of forest expansion Over the last several decades, Costa Rica built a world-renowned protected area network covering over a quarter of the national territory. The protected area network and other environmental protection policies undoubtedly play a vital role in abating deforestation and facilitating forest regrowth. Yet forest expansion does not appear to be limited to protected areas (Daniels 2004, Arroyo-Mora et al. 2005), suggesting that other important processes merit further analysis.

In this study I focused on identifying dominant land cover conversion sequences that explained net forest expansion in the Tempisque Basin of northwest Costa Rica. I analyzed the physical landscape setting, along with the socioeconomic and policy context for these

conversions to explore the factors that drove forest expansion in the landscape from 1975 to 2000. This time period was chosen in light of the differences in data availability for important factors related to forest cover dynamics like forestry incentives and a more recent program that pays for forest ecosystem services (see **Forces of Landscape Change** section for details). I quantified the contribution of forest in protected and non-protected areas to net land-cover changes and to the landscape's dominant trajectories. This allowed me to partly disentangle the effect of protected area establishment from other processes driving forest expansion in order to develop a conceptual model of forest recovery.

By contrasting this model with explanatory mechanisms posited by the forest transition concept I examine how well the latter accounts for observed forest expansion in the Tempisque Basin. Many of the significant studies on forest transitions fail to explicitly consider the biophysical landscape setting and pattern of forest expansion, or how these factors interact with the land tenure and policy arena (e.g., Mather 1992, Mater 1998, Rudel et al. 2005). Hence, my approach in this case study contributes to the development of a nuanced forest transition concept that is more robust in the developing-world context.

## Tempisque Basin: Geographical and Historical Setting

Costa Rica's Tempisque Basin lies in the northwestern province of Guanacaste, comprising 10% of the national territory (5,404 $km^2$, Figure 5-1). Mean temperature is 27.5$^{o}$C and an annual mean precipitation of 1817 mm falls between May and November (Mateo-Varga 2001). The Tempisque River runs roughly north to south through the center of the basin, increasing in volume in the southern watershed with the confluence of the Bebedero River and several other important tributaries. Thirteen Holdridge life zones (Holdridge 1967) are found in the basin, along with myriad habitats like tropical dry forest, moist pre-montane forest and vast

seasonal wetlands. The Cordillera Guanacaste defines the eastern border of the watershed, reaching over 2000 m in elevation.

With more productive soils and more readily-cleared vegetation than the rainforest-covered regions of the country, the Tempisque Basin is the only part of lowland Costa Rica to be continuously inhabited since the beginning of the colonial era (Peters 2001). Significant forest clearing had taken place in Guanacaste by the seventeeth century (Boucher et al. 1983). The Tempisque Basin is thus hardly a contemporary agricultural frontier. Since European arrival cattle ranching was the primary cause of deforestation by generating demand for grazing land; timber was a secondary motive. Land clearing was reinforced by settlement policies that made partial forest clearing integral to obtaining formalized land rights.

Certain physical constraints and socio-economic characteristics of the Tempisque Basin played an integral role in calcifying the extensive cattle-ranching land use system that dominated the region for centuries. Because of the long, harsh dry season, transhumant grazing was adopted where cattle were herded upslope to cloud-bathed, evergreen pastures during the dry season. Alternatively, herds were moved down slope to graze in the green floodplain of the Tempisque River. This extensive pattern of low-intensity resource management was evidenced by the basin's land tenure patterns: thirteen ranchers held 11,000 ha or more; one rancher alone controlled nearly 134,000 ha (Edelman 1985) or about a quarter of the basin. Minimal investment was made in land and thus production per unit area remained low for these *latifundios*. Beef yields, for example, averaged about 168 kg/ha, relative to a possible yield of 273 kg/ha (Edelman 1985).

Agricultural census data shows that ranch size and output efficiency were negatively correlated in northwest Costa Rica, suggesting that land concentration was as much speculative

as it was production-oriented (Taylor 1980). This begs the question of why and how *latifundios*, a key element of a seemingly irrational land use system that is intimately tied to forest-cover patterns, persisted for so many centuries without intensifying production. Edelman (1992) details the logic of Guanacaste's *latifundios*, highlighting the importance of institutional rent (see de Janvry 1981) including subsidized production and highly-favorable credit conditions.

Favorable credit was facilitated by the powerful cattlemen lobby's connections to the nationalized banking system. With extensive landholdings as collateral, ranching operations were as much about concentrating credit, often invested in enterprises other than ranching, and about harvesting subsidies as it was about cattle production (Taylor 1980). Since land was passed down through families, sometimes dating as far back as initial land grants, the cost of land was often not a consideration in land-use decision making (Edelman 1981).

The pattern of land cover and tenure resulting from the persistence of this extensive land use system in the Tempisque Basin (up until 1970s) superficially resembled that of a "hollow frontier". The hollow frontier model attempts to explain the persistence of deforested land despite population declines. It posits that settlers clear land for agriculture, degrade soils through poor land use management, and then abandon crop cultivation plots to move further into the forest (Preston 1959, Casetti and Gauthier 1977). Land concentration occurs in the wake of this out-migration to the frontier, or in recent years for some places, to urban centers (Browder and Godfry 1997). Despite depopulation, forests fail to regenerate because the land use adopted by the large landholders, generally cattle ranching, requires little labor to stay in production.

While similar to a hollow frontier in appearance and with regard to the pattern of land tenure, a key distinction for the Tempisque Basin is that land concentration largely *caused* out-migration rather than resulting from it. Low-intensity cattle ranching in the area required about

six worker days per hectare per year (Taylor 1980), encouraging the highest negative migration rates in the country (-22%, Universidad de Costa Rica 1976).

## Forces of Landscape Change

Several forces aligned to set the context for landscape change in the Tempisque Basin. Completion of the Panamerican Highway in the 1950s facilitated land-based connectivity to national markets (Peters 2001). In 1953 the cattlemen's lobby effectively pushed for legislation permitting the export of live cattle (Edelman 1985), later followed by beef once the first packing house was established. Sugar cane cultivation was promoted in the Tempisque lowlands after implementation of the 1959 US-Cuban trade embargo. By the late 1970s, Costa Rica was the fourth largest supplier of beef to the U.S. (Peters 2001).

The average price for beef on the international market was $2.37/kg during the 1970s; but in the early 1980s the price of beef on the international market fell precipitously (1985-1999 mean price was $1.36/kg, Arroyo-Mora et al. 2005). At the same time, interconnected with the global recession, Costa Rica experienced an acute economic crisis while having one of the highest *per capita* debt loads in the world (Harrison 1991). Interest on Costa Rica's external debt rose 15% in a short time and the nation defaulted on international loans (Lara 1995). In the structural adjustments that ensued at the behest of international lending institutions, Costa Rica reduced public spending, including subsidies for cattle production.

In the mid-1970s, construction began for Costa Rica's largest hydroelectric plant just outside the eastern border of the basin, which now generates a quarter of the nation's electricity (capacity 372 MW)(DeWitt 1977). After electricity generation, water began being channeled to the lowlands of the Tempisque Basin in the late 1980s via some 234 km of canals in a project known by its Spanish acronym, PRAT (*Proyecto Riego Arenal Tempisque*). The government redistributed land outside of the eastern floodplain of the Tempisque River in 5 ha plots for

flooded rice cultivation using water from PRAT. Otherwise, irrigation water is distributed to existing agro-industry operations, which was already extracting water from the Tempisque River and the underground aquifer.

In addition to providing much-needed electricity for economic and social development, PRAT greatly affected land use potential in parts of the basin. Year-round cultivation became possible with steady irrigation, increasing yields and profitability. This encouraged land-use intensification at a time when cattle ranching was hardly viable. Irrigation raised yields from 3 to 8 MT/ha for rice and from 8 to 14 MT/ha for sugar cane (Amador et al. 2003). Furthermore, crop diversity expanded to include cantaloupe, watermelon, sod, and occasionally vegetables like onion and bell pepper.

By the 1970s, the establishment of protected areas also acted as a force of landscape change by protecting many remaining forested areas and facilitating forest regeneration in others. It was the Forestry Law of 1969 (no. 4465) that created a legal mechanism for establishing protected areas through executive decrees. Since then, the park system has grown in area and morphed through various administrative structures (Boza 1993, Campbell 2002), having many implications for national development. Parks are often the centerpiece for tourism development at the local and regional levels, which has generally helped garner public support for conservation. The shift of Costa Rica's historically-agrarian economy toward a service economy related to tourism was highlighted in 1994 when the tourism industry surpassed all other sectors of the economy in earning foreign currency (Brockett and Gottfried 2002).

In the Tempisque Basin, the first national parks were established in the early 1970s, save a part of Santa Rosa first protected in the mid-1960s. By 2000, fourteen protected areas that at least partly intersect the basin had been established across multiple categories including national

forests and an experimental reforestation reserve (Figure 5-1, Table 5-1). These areas comprise 47,340 ha or 9.5% of the basin. While Costa Rica's protected area system does include several inhabited indigenous reserves, none fall within the Tempisque Basin. Some area incorporated into parks in the basin, like parts of Guanacaste National Park and Palo Verde National Park, had long been used as pasture. These areas were left to naturally regenerate or were managed under experimental and assisted forest regeneration regimes (Janzen 2002).

A final, critical influence relevant to land use and forest cover entailed a series of revisions to the forest regulatory framework. From 1969 to 1994, an evolving series of tax-credit-based incentives was developed to encourage reforestation and forest protection on private property. These incentives were not universal, but rather implemented by landholder application upon management plan approval. Results of tax credits were not always intended, particularly in the early years, since natural forest could be cleared and replaced by plantations to receive the tax benefit. In 1996, a new forestry law (no. 7575) was passed transforming these tax credits into direct payments for compliance with one of several modalities of forest-based land use (see Daniels et al. 2009 for details). That is, landholders began being compensated for the sale of environmental services (i.e., payments for environmental services, PES). The new forest law also disallowed changes in forest land use; extant forest cannot be cut down even on private property (article 19) without a permit.

Implementation and monitoring of both the incentives and prohibitions provided for in Law 7575 is ever-evolving. Measuring the impact of the incentives on forest cover, particularly prior to the PES scheme created in the new forestry law, has been difficult. Records of properties that received tax credits were not spatial and program administration has changed multiple times through the years making it difficult to find reliable paper-based documentation.

Data for the first phase of PES (1997-2000) consist of GPS points taken within ~1 km from the actual contracted forest. Furthermore, forest contracted for PES may be divided into multiple, distinct patches. Only in 2006 did PES rules begin to require a map of the actual forest contracted for ecosystem service provision.

In this analysis, "non-protected landscape" refers to all land outside of public protected areas. This designation obviously under represents forest protection measures by not explicitly including forests conserved due to tax incentives, PES and forest protection afforded by article 19 of the 1996 forestry law. These latter forms of forest protection are dealt with implicitly to the extent that non-spatial data would allow. Also, the appropriate selection of image dates facilitates understanding of how the discussed factors may have impacted forest cover.

## Methods

This analysis used the classification of Landsat satellite imagery to quantify land cover changes that occurred from 1975 to 2000 in the Tempisque Basin and to determine net area gains or losses of major land cover classes. Single-date land cover classifications were compiled into a trajectory layer where the value of each pixel indicated the specific land cover conversion sequence that occurred in that specific location across the three dates of analysis. The biophysical and economic landscape setting was calculated for dominant trajectories to discern geographic patterns associated with particular conversion sequences. These generalized patterns were used to construct a generalized conceptual model of landscape change.

### Land-cover Classification and Change Detection

Four Landsat images were obtained to assess land cover dynamics over time in the Tempisque Basin. All images were acquired from the early dry season since cloud cover greatly obscures land surface features during the wet season. The first two images, 1974 (path 17, Row 53) and 1975 (Path 16, Row 53), were collected by the multispectral sensor (MSS) aboard the

earliest Landsat satellite. These images were stitched together as a mosaic to create a baseline land cover representation for the study region. Since 85% of the resulting mosaic was comprised of the 1975 scene, this mosaic is referred to as the 1975 image. The middle time point was a 1987 image from the Thematic Mapper (TM) sensor; the final date was a 2000 image collected by the Enhanced Thematic Mapper plus (ETM+) sensor.

Image dates were chosen to facilitate an understanding of the drivers of landscape change. The 1975 image represents the baseline landscape largely prior to protected area establishment or forestry tax credits, and when extensive cattle ranching was still thriving. The 1987 image coincides with the inflection of the national forest transition curve for Costa Rica, the beginning of PRAT-based irrigation, and after the crash of the international beef market. The 2000 image serves as a follow-up date, by which time all protected areas in the basin had been established, but prior to effects from the systematic application of the new forestry law (Ley 7575). Data availability for PES after 2000 lends itself to a spatially-explicit analysis since maps of properties were collected (though not the actual forest patches within properties). For this reason, the present study stops at year 2000.

Geometric rectification of the 2000 image used GPS-based ground-control points and root mean square error (RMSE) was below 0.5 or 15 m. Other dates were then co-registered to the 2000 image (RMSE $\leq$ 0.5). To ensure that data variance across the series corresponded to changes on the ground, rather than differences at the sensor level, all images were corrected for atmospheric haze, sensor bias and differences due to variations in solar angle across non-anniversary dates (Jensen 1996). These resulting calibrated data were unitless in the form of reflectance (albedo), the ratio of energy reflected by a surface to the energy received. To ensure uniform spatial extent across dates, pixels corresponding to clouds occurring in any image date

were eliminated from the other dates. The 1975 scene boundary was used in subsetting images to the watershed extent because its footprint truncated the northern extent of the watershed (by 110 $km^2$). These adjustments resulted in an extent of 5,153 $km^2$ (95.35% of actual basin).

Training samples for the 2000 classification were collected during fieldwork in 2001 (323 reference points). Historic photo-mosaics corresponding to the dates of the satellite imagery were used to generate training samples for the 1975 and 1987 image (393 and 368 reference points, respectively). Over half of the land cover training data were set aside for accuracy assessment. During fieldwork, 29 semi-structured land use history interviews were conducted throughout the watershed to construct agricultural calendars, verify image interpretation and better understand land-use systems and changes therein. Land cover classes employed were crop (*c*), wetland (*w*), grassland (*g*) and forest (*f*) (Table 5-2). Note that there are no natural dry grasslands and grassland cover thus generally corresponds to cattle ranching land use.

Land cover was independently determined for each image using a rule-based classification (Daniels 2006). This approach was developed because statistical clustering of spectral data alone failed to accurately identify the land cover classes of interest. The rule-based classification technique incorporated domain knowledge, spatial relationships and two distinct clustering algorithms. Overall classification accuracy exceeded 90% for each date and class-wise accuracy was $\geq$ 85% for each date. For details see Daniels (2006). Area for each land cover class was compared across the three-date image series to determine net-area changes across the study period. Also for each date, land cover inside and outside of protected areas was calculated for comparison.

**Trajectory Analysis**

Land-cover composition was assessed for each of the three dates to determine trends of net-area change (e.g. net increase in forest cover). These net-area trends then guided the

spatially-explicit analysis of land cover conversion sequences (i.e., what conversion processes accounted for the observed net changes in land cover).

Land cover across all three dates was compiled into a single raster layer of land cover trajectories where each possible sequence of land cover conversion was given a unique value. For example, a single pixel followed over the three original land-cover maps may have transitioned from grassland (g) to forest (f) from 1975 to 1987, and then remained forest through 2000. This trajectory was given a unique integer value and labeled "*g-f-f*" to indicate the sequence. For this analysis, a total of $4^3 = 64$ unique trajectories was possible. The area of each trajectory was calculated for protected and non-protected parts of the landscape by summing over the pixels in each of the two categories.

**Dominant Explanatory Trajectories**

Because of the relatively large number of possible trajectories (i.e., 64 classes) in a three-image time series, criteria were developed to determine the dominant trajectories in the landscape. If a particular land cover experienced a net-area gain from 1975 to 2000, the goal was to understand which trajectories accounted for the increase. For instance, for a net crop area increase over the study period, what was the predominant land cover converted *to* cropland that explains the net increase in crop area? More generally, for an increase in land cover *x*, the areas of all trajectories ending in *x* for year 2000 (explanatory trajectories) were compared. Similarly, if land cover *y* experienced a net area decrease over the study period, all possible explanatory trajectories starting with that land cover in 1975 were compared to understand what conversion processes explain the decrease in land cover *y* by 2000. Dominant explanatory trajectories were identified as those accounting for the majority of net-area change for a particular land cover (Table 5-3).

### Landscape Setting for Trajectories

Five raster layers were generated to provide spatial data for key physical and socioeconomic landscape gradients related to land-cover trajectories based on previous research (see Daniels and Cumming 2008). The resolution of each of these layers matched the pixel size for land-cover data (30 x 30 m). Layers included a digital elevation model (DEM), slope, distance to nearest road (d.rd), distance to large forest (d.forest), and distance to nearest population center (d.popctr).

The DEM (1 m vertical resolution) was obtained from the Organization for Tropical Studies and slope was computed from it. Public roads were obtained from the *Instituto Geografico Nacional* (IGN) of Costa Rica. GPS coordinates were recorded for the six main population centers accounting for over 95% of the population in the watershed. Large forests were defined as those on the first image with an area $\geq$ 50 ha. These patches were queried and isolated from the 1975 land cover map via an eight neighbor rule for patch definition. ArcView Spatial Analyst calculated distance surfaces. For dominant trajectories means ($\mu$) and standard deviations ($\sigma$) for each of the landscape setting variables were calculated.

## Results

### Land Cover Area: Net Trends

Net changes in land cover areas revealed the following trends in the basin: an increase in crop area, a decrease in wetland and grassland, and an increase in forest (Figure 5-2). Crop area increased 47,829 ha from 1975 to 2000, almost exclusively in the non-protected landscape (Figure 5-3) indicating that no threat of incursion on parks existed. Wetland area decreased by 7,019 ha. Nearly half of the wetlands remaining by 2000 fell in protected areas. By 2000, grassland had decreased to nearly half its 1975 extent, a loss of 141,102 ha. The proportion of

grassland falling in protected areas increased with each date, however, since land acquired for parks within the basin often required restoration from non-forest uses like cattle ranching.

For the 101,404 ha increase in forest, an increasing proportion fell within protected areas. While only 11 ha or 0.1% (Figure 5-4) of modern PES was invested in reforestation in the basin, this modality was the focus of all past iterations of forest incentives, save only Forest Conservation Tax Vouchers or CPB which was the direct pre-cursor of PES (year 1995 only). Unfortunately, reliable data are not available for estimating how much of this net increase in forest may be attributable to early reforestation incentives (Figure 5-4). Of the 256,708 ha of forest by 2000, nearly 12% (26,466 ha) fell within protected areas.

**Explanatory Trajectories**

The dominant trajectories accounting for the net area increase in cropland were grassland conversions (*g-c-c* with 14,290 ha and *g-g-c* with 16,116 ha, Figure 5-5). Only about one-fifth of total 2000 crop area had been cropland throughout (*c-c-c*), indicating a substitution of crop cultivation for former pastures with grassland cover. Net loss of wetlands was largely accounted for by conversion to cropland, with nearly four thousand hectares converted by 1987 (*w-c-c*) and half again as many by 2000 (2,335 ha for *w-w-c*). Wetland conversion represents a minor fraction of the total area converted to crop cover in the Tempisque Basin over the study period. Yet the proportion of wetlands lost through agricultural conversion (6,415 ha) is much greater, 21% of their original extent.

The importance of protected area establishment for wetland conservation (*w-w-w*) is evidenced by the fact that over half of conserved wetlands fell within protected areas (Figure 5-5) (see Daniels and Cumming 2008). Reforestation trajectories were dominant in explaining both net loss of grasslands and net increase in forest. A majority of grassland reforestation (77,224 ha) occurred from 1975 to 1987 (*g-f-f*) and over half again as much grassland (45,981

ha) reforested by 2000 (Figure 5-5). Protected areas played a more significant role in reforestation trajectories than in explaining other trends of land-cover change. Eight percent and 7.5% of trajectory area for *g-f-f* and *g-g-f,* respectively, occurred within protected areas, sizeable considering that parks comprise less then 10% of the basin area Forestry incentives were likely important too, though their contribution cannot be estimated directly (Figure 5-4).

Figure 5-6 illustrates that of the total 2000 forest area within protected areas, 16,102 ha (53%) of it was already forest in 1975 and remained forest throughout. This *f-f-f* trajectory in protected areas explained more of the 2000 forest area than grassland reforestation by 1987 (*g-f-f*; 6,211 ha or 20%) or by 2000 (*g-g-f*; 3,451 ha or 11%) combined. The forest conservation trajectory (*f-f-f*) explained 37% (83,384 ha) of non-protected 2000 forest area, compared with 50% for combined *g-f-f* and *g-g-f* trajectories (31% or 71,013 ha and 19% or 42,531 ha, respectively).

**Protected and Non-Protected Landscape Comparisons**

Results from comparing physical and socioeconomic landscape variables for protected and non-protected areas of the Tempisque Basin revealed that areas established as parks are different from the surrounding landscape (Table 5-4). Park land on average has a higher elevation (232 vs. 191 m) and on slightly steeper slopes relative to land outside of parks (12 vs. 10%). Protected areas also occur at greater distances from population centers (16.1 vs. 10.2 km) and roads (3.2 vs. 1.5 km). Protected areas pixels are, on average, substantially shorter distances from large forest patches (0.4 vs. 1.1 km).

**Landscape Setting for Dominant Trajectories**

The means (μ) and standard deviations (σ) of landscape variables are listed for each landscape trajectory (Table 5-5) indicating spatial patterns with regard to the physical or socioeconomic landscape setting. On average, trajectories accounting for wetland loss (*w-c-c*

and *w-w-c*) occurred closer to population centers and roads than conserved wetlands (*w-w-w*). Conversion to cropland also occurred on slightly higher, more-sloped land relative to wetland conservation (*w-w-w*). The latter trajectory occurs at the lowest elevation (μ = 9 m) and slope (μ = 0%), and in sites that are most-isolated relative to infrastructure like roads (μ = 2.9 km) and towns (μ = 17.3 km).

Explanatory trajectories for cropland gains were those of grassland conversion, where conversion to cropland in the first time step (*g-c-c*) occurred, at lower elevation, gentler slopes and farther from forest and infrastructure, relative to conversion that occurred in the second time step (*g-g-c*). The dominant grassland loss trajectories were *g-f-f* and *g-g-f*, though clearly conversion to cropland played a significant role. Reforestation in the first time step (*g-f-f*), relative to the second time, occurred at higher elevations (239 vs. 175 m), steeper slopes (14 vs. 12%), farther from population centers (11.1 vs. 10.0 km), and effectively the same distance from roads (1.5 km vs. 1.4 km).

On average the *g-f-f* sites occurred closer to existing large forest patches (1.0 km) compared with grassland reforestation in the second time step (1.3 km). Forest conservation (*f-f-f*) occurred at highest elevations (281 m) and steep slopes (14%). This trajectory was also closer to extant forest (398 m) than any other trajectory, in part by definition. And finally, *f-f-f* trajectories were surpassed in distance to infrastructure (d.rd 2.0 km and d.popctr 12.0 km) only by the wetland conservation trajectory.

Sites that were grassland throughout (*g-g-g*) occurred at elevations (145 m) and slopes (7%) intermediate to grasslands that reforested and grasslands converted to cropland. Similarly, *g-g-g* sites were intermediate, relative to conversion to cropland and reforestation trajectories, with regard to distance from large forest patches (1.4 km). On average, sites characterized by the

*g-g-g* trajectory were more distant from population centers (9.8 km) and roads (1.6 km) than grasslands converted to cropland.

## Discussion

Gains in cropland were greatest from 1987 to 2000, after the implementation of PRAT which underscores the importance of regional irrigation infrastructure in changing the land-use potential for this seasonally-arid landscape. Logically, wetland loss was also greatest in the second time step since wetlands were predominantly converted to crops. Greater gains in forest area were seen in the first time step corresponding to forestry tax credits, the establishment of several protected areas, and decreased profitability of cattle ranching resulting from a significant decline in beef prices and reduced production subsidies.

Grassland loss was greatest in the first time step, effectively reciprocal to gains in forest area. Since this time step included the crash of beef prices, this indicates a tradeoff between cattle ranching and forest-related land uses. The decline of grasslands in the second time step was also substantial, however, owing to continued reforestation and intensification facilitated by PRAT. Grasslands in the lowland areas of the watershed were converted to crop cultivation. Also in the second time step, incremental additions of forest on private land was facilitated through forestry incentives and PES. For example, about 204 ha of reforestation were afforded during this period through the reforestation modality of PES (Figure 5-4).

Results suggest that conversion processes had unique domains within the broader landscape. Wetland conservation (*w-w-w*) occurred on the lowest-lying, flattest and most-isolated regions of the basin, whereas those wetlands first converted to cropland (*w-c-c*) were on higher, faintly-sloping land. These patterns reflect the importance of both drainage and accessibility for crop cultivation. In especially boggy sites, crop yields may be diminished by

seasonal flooding. Or the required investment for wetland drainage may deter conversion in such locations.

Grassland areas undergoing intensification through conversion to cropland were located on lower and flatter land. In contrast, grasslands on higher, sloped lands—marginal areas for intensive cultivation or pasture use—generally became reforested. These findings concur with results from an analysis of forest cover change for a broader region of northwest Costa Rica (Arroyo-Mora et al. 2005), also described in Kozak (this volume) for the Polish Carpathians. Further, in agreement with patterns observed in developed-world forest transitions, both agriculturally-marginal and isolated lands preferentially experienced reforestation (Mather and Needle 1998). Grassland areas where forest regenerated were farther from population centers and roads, but closer to large patches of extant forest, than was grassland that converted to cropland.

A substantial portion of the net increase in forest area observed by year 2000 occurred due to reforestation and forest conservation on privately-held land. This provides a quasi-control for the influence of protected areas on forest expansion in the landscape, lending support to evolving forestry incentives and the decreased profitability of cattle-ranching as drivers of forest recovery. The latter set a favorable context for the transformation of the dominant land use system, reinforced by PRAT's implementation. Low land rent from cattle ranching also contributed to spontaneous forest regrowth by facilitating the retirement of marginal areas of the landscape. Coupled with the forestry-incentive tax credits and later PES, relatively unprofitable cattle ranching enhanced the economic attractiveness of forest-related land use. Since no systematic spatial records are available for landholders that participated in the forestry-incentives program, their historic contribution to forest recovery was not explicitly quantified (but see

Sanchez-Azofeifa et al. 2007 which used GPS points taken in the general vicinity of PES projects from 1997-2000, as opposed to polygons needed for this spatially-explicit analysis).

Protected areas were established preferentially in locations where forest had been conserved; on average, parks had higher elevations, steeper slopes and were more isolated. This is not unique among analyses of protected area site selection patterns (e.g., Rouget et al. 2003, Southworth et al. 2004). Precisely these characteristics contribute to the persistence of forest cover by making such areas less competitive for economically-driven and deforesting land uses (noted in several case studies in this volume). These results, in addition to the forest persistence trajectory (*f-f-f)* on marginal lands in the non-protected landscape, suggest that forest in some now-protected locations would have been conserved during the study period due to the landscape's physical constraints alone. In terms of forest expansion within protected areas, reforestation of grasslands nearly equaled extant forest that became park land. Trends in the non-protected landscape suggest that some of these grasslands in lowland areas may have been converted to cropland while uplands may have reforested even without protected area status (exemplifying Grainger's point that under certain conditions, .

The history of this landscape underscores that forest cover expands and shrinks, in part, with economic trends that influence the dominant land use system(s). Establishing protected areas in perpetuity ensures that at least some minimum forest remains regardless of the economic context, at least in Costa Rican, where protected area boundaries are generally respected. While there is forest cover in the non-protected landscape that appears relatively unthreatened either due to geographic setting or short-term (5 year) protection through PES, deforesting or forest degrading land uses for these such areas is conceivable in the future. Further, most of the basin's non-protected forest is in varying successional stages. Protected area establishment may not have

been determinant of forest expansion in the basin, but is likely a critical element for the long-term recovery of old-growth tropical dry forest—one of the most endangered ecosystems in the world.

The unique landscape niches associated with particular trajectories, along with the broader drivers of landscape change, can be synthesized in a conceptual model explaining observed forest expansion in the Tempisque Basin (Figure 5-7). Grasslands in sufficiently flat, accessible areas intensified from pasture use to crop cultivation; on more marginal, isolated grasslands, or those falling within protected areas, reforestation occurred. Forces behind forest recovery in the basin are multiple and interrelated as with other case studies in this volume (e.g., the combination of economic and political drivers of Vietnam's transition, see Meyfroidt and Lambin).

The causal structure that had long ensured the maintenance of cleared land got rearranged (see Perz and Almeyda, this volume) in the Tempisque Basin. The conjuncture of protected area establishment, declining beef prices, diminished subsidies for beef production, implementation of regional irrigation and the advent of forest incentives manifested in significant forest recovery. The decreased profitability of cattle ranching appears to have set a favorable context, whereby its synergy with land-use intensification and forest protection, acted at a critical moment in the history of the landscape to favor forest expansion. Had nothing changed regarding the economics of cattle ranching, the impact of forest incentives and irrigation on forest cover in the Tempisque Basin would have probably been diminished.

Models attempting to explain forest-cover dynamics may fail to address mechanisms of non-forest land-cover change concomitant with changes in forest area. Understanding the relationship between different conversion processes in landscapes where forest expansion has

been observed will contribute to a more nuanced understanding of forest recovery. In the case of the Tempisque Basin, considering the landscape as a forest/non-forest dichotomy would have failed to reveal the clear trend of land-use intensification, a critical component of landscape change. Intensification, while beneficial for forest recovery, has been detrimental to wetlands given that roughly two-thirds of non-protected wetlands were converted to cropland. The ways in which land- cover patterns are affected by shifts in the dominant land use system (e.g. from extensive cattle ranching to intensive use) must be anticipated to ensure policy coherence. For instance, a system of wetland conservation or restoration on private land—analogous to tax incentives or, more recently, the PES scheme for forests—may mitigate negative tradeoffs from the indirect competition between reforestation and wetland conservation, processes that interact despite occurring in distinct niches of the landscape.

The proposed model of forest expansion (Figure 5-7) begs several important questions about the longevity of the observed forest recovery in the Tempisque Basin. The unique landscape niches of different trajectories highlight the need for spatially-conscious management decisions regarding how shifts in land use are facilitated or constrained by the physical landscape. Significant portions of the non-protected landscape, occurring on sites marginal for cultivation or pasture use, appear to comprise "conditional forests," land that is forested depending upon the economic and policy context. If forest protection (article 19 of Ley 7575) is not fully implemented, if PES initiatives are eliminated, and/or if extensive (as opposed to intensive) cattle ranching is profitable again to the extent that it had been, will forests on marginal areas persist as implied in the enduring forest recovery concept of a "forest transition"?

Forest protection mechanisms and incentives like PES are clearly critical to facilitating secondary forest regeneration and conservation. In Costa Rica, these factors were locked in at an

optimal time when land-use economics were being reorganized. Recent trends (i.e., post-2000) certainly support this notion in that there are now more head of cattle in Guanacaste than in the best years of extensive ranching model of the past—yet on far less pasture. The Ministry of Agriculture (MAG) still subsidizes cattle production, but of the intensive sort. Synthesizing production patterns and forest recovery patterns suggests that Costa Rica has crafted relatively effective institutions for addressing historic deforestation threats. But are these institutions equipped to conserve the observed forest recovery for the long-term (see Daniels et al. 2009)?

Demographic factors have often figured prominently in explaining forest cover trends (Meyer and Turner 1992). Indeed, about half of the variance in the extent of deforestation can be explained by population in the long run, though the relationship is far from simple or static (Mather et al. 1998). Similarly, one of the major forest transition pathways is thought to be the demographic trend of out-migration in rural landscapes where forest regeneration occurs in the wake of a diminished labor pool and associated land abandonment (Rudel et al. 2005, Taff et al. this volume). In northwest Costa Rica, however, Harrison (1991) found that forest cover was not correlated with population trends from 1950 to 1984, the most intense period of deforestation (Harrison 1991). Given the labor-saving, extensive land tenure and land-use system in place during the centuries of deforestation, the lack of correlation is intuitive and noted to characterize much of Latin America (Sloan 2007).

The relationship between forest cover and population in recent years for the Tempisque Basin has not been examined. Yet from 1984 to 2000, corresponding roughly to the second time step in this analysis (1987 to 2000), the population growth rate for Guanacaste was nearly 2% (compared with less than 1% from 1973 to 1984, roughly the first time step in this study). Trends of net emigration diminished four-fold in the period from 1995 to 2000 relative to trends

in the 1970s. Net negative migration even reversed in some cantons to net positive immigration.[6] With half again as many people in 2000 as at the beginning of the study (represented by 1973 census data) and a declining or reversing trend for out-migration in the province, forest recovery does not appear to be driven by depopulation. This suggests that population patterns were independent of deforestation or reforestation of the Temspique Basin. A carefully-designed analysis of demographic trends with disaggregated data and corresponding land cover maps is needed to test this.

Other broad classes of explanations for forest transitions discussed in the literature find greater support in the Tempisque Basin. Grainger (1995) describes several mechanisms that resulted in forest transitions over the course of economic development. Agricultural intensification leads to abandonment of marginal lands which, in turn, reforest. He attributes this process to industrialization/urbanization and decreased competitiveness of small-scale agriculture. While intensification was certainly an important component of forest expansion in the Tempisque Basin, its drivers were distinct from Grainger's generalized scenario. This study illustrates the importance of context-specific factors like PRAT, in explaining forest transitions.

Grainger also describes a mechanism whereby improved land management and a shift in attitudes about forest that occurs through development. This certainly characterizes the case of the Tempisque Basin, and Costa Rica more generally, over the last several decades (also see Nagendra, this volume). The shift in forest valuation was not driven only by nature-based tourism, but was facilitated by exogenous forces like the global recession, crash of the beef market and reduced public spending. The latter circumstances brought about a new economic context and a reorganization of national development goals. Hence, according to the experience

of the Tempisque Basin, a robust forest transition theory cannot neglect to incorporate both context-specificity and non-local influences on land use and land cover.

Agriculture—extensive or intensive—is hardly the major threat to forests anymore in the Tempisque Basin. Results illustrate that low-intensity land uses account for an ever-diminishing area in the Tempisque Basin. Perhaps the greatest uncertainty regarding the future of forest cover and the longevity of the observed forest expansion is landscape gentrification and associated real-estate development. The forestry law's ban on forest land-use change proves costly to enforce and arguably altogether unrealistic. Property values in the basin have increased steeply from real and speculative land investment but PES does not exceed the opportunity cost of land development which externalizes ecosystem degradation. About 24,853 ha (Figure 5-4) of forest were protected through tax vouchers (17,000 ha) and direct payments (7,853 ha), representing 10% of the 2000 forest area in the basin. Yet once these ecosystem-service contracts expire, the fate of these forested lands is uncertain.

A critical but largely absent dimension in forest transition theory is the role of regional and international timber trade in facilitating forest recovery at local and regional scales. At the national level, Costa Rican timber imports have increased dramatically in recent years (ONF 2006), underscoring that forest recovery does not imply a net decrease in timber demand. Analogous to the forest transition path of many developed nations, forest conservation and regeneration in Costa Rica could be simply afforded by displaced deforestation and/or expansion and intensification of timber plantations elsewhere.

Ultimately, context-specific case studies must be linked to broader global forest cover trends and the supply/demand of forest goods in order to appropriately contextualize regional or national-level forest recovery, its driving forces, and its degree of permanence. These are all

critical points for post-Kyoto REDD initiatives (see also Grainger, this volume). As of now, the issues of land development pressure, the timber trade and deforestation displacement are not explicitly addressed in the forest transition literature for developing countries.

The phenomenon of forest recovery in the Tempisque Basin is instructive. It casts local forest expansion, not as an inevitable by-product of economic development as suggested by forest transition theory, but as a process driven by the conjuncture of key global, national and local factors affecting land use patterns like the international beef market; national forest protection policies and irrigation development; and local land use intensification respectively. Evidence from this research suggests that patterns of forest cover like those characterized by the "hollow frontier" and "forest transition" concepts are not necessarily competing models in explaining forest-cover trends after the passing of an agricultural frontier. Rather, these models may describe distinct phases of a longer-term process of forest change in different economic and development contexts. The challenge, insofar as the provision of forest goods and services is concerned, lies in understanding the levers that maintain a favorable context for the long-term conservation of this forest recovery in a dynamic, developing economy.

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Table 5-1. Protected area in the Tempisque Basin over the three dates of land cover analysis.

| Year | Protected Area (ha) intersecting watershed | Protected Area (ha) contained in watershed | Percent of Watershed* |
|---|---|---|---|
| 1975 | 55,114 | 6,208 | 1.2 |
| 1987 | 76,472 | 28,176 | 5.6 |
| 2000 | 127,561 | 47,340 | 9.5 |

* after eliminating pixels that were not represented across all three image dates either due to cloud cover in one or more dates or WRS1 scene truncation. The latter eliminated most of Santa Rosa and Guanacaste National Parks.

Table 5-2. Description of land cover classes (abbreviations in parentheses).

| Class | Description |
|---|---|
| Crop (*c*) | Dominant crops include rice, sugar cane and melon; includes all phenological stages of each crop (including bare soil associated with pre- and post-harvest crop cover on agricultural plots |
| Wetlands (*w*) | RAMSAR definition of wetland was employed[1] with the exception that flooded rice fields were considered crops; includes open water, mangroves, flooded forests, fresh marshes with emergent and floating vegetation, fresh meadows of grasses and sedges; includes all phenological stages of each sub-classes |
| Grasslands (*g*) | Includes strict pasture of grasses only, pasture with occasional trees, recently fallowed pasture scrub (<3 years) still dominated by grasses, grassy areas along roads, and finally, regions frequently burned that cannot support appreciable woody growth |
| Forest (*f*) | Limestone forests, deciduous lowland forest, evergreen lowland forest, and pre-montane moist forest; rainy season canopy closure $\geq$ 35% |

[1] *Areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters.*

Table 5-3. Of the possible 64 trajectories (left), only the dominant explanatory trajectories were retained for analysis of each trend of net land cover change. Explanatory trajectories account for the majority of net area change (gain or loss).

| | Explanatory Trajectories[1] | | |
|---|---|---|---|
| All possible trajectories | Forest Gain | Cropland Gain | Wetland Loss |
| 1. g-g-g | g-f-f | g-c-c | w-c-c |
| 2. g-g-f | g-g-f | g-g-c | w-w-c |
| 3. g-g-c | f-f-f | c-c-c | w-w-w |
| n. etc. | | | |
| 64. w-g-c | | | |

Table 5-4. Population means (μ) of key variables representing physical and socioeconomic landscape setting in the Tempisque Basin, stratified by protected and non-protected status. D.popctr, D.rds and D.forest signify distance to nearest population center, nearest road and large forest patch (> 50 ha in 1975), respectively.

| | N | Elevation (m) | Slope (%) | D.popctr (m) | D.rd (m) | D.forest (m) |
|---|---|---|---|---|---|---|
| Non-Protected | 3,813,062 | 191 (205) | 10 (16) | 10,197 (6,121) | 1,527 (1,379) | 1,108 (1,409) |
| Protected | 394,436 | 232 (291) | 12 (18) | 16,129 (5,988) | 3,198 (2,248) | 415 (790) |

Standard deviations (σ) in parentheses.

Table 5-5. Population means (μ) of variables representing physical and socioeconomic landscape setting in the Tempisque Basin, stratified by unique land cover trajectories. Tempisque Basin for. D.popctr, D.rds and D.forest signify distance to nearest population center, nearest road and large forest patch (> 50 ha in 1975), respectively. The letters in each trajectory name represent land cover for 1975-1987-2000, respectively, with the following abbreviations: c = crop, g = grassland, f = forest, and w = wetland.

| | N | Elevation (m) | Slope (%) | D.popctr (m) | D.rd (m) | D.forest (m) |
|---|---|---|---|---|---|---|
| g-f-f | 858,042 | 239 (202) | 14 (18) | 11,119 (6661) | 1,527 (1451) | 1,010 (1297) |
| g-g-f | 510,910 | 175 (182) | 12 (17) | 9,961 (6315) | 1,416 (1475) | 1,261 (1419) |
| w-c-c | 43,977 | 13 (12) | 1 (2) | 11,450 (4350) | 2,555 (1547) | 1,430 (1236) |
| w-w-c | 27,295 | 13 (9) | 0 (2) | 9,434 (5133) | 1,810 (1480) | 1,537 (1320) |
| g-c-c | 158,955 | 20 (16) | 1 (2) | 7,891 (3700) | 1,397 (1033) | 2,050 (1669) |
| g-g-c | 180,829 | 46 (73) | 2 (5) | 8,420 (4838) | 1,153 (1027) | 1,871 (1567) |
| f-f-f | 1,105,410 | 281 (252) | 14 (18) | 12,035 (6356) | 2,000 (1609) | 398 (994) |
| g-g-g | 934,159 | 145 (154) | 7 (13) | 9,755 (6096) | 1,600 (1689) | 1,357 (1403) |
| c-c-c | 65,815 | 18 (15) | 1 (2) | 7,698 (3435) | 1,326 (1063) | 2,250 (1573) |
| w-w-w | 84,159 | 9 (6) | 0 (2) | 17,310 (6897) | 2.946 (1596) | 1,375 (1066) |

Standard deviations (σ) in parentheses.

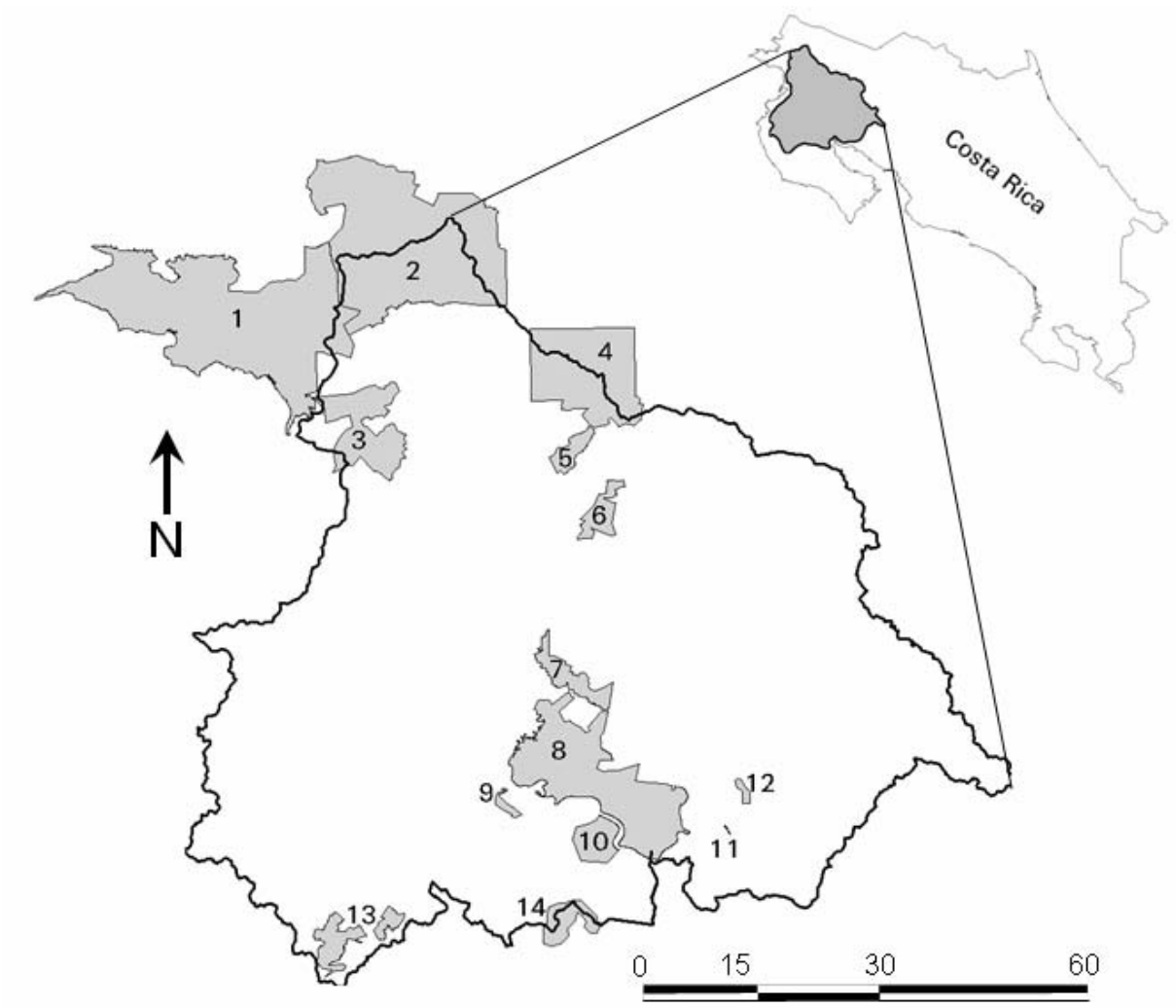



Figure 5-1**.** Map of Tempisque River Basin in northwest Costa Rica (5414 km$^2$). Shaded polygons represent protected areas within, or that partly intersect with, the basin. For this analysis, portions of protected areas falling outside the basin were excluded. Protected areas are labeled as follows: (**1**) Santa Rosa National Park (38,656 ha, 1966+), (**2**) Guanacaste National Park (34,651 ha, 1991), (**3**) Horizontes Experimental Forestry Station (7330 ha), (**4**) Rincon de la Vieja National Park (14,161 ha, 1974), (**5**) La Virgen State Farm (1,923 ha), (**6**) Las Delicias State Farm (1,378 ha), (**7**) Lomas de Barbudal Biological Reserve (2,645 ha, 1986), (**8**) Palo Verde National Park (18,410 ha, 1977+), (**9**) Mata Redonda Lagoon (372 ha, 1994) (**10**) Corral de Piedra Wetland (2,484 ha, 1994), (**11**) Madrigal Lagoon (12 ha, 1994) (**12**) Taboga Forest Reserve (303 ha, 1978) (**13**) Diria Nacional Forest (13,402 ha combined, 1991), (**14**) Barra Honda Nacional Park (2,297 ha, 1974).

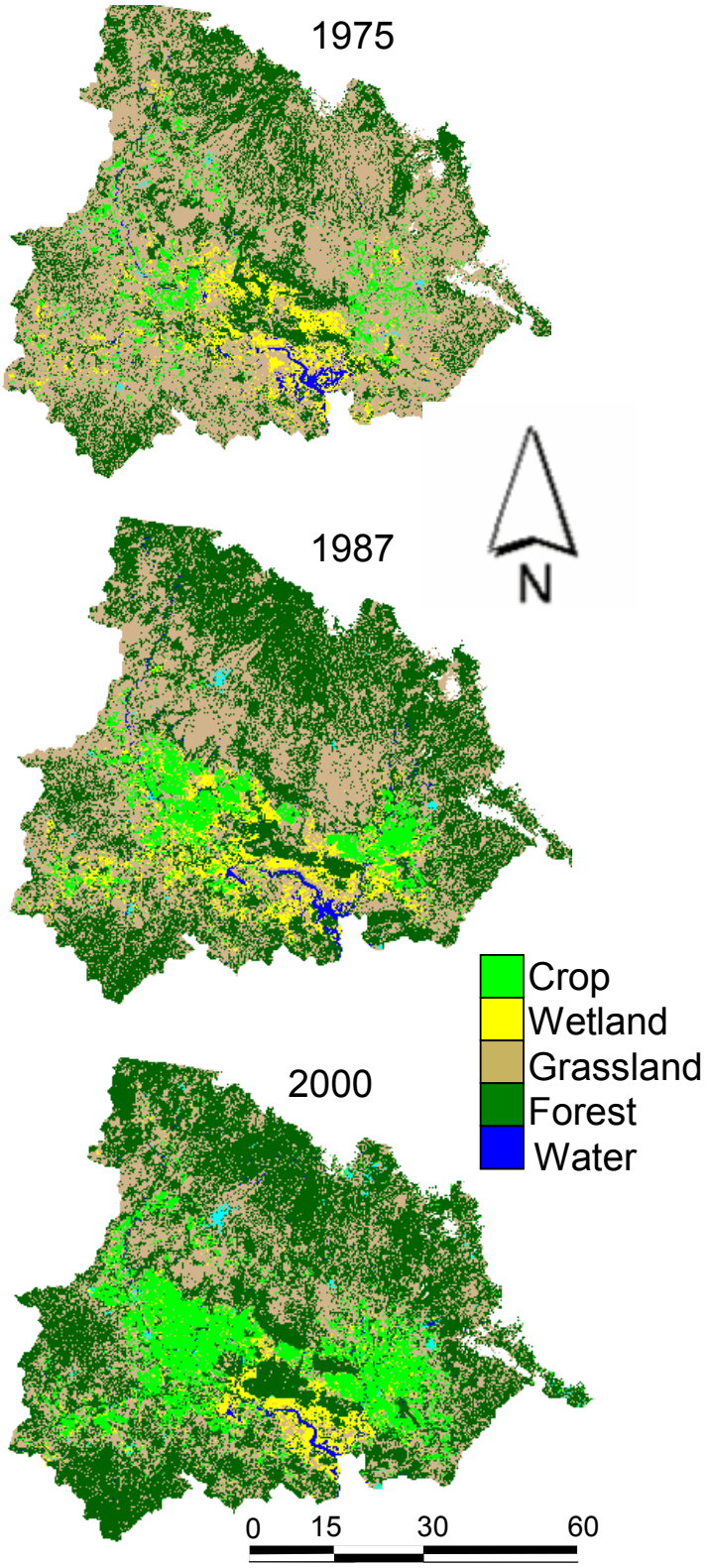



Figure 5-2. Four class land cover maps (plus water) for 1975, 1987 and 2000.

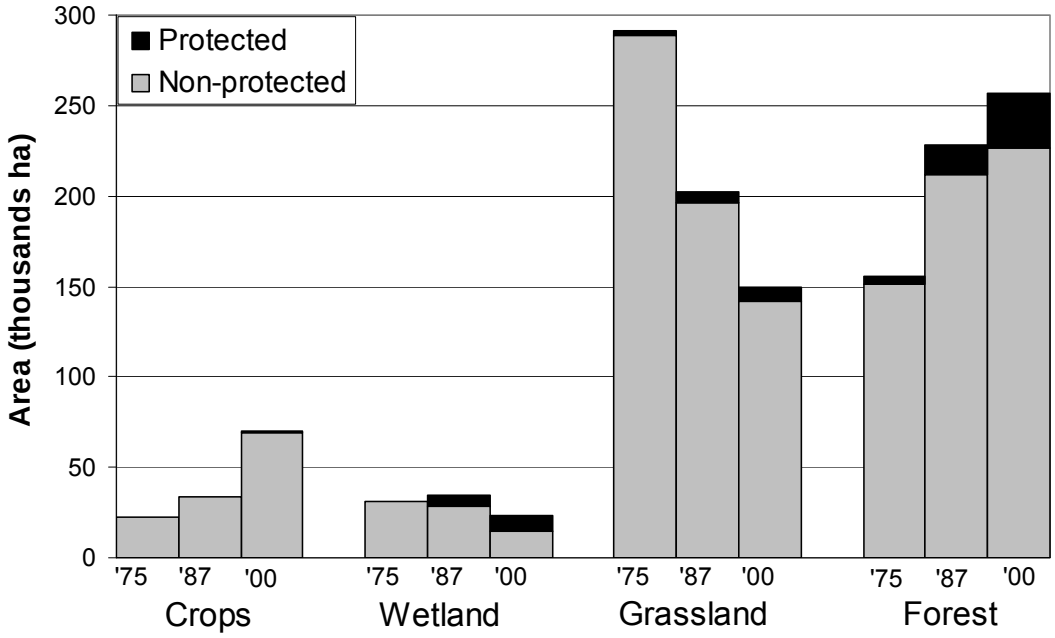
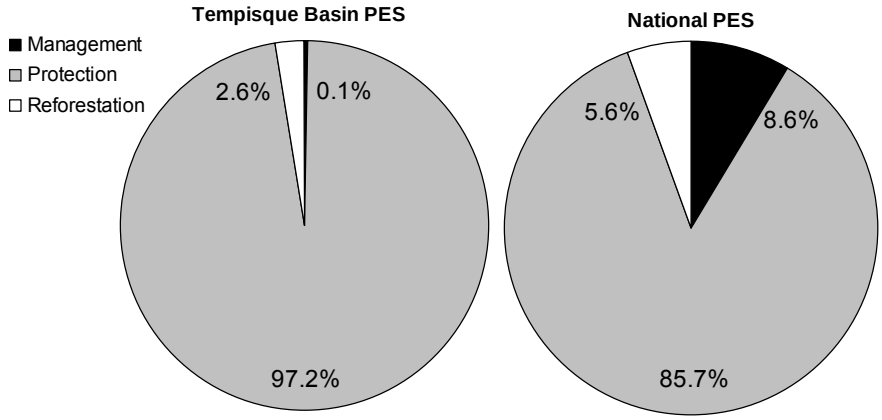



Figure 5-3. Trends of net area land-cover change for 1975, 1987 and 2000 in the Tempisque River Watershed. The black region of each bar indicates the contribution of each land cover class within protected areas.

| Incentive[1] | Active Period | Basin (ha) | National (ha) |
|---|---|---|---|
| IDR | 1969 - 1988 | unknown | 35,597 |
| CAF | 1986 - 1995 | unknown | 38,086 |
| CAFA | 1988 - 1995 | unknown | 33,818 |
| CAFMA | 1993 - 1995 | unknown | 22,120 |
| CPB | 1995 | 17,000[2] | 22,199 |
| PES | 1997-2000 | 7,853[3] | 256,520 |




[1]See Daniels et al. 2009 for details on the evolution of incentives. Abbreviations are acronyms for the following titles that have been translated to English: Income Tax Deduction, Certified Forestry Tax Voucher, Advanced Certified Forestry Tax Voucher, Advanced Certified Forest Management Tax Voucher, Forest Conservation Tax Voucher, and Payments for Environmental Services.
[2]Estimated by Ing. Francisco Ramirez, Area de Conservacion de Guanacaste (SINAC). No other data were available.
[3]These data are from 1999 and 2000 either because no projects were located in the basin from 1997-1998 or because of poor data management when PES field administration changed from SINAC to FONAFIFO.

Figure 5-4**.** Area protected through various forms of forestry incentives for the Tempisque Basin and at the national level. The pie charts illustrate the break-down of 1997-2000 PES area across the three forestry modalities.

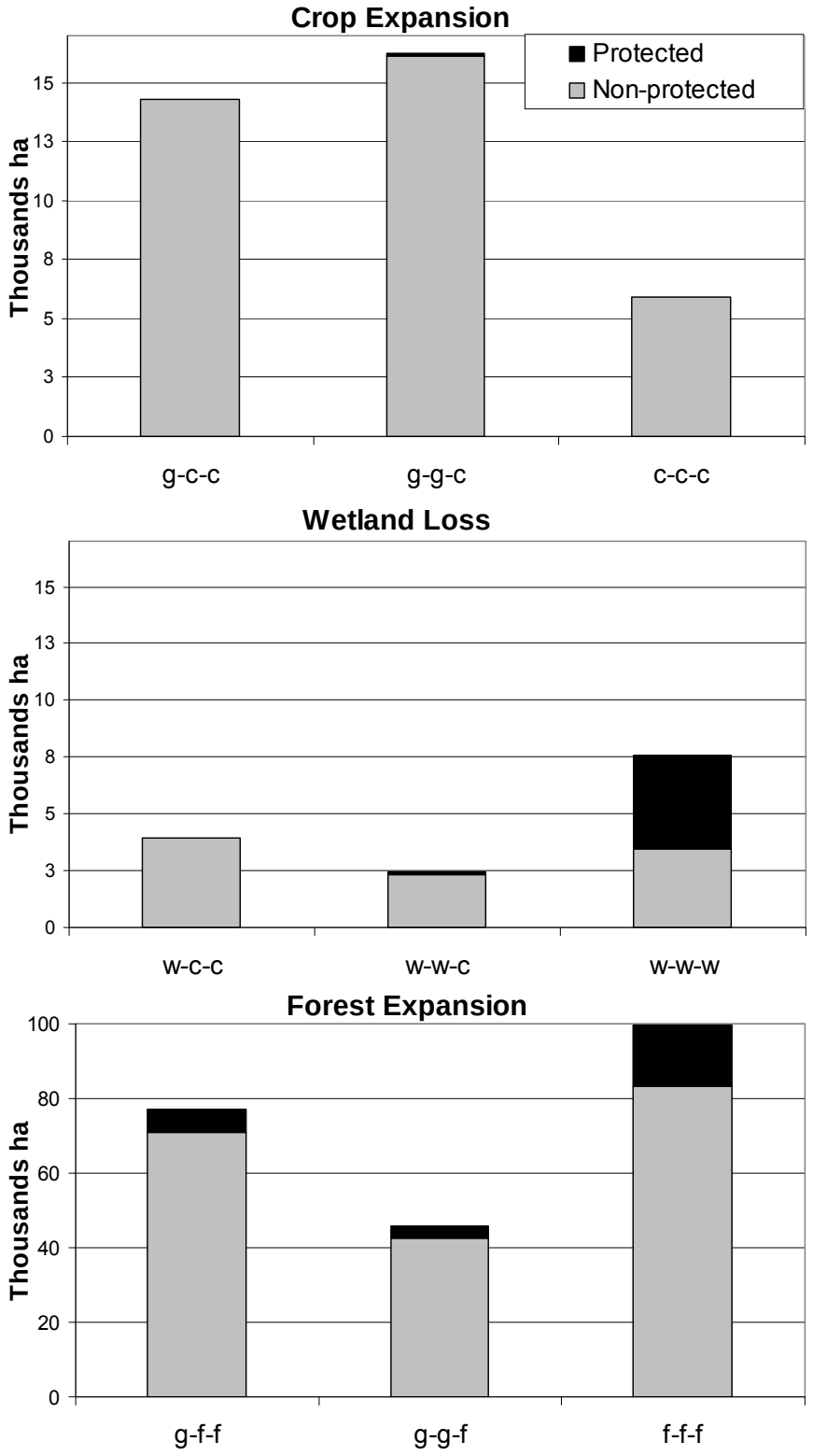



Figure 5-5. Dominant land cover trajectories explaining the observed net area changes in land cover. The black region of each bar indicates the contribution of respective trajectories within protected areas. The three letters in each trajectory name represent land cover for 1975-1987-2000, respectively, using these abbreviations: c = crop, g = grassland, f = forest, and w = wetland.

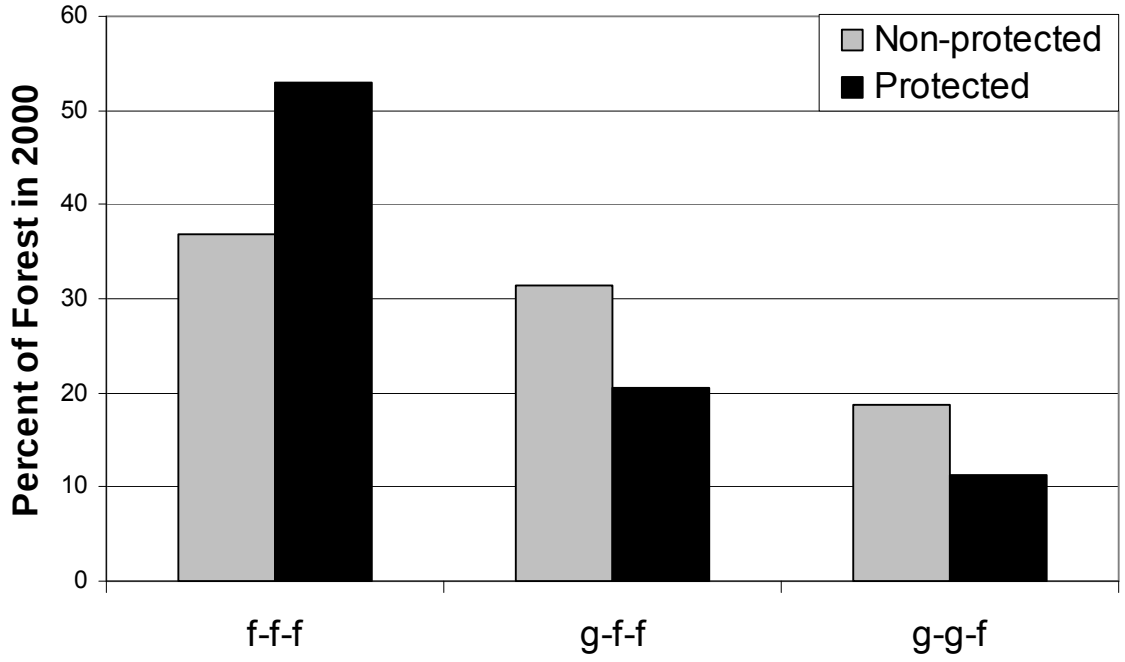



Figure 5-6. Graph of dominant trajectories explaining the expansion of forest observed by year 2000 for protected and non-protected areas of the landscape. The bars indicate the percent of non-protected (gray) or protected (black) forest in year 2000 explained by each trajectory.

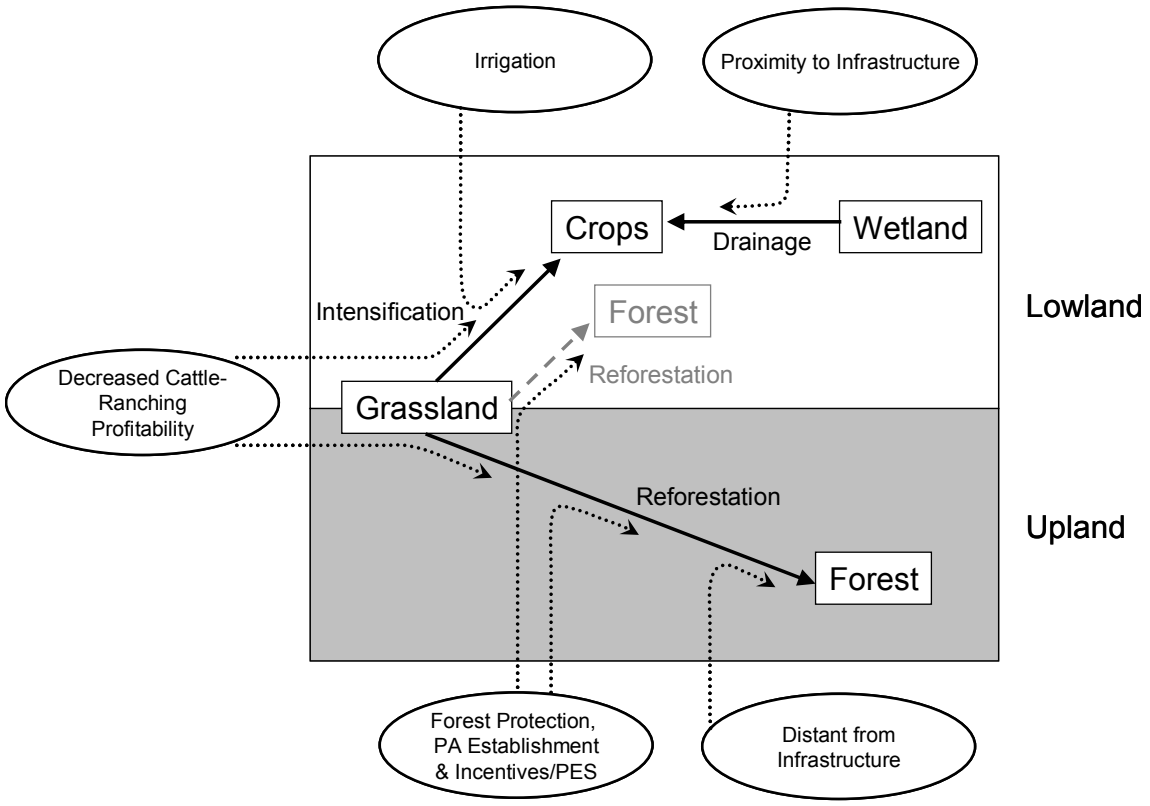



Figure 5-7. Conceptual model of dominant trends accounting for forest expansion in the Tempisque Basin. The landscape is represented by the large rectangle, divided into uplands (gray) and lowlands (white). The four land cover classes analyzed in this study are represented by small rectangles with arrows depicting the dominant trajectories (e.g. grassland conversion to crops). Ovals around the landscape depict the forces driving observed processes of landscape change. The dominant reforestation trend occurred in upland areas (black). Lowland reforestation was a minor factor in accounting for forest expansion (gray).

# CHAPTER 6
# CONCLUSIONS

At the time scale of an average human lifespan, humans may be the most dominant force shaping ecosystem function and structure (Meyer and Turner 1994). Society has domesticated landscapes and simplified ecosystem functioning to enhance the rate and amount of particular ecosystem goods produced or capable of being extracted at any single location (e.g. food and fiber production through crop cultivation, timber production through managed forestry systems, or energy extraction through fossil fuel mining). Ecosystem specialization, by necessity, expands the human footprint or dimensions of the resource catchment from which the needs of a typical person are met, particularly in the developed world. Ecosystem specialization is thus both a cause and a consequence of globalization as disparate and remote locations become increasingly interdependent through trade. Forces in distant locations drive local land cover change. These changes alter the global mix of ecosystem services, but may have disproportionate effects on local ecosystem service provision like water quality control or the maintenance of soil productivity. Processes of environmental change and ecosystem service tradeoffs are complex, and multidirectional system. Conservation dialogues and financial investments focusing narrowly on traditional protected areas are therefore ever more irrelevant (Kareiva et al. 2007). Protected areas undoubtedly serve an important function within conservation agendas; but traditional parks cannot cover sufficient area to preclude difficult present and future decisions about resource and land use on the margin.

Minimizing tradeoffs between conservation and economic production functions depends in part on the spatial pattern of society's resource and land use (Polasky et al. 2005). As revealed in chapter two, spatial juxtapositions, like *milpa*/forest adjacency in the tropical dry forest of the Yucatan, affect ecosystem structure. Increasingly, scientists highlight the importance of past

land use on contemporary ecosystem properties (Foster et al. 2003). Rarely, however, are spatial patterns of past land uses considered as emphasized in chapter two. These results underscored the importance of characterizing spatial patterns over time when attempting to understand the legacy of past land use systems and also the need to measure changes to ecosystem structure within a given land cover class (forest in this case). Results showed that the trait values or forest structure for vegetation occurring at milpa edges were different from background forest. If basal area roughly indicates carbon storage, just one particular ecosystem service, then minimizing the length of edge per area of *milpa* within *ejidos* would enhance this particular service.

The wetland conversion model in chapter three provided a detailed case study that crystallized an important ecosystem tradeoff occurring around the globe: the loss of ecosystem services provided by wetlands in exchange for crop cultivation. Whether this transformation of the lower Tempisque basin is reversible remains to be seen and hinges on the degree to which previous landscape hydrologic connectivity has been severed through geomorphic alterations. For the foreseeable future, these low-lying, flat areas—cultivable now that they are drained—will likely remain in crop land use. Costa Rica has a biofuel mandate that will likely keep the demand for sugar cane high in the coming years. This spatially-explicit model not only estimates the probability of conversion based on past conversion patterns, but it can be used as a spatial-targeting policy tool. For example, if a wetland-analogue to the forest-based PES system were implemented, this model would identify the sites most likely to be converted to other land uses and aid in the differentiation of payments according to the opportunity cost.

Chapter five emphasized the critical links between institutions and landscape patterns. For much of the Tempisque Basin's history, the prevailing extensive *latifundio* assured that extensive cattle ranching was the predominant land use system, at the expense of forest cover.

Once the global market acted to remove the pillars propping up this antiquated institution, the landscape was subject to reorganization. Both agricultural intensification and forest conservation policies transformed land-cover patterns in recent decades. Protected area establishment certainly facilitated conservation of existing forest and encouraged some degree of reforestation, though much of the land incorporated into parks was disproportionately forested already. Further, much of the gain in forest cover was achieved through reforestation on private land, particularly upland areas.

New institutional driving forces in the landscape facilitated forest regrowth and thus forest-related ecosystem services like carbon sequestration. Though land-use intensification released marginal lands from extensive cattle ranching for reforestation, intensification played an important role in the loss of wetlands in the Tempisque Basin. Wetlands and forests were not in direct competition since these ecosystems occur in distinct domains of the landscape. Institutions did, however, act to create an indirect tradeoff between these two ecosystem types and their respective ecosystem services. Forest protection through park establishment and PES alone may not have been sufficient to transform land cover patterns and the mix of ecosystem services in the Tempisque Basin. But these factors acting in just the right economic context locked the landscape on a new trajectory. The longevity of this forest transition depends largely on the degree to which Law 7575 is enforced and the extent to which opportunity costs associated with PES continue to increase in the basin (relative to one-time land sales for real-estate development or land rent from intensive agriculture).

The analysis of Costa Rica's PES scheme in chapter five demonstrates both the progress and room for improvement within this innovative institutional scheme. Development agencies seek to replicate this model in other parts of the developing world. Yet the context elsewhere

may not be as amenable due to confounding factors like abject poverty, a lack of administrative experience, the absence of land titles, and/or a history of political violence, for example. Costa Rica's experience may minimally serve as a baseline from which other programs may build, tailoring implementation according to the unique challenges presented.

Several suggestions for PES include decoupling the monitoring of PES from the mechanism financing PES monitoring; strategically targeting contracts in order to seek landscape-pattern-based synergies; and reverse auctioning contracts in order to optimize service provision for each dollar spent. Many of the challenges for Costa Rica's PES lay in the actual ground-based implementation, not in technical or policy realms. Even when the design is sound, implementation can be difficult because the human-resources and capacity-building aspects of PES are largely unappreciated.

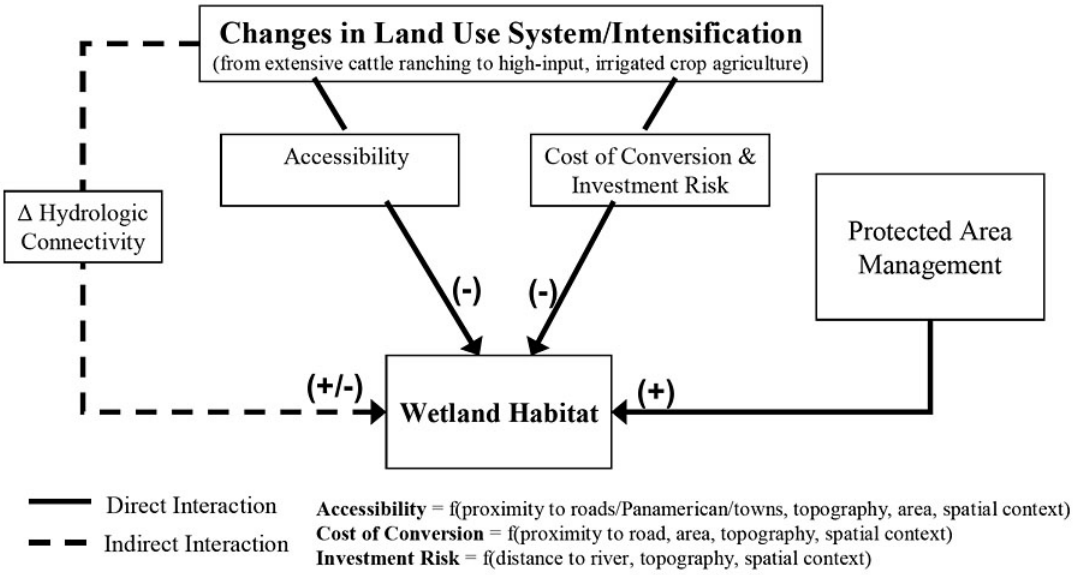
This is a particularly important and timely conclusion from this research. The Norwegian government recently announced that it would contribute $500 million per annum to forest conservation activities that reduce emissions from deforestation and degradation (United Nations 2008). This immense investment will double the total amount of foreign aid dedicated to forests and forestry programs worldwide (Zarin 2008). The dearth of communication between field officers and high-ranking officials in the PES administration is, perhaps, in large part by design. PES administrators and financiers at the World Bank anxiously, and rightfully, await returns on their investment of public funds in the PES scheme. Yet, rather than openly assessing, conceding and re-thinking program weaknesses, the channels for communicating constructive criticism or suggesting program design enhancements are not well-established. Perhaps, in large part, the administration legitimately fears that admitting shortcomings in the PES scheme may deflect potential carbon offset investments and that small Costa Rica would be by-passed for

investments in tropical deforestation hotspots like Brazil and Indonesaia. These complex institutional dynamics underscore the critical importance and challenge of defining PES baselines and measuring additionality in ways that do not penalize Costa Rica for its decades of continually-improved forest stewardship and environmental management.

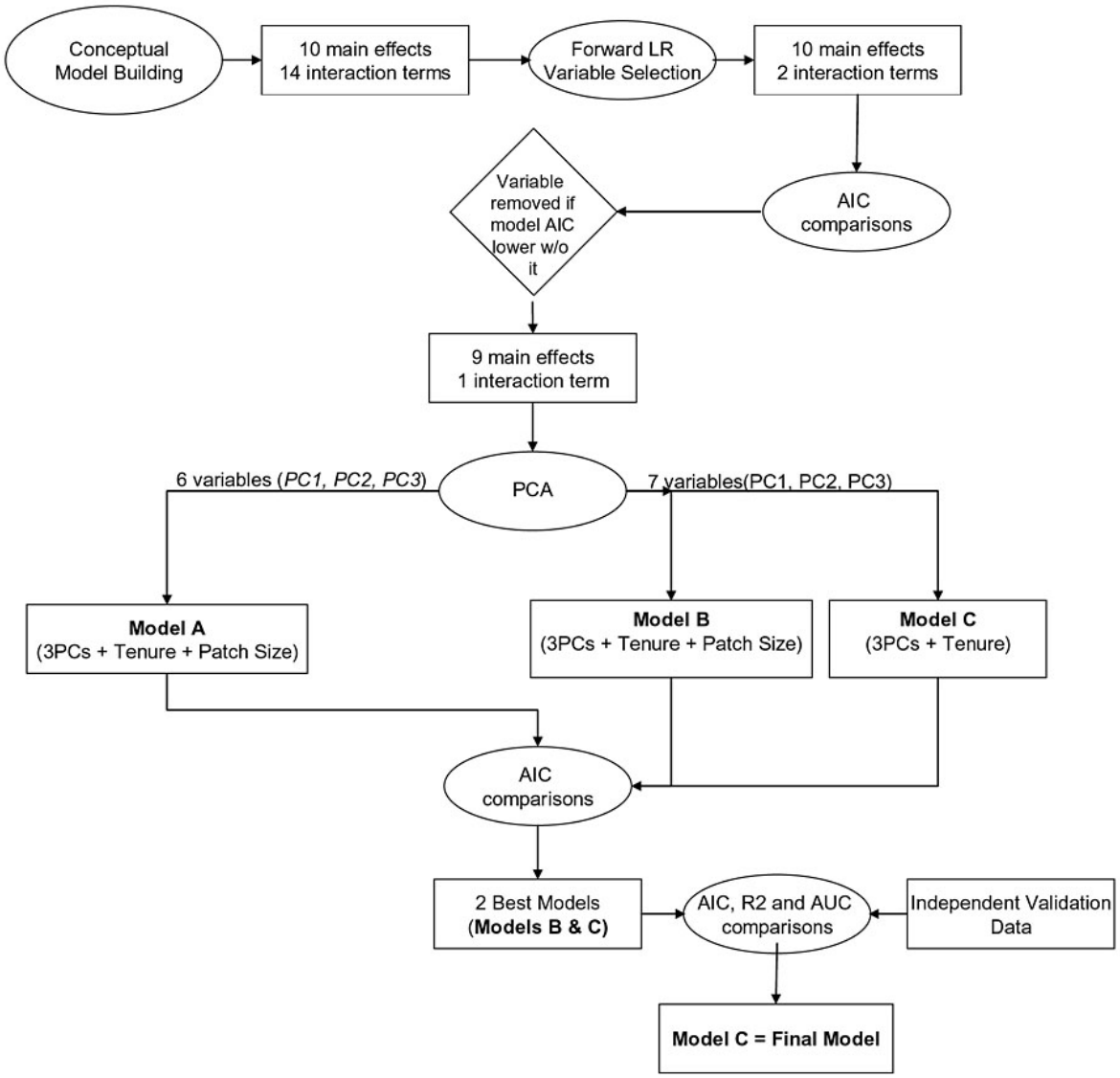
Hence, the real challenge in making use of spatial analyses like those performed in chapters two, three and five, lies in developing the institutions to evaluate and manage the inherent tradeoffs in ecosystem services that arise. Ultimately these decisions about tradeoffs are often value-based. If the aggregate global land use and land cover pattern is explicitly framed as representing the collective ecosystem services most valued at a particular moment in time, this may better facilitate dialogue and deliberate choices regarding their tradeoffs, many of which are irreversible and redirect ecosystem functioning in uncertain ways (Rodriguez et al. 2005). Land change science offers a particularly powerful approach for understanding the patterns and drivers of land conversion and, thus, broad-scale tradeoffs in ecosystem services.

APPENDIX
MODEL BUILDING AND MODEL SELECTION STEPS FOR PREDICTING WETLAND CONVERSION

**Conceptual Model of Wetland Change in the Tempisque Basin**



## Flowchart of Modeling Process



**Data from Akaike Information Criterion Comparisons for Variable Selection and Model Selection**

| Variable Removed | Notes | -2LL | AIC | Δ AIC |
|---|---|---|---|---|
| y coordinate | | 141,090 | 141,108 | 15,353 |
| **x coordinate** | 1 | **117,514** | **117,532** | **-8,223** |
| elevation | | 149,958 | 149,976 | 24,221 |
| slope | | 143,054 | 143,072 | 17,317 |
| distance to river | | 134,668 | 134,686 | 8,932 |
| distance to road | | 141,504 | 141,520 | 15,765 |
| distance to population center | | 141,358 | 141,374 | 15,619 |
| distance to Panamerican | | 142,634 | 142,650 | 16,896 |
| **xy3** | 1 | **114,407** | **114,425** | **-11,330** |
| acs.index | 2 | 125,811 | 125,829 | 74 |
| none (i.e. saturated) | 3 | 125,735 | 125,755 | -- |
| Model A (PC1+PC2+PC3+tenure+patch size) | 4 | 166,303 | 166,313 | |
| Model B (*PC1+PC2+PC3* +tenure+patch size) | 5 | 119,809 | 119,819 | |
| **Model C (*PC1+PC2+PC3* +tenure)** | 5,6 | **125,395** | **125,403** | **Final Model** |

1. Variables were eliminated if their removal resulted in a lower AIC relative to the saturated model.
2. The "asc.index" variable refers to an accessibility index (distance to road * distance to Panamerican).
3. The saturated model included all of the variables listed in the "variable removed" column plus patch size and the tenure indicator. As explained in the text, we wished to quantify the contribution of these variables for specific reasons, so their removal was not considered at this stage.
4. Model A included these variables in the PCA: *y*-coordinate, elevation, slope, distance to river, distance to population center and acs.index.
5. Models B and C included these variables in the PCA: *y*-coordinate, elevation, slope, distance to river, distance to population center, distance to road and distance to Panamerican in the PCA.
6. Model C was the final, selected model despite a slightly higher AIC value. This is because for models B and C we also evaluated $R^2$ and AUC values using independent data. See text for details.

LIST OF REFERENCES

Amador JA, R.E. Chacon, and S. Laporte. 2003. Climate and climate variability in the Arenal River Basin of Costa Rica. In H. F. Diaz and B. J. Morehouse, editors. Climate and water: transboundary challenges in the Americas. Kluwer Academic Publishers, Boston, USA.

Amador, J.A., S. Laporte, and R.E. Chacon. 2000. Cuenca del Rio Arenal: analisis del los eventos Nino de los anos 1992-93, 1994-95 y 1997-98. Topicos Meteorologicos y Oceanograficos 7(1): 1-20.

Ando, A., J. Camm, S. Polasky, and A. Solow. 1998. Species distributions, land values, and efficient conservation. Science: 279: 2126-2128.

Arroyo-Mora, J. P., G. A. Sanchez-Azofeifa, B. Rivard, J. C. Calvo, and D. H.Janzen. 2005. Dynamics in landscape structure and composition for the chorotega region, costa rica from 1960 to 2000. Agriculture Ecosystems & Environment 106: 27-39.

Avalos, A.R. 2007. Ice suspende coretes de electricidad en todo el pais. La Nacion. Sabado 5 de mayo. Available at http://www.nacion.com/ln_ee/2007/mayo/05/pais1085589.html, Accessed on May 5, 2007.

Ayuk, E. T. 1997. Adoption of agroforestry technology: The case of live hedges in the central plateau of Burkina Faso. Agricultural Systems 54: 189-206.

Babcock, B.A., P.G. Lakshminarayan, J. Wu, and D. Zilberman. 1996. The economics of a public fund for environmental amenities: A study of CRP contracts. American Journal of Agricultural Economics 78: 961-971.

Babcock, P.G., P.G. Lakshminarayan, J. Wu, and D. Zilberman. 1997. Targeting tools for the purchase of environmental amenities. Land Economics 73(3): 325-339.

Baldi, A., and T. Kisbenedek. 1999. Species-specific distribution of reed-nesting passerine birds across reed-bed edges: effects of spatial scale and edge type. Acta Zoologica Academiae Scientiarum Hungaricae 45: 97-114.

Banerjee, A. K. 1995. Rehabilitation of degraded forests in Asia. World Bank.

Bierregaard, R.O., Lovejoy, T.E., Kapos, V., Augusto dos Santos, A., Hutchings, R.W., 1992 The Biological Dynamics of Tropical Rainforest Fragments. BioScience 42(11): 859-866.

Baumert, K., T. Herzog, and J. Pershing. 2005. Navigating the numbers: greenhouse gas data and international climate policy. World Resources Institute, Washington D.C., USA.

Bedford, B.L., and E.M. Preston. 1988. Developing the scientific basis for assessing cumulative effects of wetland loss and degradation on landscape functions: Status, perspectives, and prospects. Environmental Management 12(5): 751-771.

Beier, C., and T. Patterson. In review. Targeted social-ecological vulnernability at the landscape scale: ecosystem services, social importance and disturbance. Submitted to Ecosystems.

Bengston, D.N., and Y.C. Youn. 2006. Urban Containment Policies and the Protection of Natural Areas: The Case of Seoul's Greenbelt. Ecology and Society 11: 3.

Berentsen, B.M., A. Hendriksen, W. Heijman, and H. van Vlokhoven. 2007. Costs and benefits of on-farm nature conservation. Ecological Economics 62: 571-579.

Biesanz, R., K.Z. Biesanz, and M.H. Biesanz. 1982. The Costa Ricans. Waveland Press, Prospect Heights, IL, USA.

Bonan, G.B., S. Levis, S. Sitch, M. Vertenstein, and K.W. Oleson. 2003. A dynamic global vegetation model for use with climate models: concepts and description of simulated vegetation dynamics. Global Change Biology 9: 1543-1566.

Bunn, S. E., P. M. Davies, and T. D. Mosisch. 1999. Ecosystem measures of river health and their response to riparian and catchment degradation. Freshwater Biology 41: 333-345.

Boylan, K.D., and D.R. Mclean. 1997. Linking species loss with wetlands loss. National Wetlands Newsletter 19(6): 13-17.

Boucher D.H., M. Hansen, S. Risch et al. 1983. Agriculture. In D.H. Janzen, editor. Costa Rican Natural History. University of Chicago Press, Chicago, IL, USA.

Boza M.A. 1993. Conservation in action - past, present, and future of the national park system of Costa-Rica. Conservation Biology 7: 239-247.

Brandon, K., K.H. Redford, and S.E. Sanderson, editors. 1998. Parks in peril: people, politics, and protected areas. Island Press, Washington, D.C., USA.

Brennan, A., 1995. Ethics, ecology and economics. Biodiversity and Conservation, 4 (8): 798-811.

Brockett C., and R. Gottfried. 2002. State policies and the preservation of forest cover: lessons from contrasting public-policy regimes in Costa Rica. Latin American Research Review 37(1): 7-40.

Browder J.O., and B.J. Godfrey. 1997. Rainforest cities: urbanization, development, and globalization of the Brazilian Amazon. Columbia University Press, NY, NY, USA.

Bruijnzeel, L.A. 2004. Hydrological functions of tropical forests: Not seeing the soil for the trees? Agriculture, Ecosystems, and Environment 104: 185-228.

Bruijnzeel, L.A. 2006. Cloud forests, hydrology and payment for Ecosystem Services in Costa Rica. http://www.iucn.org/themes/cem/news/newsletter/2006/cem_newsletter03_2006.htm.

Bruner, A.G., R.E. Gullison, R.E. Rice and G. Fonseca. 2001. Effectiveness of parks in protecting tropical biodiversity. Science 291: 125-128.

Brown, D.G., P. Goovaerts, A. Burnicki, and M.Y. Li. 2002. Stochastic simulation of land-cover change using geostatistics and generalized additive models. Photogrammetric Engineering & Remote Sensing 68(10): 1051–1061.

Bunn, S.E., P.M. Daview and T.D. Mosisch. 1999. Ecosystem measures of river health and their response to riparian and catchment degradation. Freshwater Biology 41: 333-345.

Campbell, L.M. 2002. Conservation narratives in Costa Rica: Conflict and co-existence. Development and Change 33: 29-56.

Carpenter F.L., J.D. Nichols, and E. Sandi. 2004. Early growth of native and exotic trees planted on degraded tropical pasture. Forest Ecol Manag 196 (2-3): 367-378.

Andre MF (1998) Depopulation, land-use change and transformation in the French Massif Central. Ambio 27: 351-353.

Casetti E, and H.L. Gauthier. 1977. Formalization and Test of Hollow Frontier Hypothesis. Economic Geography 53: 70-78.

Castillo, E.A. 2006. Proyecto Ecomercdos: Consultoria Para Actualizacion de la Propuesta Technica de Ordenamiento Territorial Con Fines de Conservacion de Biodiversidad en Costa Rica (GRUAS II).

Chang, C.R., P.F. Lee, M.L. Bai, and T.T. Lin. 2006. Identifying the scale thresholds for field-data extrapolation via spatial analysis of landscape gradients. Ecosystems 9: 200-214.

Chomitz, K.M., E. Brenes, and L.Constantino. 1999. Financing environmental services: The costa rican experience and its implications. Science of the Total Environment 240: 157-169.

Chomitz, K.M., G.A.B. Fonseca, K. Alger, D.M. Stoms, M. Honzák, E.C. Landau, T.S. Thomas, W.W. Thomas, and F. Davis. 2006. Viable reserve networks arise from individual landholder responses to conservation initiatives. Ecology and Society 11: 40.

Chowdhury, R.R., L.C. Schneider, Y. Ogneva-Himmelberger, P.M. Mendoza, S.C. Villar, and A. Barker-Plotkin. 2004. Land Cover and Land Use: Classification and Change Analysis. Pages 105-141 *in*: B. L. Turner II, J. Geoghegan, D. Foster, editors. Integrated Land-Change Science and Tropical Deforestation in the Southern Yucatán: Final Frontiers. Clarendon Press, Oxford, UK.

CIA, 2007. The World Fact Book – Costa Rica. Available at https://www.cia.gov/cia/publications/factbook/geos/cs.html, accessed on May 2, 2007.

CIFOR,1999. Your subsidies are my incentives. POLEX. Available at http://www.cifor.cgiar.org/Publications/Polex/polexdetail.htm?pid=19, accessed on December 18, 2006.

Cincotta, R.P., J. Wisnewski, and R. Engelman. 2000. Human population density in the biodiversity hotspots. Nature 404: 990-992.

Coomes, O. T., F. Grimard, G. J. Burt. 2000. Tropical forests and shifting cultivation: secondary forest fallow dynamics among traditional farmers of the Peruvian Amazon. Ecological Economics 32: 109-124.

Corbett, E., and R. C. Anderson. 2006. Landscape analysis of Illinois and Wisconsin remnant prairies. Journal of the Torrey Botanical Society 133: 267-179.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387: 253-260.

Costanza, R. 2001. Visions, Values, Valuation, and the Need for an Ecological Economics. BioScience 51(6): 459-468.

Costanza, R. and J. Farley. 2009. Introduction: Theory and Practice of Payments for Ecosystem Services: The Heredia Declaration. Ecological Economics. In press.

Cowan Jr., J.H., and R.E. Turner. 1988. Modeling wetland loss in coastal Louisiana: geology, geography and human modifications. Environmental Management 12: 827-838.

Cumming, G.S. 2000. Using between-model comparisons to fine-tune linear models of species ranges. Journal of Biogeography 27: 441-455.

Daly, H.E. and J. Farley. 2004. Ecological Economics: Principles and Applications. Island Press, Washington, DC, USA.

Daniels, A.E. 2004. Protected area management in a watershed context: a case study of Palo Verde National Park, Costa Rica. Thesis. University of Florida, Gainesville.

Daniels, A.E. 2006. Incorporating domain knowledge and spatial relationships in land cover classifications. International Journal of Remote Sensing 14(20): 2949-2975.

Daniels A.E., G.S. Cumming. 2008. Conversion or conservation? Understanding wetland change in northwest Costa Rica. Ecological Applications 18: 49-63.

Daniels A.E., V. Esposito, K. Bagstad et al. 2009. A decade of PES: building on Costa Rica's model and applying lessons learned. Ecological Economics, special issue.

De Camino, R., O. Segura, L.G. Arias, and I. Pérez. 2002. Costa Rica: Forest Strategy and the Evolution of Land Use. World Bank Evaluation Country Case Study. Series 148. The World Bank, Washington, DC.

Defries, R., A. Hansen, A.C. Newton, and M.C. Hansen. 2005. Increasing isolation of protected areas in tropical forests over the past twenty years. Ecological Applications 15(1): 19-26.

DeFries, R.S., M.C. Hansen, and J.R.G. Townsend. 2000. Global continuous fields of vegetation characteristics: a linear mixture model applied to multi-year 8 km AVHRR data. International Journal of Remote Sensing 21(6-7): 1389-1414.

A. De Janvry. 1981. The agrarian question and reformism in Latin America. The Johns Hopkins University Press, Baltimore.

de Jong W, L. Freitas, J. Baluarte et al. 2001. Secondary forest dynamics in the Amazon floodplain in Peru. Forest Ecology and Management 150:135-146.

DeWitt R.P. 1977. The Inter-American Development Bank and political influence, with special reference to Costa Rica. Praeger, NY, NY, USA.

Eamus, D. 2001. How does water balance influence net primary productivity? A discussion. NEE Proceedings Workshop, pp. 62-70.

Eaton, J. and D. Lawrence. 2006. Woody debris stocks and fluxes during succession in a dry tropical forest. Forest Ecology and Management 232: 46-55.

Edelman M. 1981. Apuntes sobre la consolidacion de las haciendas en Guanacaste. Avance de Investigación No. 44. Instituto de Investigaciones Sociales, Facultad de Ciencias Sociales. Universidad de Costa Rica. San Jose, Costa Rica.

Edelman M. 1985. Extensive Land-use and the Logic of the Latifundio - a Case-Study in Guanacaste Province, Costa-Rica. Human Ecology 13:153-185.

Edelman M. 1992. The logic of the latifundio : the large estates of northwestern Costa Rica since the late nineteenth century. Stanford University Press, Stanford, CA, USA.

Efron, B., and R.J. Tibshirani. 1993. An introduction to the bootstrap. Chapman and Hall. NY, NY, USA.

Estado de la Nacion. 2006. Duodecimo informe estado de la nacion en desarrollo humano sostenible. Programa Estado de la Nacion, San Jose, Costa Rica. (ISBN: 9968-806-37-4).

Ellison, A.M. 2004. Wetlands of Central America. Wetlands Ecology and Management 12: 3-55.

Farley, J., A. Aquino, A. Daniels, D. Lee, and A. Krause. In press. Global mechanisms for sustaining and enhancing PES systems. Ecological Economics.

Ferguson, B. G., J. Vandermeer, H. Morales, and D.M. Griffith. 2003. Post-agricultural succession in El Peten, Guatemala. Conservation Biology 17: 818-828.

Ferraro, P.J. 2001. Global habitat protection: Limitations of development interventions and a role for conservation performance payments. Conservation Biology 15(4): 990-1000.

Fielding, A.H., and J.F. Bell. 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. Environmental Conservation 24(1): 38-49.

Flather, C.H., and M. Bevers. 2002. Patchy reaction – diffusion and population abundance: the relative importance of habitat amount and arrangement. The American Naturalist 159: 40-56.

Forester, D.J., and G.E. Machlis. 1996. Modeling human factors that affect the loss of biodiversity. Conservation Biology 10(4): 1253-1263.

Gibbs, J.P. 2000. Wetland loss and biodiversity conservation. Conservation Biology 14(1): 314-317.

Foster D.R. 1992. Land-use history (1730-1990) and vegetation dynamics in central New-England, USA. Journal of Ecology 80:753-772.

García, R. 1996. Propuesta tecnica de ordenamiento territorial con fines de conservacion de biodiversidad: protyecto GRUAS. Ministerio del Ambiente y Energía (MINAE) and sistema Nacional de Areas de Conservacion (SINAC), Costa Rica.

Godoy, R. A. 1992. Determinants of smallholder commercial tree cultivation. World Development 20: 713-725.

Geist H.J., E.F. Lambin. 2002. Proximate causes and underlying driving forces of tropical deforestation. Bioscience 52:143-150.

Grainger A. 1995. The forest transition: An alternative approach. Area 27:242-251.

Green, G.M., C.M. Schweik, and J.C. Randolph. 2005. Retrieving Land-Cover Change Information from Landsat Satellite Images by Minimizing Other Sources of Reflectance Variability. Pages 131–160 in E.F. Moran, E. Ostrom, editors. Seeing the Forest and the Trees: Human-Environment Interactions in Forest Ecosystems. MIT Press, MA, USA.

Grieg-Gran, M., I. Porras, and S. Wunder. 2005. How can market mechanisms for forest environmental services help the poor? Preliminary lessons from Latin America. World Development 33: 1511-1527.

Haig, S.M., D.W. Mehlman, and L.W. Oring. 1998. Avian movement and wetland connectivity in landscape conservation. Conservation Biology 12(4): 749-758.

Harris, M.B., W. Tomas, G. Mourao, C.J. Da Silva, E. Guimaraes, F. Sonoda, and E. Fachim. 2005. Safeguarding the Pantanal wetlands: threats and conservation initiatives. Conservation Biology 19(3): 714-720.

Hey, D.L., and N.S. Philippi. 1995. Flood reduction through wetland restoration: the Upper Mississippi River Basin as a case history. Restoration Ecology 3: 4-17.

Hoeting, J.A., R.A. Davis, A.A. Merton, and S.E. Thompson. 2006. Model selection for geostatistical models. Ecological Applications 16(1): 87-98.

Holdridge, L.R., 1971. Forest environments in tropical life zones; a pilot study. Pergamon Press, NY, NY, USA.

Huhta, E., J. Jokimaki, and P. Helle. 1998. Predation on artificial nests in a forest dominated landscape—the effects of nest type, patch size and edge structure. Ecography 21(5): 464-471.

Hunter, J.R. 1994. Is Costa Rica truly conservation-minded. Conservation Biology 8: 592-595.

Husson. A., J. Font, H. Jeanjean, C. Miquel, H. Puig, and C. Solier. 1995. Study of Forest/Non-forest Typology of Fragmentation. TREES Series B, Research Report No. 2. European Commission, EUR 16291 EN.

Huston, M.A. 2005. The three phases of land-use change: implications for biodiversity. Ecological Applications 15(6): 1864-1878.

Irwin, E.G., and J. Geoghegan. 2001. Theory, data, methods: developing spatially explicit economic models of land use change. Agriculture, Ecosystems and Environment 85: 7-23.

James P.E. 1959. Latin America. Odyssey Press, NY, NY, USA.

Janzen D.H. 2002. Tropical dry forest: area de conservacion de Guanacaste, northwestern Costa Rica. In M.R. Perrow, and A.J. Davy, editors. Handbook of Ecological Restoration, Volume 2, Restoration in Practice. Cambridge University Press, Cambridge, UK.

Jeng, H. and Y. Hong. 2005. Assessment of a natural wetland for use in wastewater remediation. Environmental Monitoring and Assessment 111(1-3): 113-131.

Jensen, J.R. 2000. Remote Sensing of the Environment: An Herat Resource Perspective. Prentice Hall. Upper Saddle River, NJ, USA.

Jensen, J. R., 1996. Introductory Digital Image Processing: A Remote Sensing Perspective. Prentice Hall, NJ, USA.

Jimenez, J.A., E. Gonzalez, and J. Calvo. 2003. Recomendaciones tecanicas para la restauracion hidrologica del Parque Nacional Palo Verde: casos humedal Palo Verde y La Bocana. http://www.ots.ac.cr/en/paloverde/docs/pv_bocana.pdf.

Johnson, N.L. 2001. Tierra y libertad: will tenure reform improve productivity in Mexico's ejido agriculture? Economic Development and Cultural Change 49(2): 291-309.

Junk, W. 2002. Long-term environmental trends and the future of tropical wetlands. Environmental Conservation 29(4): 414-435.

Killion, T. W., J. A. Sabloff, G. Tourtellot, N. P. Dunning. 1989. Intensive Surface Collection of Residential Clusters at Terminal Classic Sayil, Yucatan, Mexico. Journal of Field Archaeology 16: 273-294.

Kleinn C., L. Corrales, D. Morales. 2002. Forest area in Costa Rica: A comparative study of tropical forest cover estimates over time. Environmental monitoring and assessment 73: 17-40.

Klein, E., E.E. Berg, and R. Dial. 2005. Wetland drying and succession across the Kenai Peninsula Lowlands, south-central Alaska. Canadian Journal of Forest Resources 35: 1931-1941.

Klepeis, P., C. Vance. 2003. Neoliberal policy and deforestation in southeastern Mexico: an assessment of the PROCAMPO program. Economic Geography 79: 221–240.

Klepeis, P. 2003. Development policies and tropical deforestation in the southern Yucatan peninsula: centralized and decentralized approaches. Land Degredation and Development 14: 541-561.

Klepeis, P. 2004. Forest extraction to theme parks: the modern history of land change. Pages 39-62 in B. L. Turner II, J. Geoghegan, D. Foster, editors. Integrated Land-Change Science and Tropical Deforestation in the Southern Yucatán: Final Frontiers. Oxford Geographical and Environmental Studies. Clarendon Press, Oxford, UK.

Klooster, D. 2003. Forest transitions in Mexico: Institutions and forests in a globalized countryside. Professional Geographer 55: 227-237.

Koneff, M.D., and J.A. Royle. 2004. Modeling wetland change along the United States Atlantic Coast. Ecological Modelling 177: 41-59.

Koop G, and L. Tole. 1999. Is there an environmental Kuznets curve for deforestation? Journal of Development Economics 58: 231-244.

Kosoy, N., M. Martinez-Tuna, R. Muradian, and J. Martinez-Alier. 2007. Payments for environmental services in watersheds: Insights from a comparative study of three cases in Central America. Ecological Economics 61(2-3): 446-455.

Krueger, R.C., B. L. Kerans, E. R. Vincent, and C. Rasmussen. 2006. Risk of Myxobolus cerebralis infection to rainbow trout in the Madison River, Montana, USA. Ecological Applications 16: 770-783.

Lambin E.F. 1997. Modelling and monitoring land-cover change processes in tropical regions. Progress in Physical Geography 21: 375-393.

Lambin, E.F., and H.J. Geist. 2001. Global land-use and land-cover change: what have we learned so far? Global Change Newsletter. http://www.geo.ucl.ac.be/LUCC/pdf/Pages%20from%20NL%2046.pdf

Lambin E.F., B.L. Turner, and H.J. Geist. 2001. The causes of land-use and land-cover change: moving beyond the myths. Global Environmental Change 11: 261-269.

Lambin, E.F., B.L. Turner II, H. Geist, S. Agbola, A. Angelsen, J.W. Bruce, O. Coomes, R. Dirzo, G. Fischer, C. Folke, P.S. George, K. Homewood, J. Imbernon, R. Leemans, X. Li,, E.F. Moran, M. Mortimore, P.S. Ramakrishnan, J.F. Richards, H. Skånes, W. Steffen, G.D. Stone, U. Svedin, T. Veldkamp, C. Vogel, and J. Xu. 2001. Our emerging understanding of the causes of land-use and -cover change. Global Environmental Change 11: 261–269.

Landell-Mills, N., and I.T. Porras. 2002. Silver bullet or fool's gold? A global review of markets for forest environmental services and their impact on the poor. International Institute for Environment and Development, London.

Lara, S., T. Barry, and P. Simonson. 1995. Incide Costa Rica—The essential guide to its politics, economy, society and environment. Resource Center Press. Albuquerque, New Mexico, USA.

Latacz-Lohmann, U., C. van der Hamsvoort. 1997. Auctioning conservation contracts: A theoretical analysis and an application. American Journal of Agricultural Economics, 79(2): 407-418.

Latacz-Lohmann, U., S. Schilizzi. 2005. Auctions for conservation contracts: A review of the theoretical and empirical literature. Report to the Scottish Executive Environment and Rural Affairs Department. Project No. UKL/001/05.

Lawrence, D., H. Vester, D. Pérez-Salicrup, R. Eastman, B. L. Turner II, and J. Geoghegan. 2004. Integrated Analysis of Ecosystem Interactions with Land-Use Change: The Southern Yucatán Peninsular Region. Pages 277-292 in R. DeFries, G. Asner, and R. Houghton, editors. Ecosystem Interactions with Land Use Change. American Geophysical Union, Washington, D.C., USA.

Lawrence, D.C., R. Peart, and M. Leighton. 1998. The impact of shifting cultivation on a rainforest landscape in West Kalimantan: spatial and temporal dynamics. Landscape Ecology 13: 135-148.

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

Lawrence, D., and D. Foster. 2004. Recovery of Nutrient Cycling and Ecosystem Properties following Swidden Cultivation: Regional and Stand-Level Constraints. Pages 81-104 in B. L. Turner II, J. Geoghegan, D. Foster, editors. Integrated Land-Change Science and Tropical Deforestation in the Southern Yucatán: Final Frontiers, Clarendon Press, Oxford, UK.

Legendre, P., and L. Legendre. 1998. Numerical Ecology (2nd Ed.). Elsevier, NY, NY, USA.

Lemly, A.D., R.T. Kingsford, and J.R. Thompson. 2000. Irrigated agriculture and wildlife conservation: conflict on a global scale. Environmental Management 25(5): 485-512.

Leonard H.J. 1986. Recursos naturals y desarrollo economico en America Central: un perfil ambiental regional. Seria tecnica, informe tecnico/CATIE, No. 127.

Liu, A.J., and G.N. Cameron. 2001. Analysis of landscape patterns in coastal wetlands of Galveston Bay, Texas, USA. Landscape Ecology 16(7): 581-595.

Lutz, E., and H. Daly. 1991. Incentives, regulations and sustainable land use in Costa Rica. Environmental and Resource Economics 1: 179-194.

Lutz, W., L. Prieto, and W. Sanderson. 2000. Population, Development and Environment on the Yucatan Peninsula. International Institute for Applied Systems Analysis. Research Report RR-00-14.

Marsik, M., and P. Waylen. 2006. An application of the distributed hydrologic model CASC2D to a tropical montane watershed. Journal of Hydrology 330: 481-495.

Mas, J.F., H. Puig, J.L. Palacio, and A. Sosa-Lopez. 2004. Modelling deforestation using GIS and artificial neural networks. Environmental Modeling and Software 19(5): 461-471.

Mas, J.F. 2005. Assessing protected area effectiveness using surrounding (buffer) areas environmentally similar to the target area. Environmental Monitoring and Assessment 105(1-3): 69-80.

Mateo-Varga, J. 2001. Caracteristicas generales de la cuenca del Rio Tempisque. Pages 32-72 in J.A. Jiménez and E. Gonzalez, editors. La Cuenca del Rio Tempisque, Perspectivas para un Manejo Integrado. Organización para Estudios Tropicales, San Jose, Costa Rica.

Mather A.S. 1992. The Forest Transition. Area 24: 367-379.

Mather A.S., and C.L. Needle. 1998. The forest transition: a theoretical basis. Area 30: 117-124.

Mather A.S., J. Fairbairn, and C.L. Needle. 1999. The course and drivers of the forest transition: the case of France. Journal of Rural Studies 15: 65-90.

McCoy, M.B., and J.M. Rodriguez. 1994. Cattail (Typha dominguensis) eradication methods in the restoration of a tropical, seasonal, freshwater marsh. In W.J. Mitsch, editor. Global Wetlands: Old World and New. Elsevier, NY, NY, USA.

McGarigal, K., S. Cushman, and S. Stafford. 2000. Multivariate Statistics for Wildlife and Ecology Research. Springer-Verlag, York, PA, USA.

Meijerink, A., A. Gieske, and Z. Vekerdy. 2005. Surface energy balance using satellite data for the water balance of a traditional irrigation-wetland system in SW Iran. Irrigation and Drainage Systems 19(1): 89-105.

Mertens, B. and E.F. Lambin. 1997. Spatial modeling of deforestation in southern Cameroon. Applied Geography 17(2): 143-162.

Mertens, B. and E.F. Lambin. 2000. Land cover change trajectories in southern Cameroon. Annals of the Association of American Geographers 90(3): 467-494.

Meyer W.B., and B.L. Turner. 1992. Human-Population Growth and Global Land-use Cover Change. Annual Review of Ecology and Systematics 23: 39-61.

Millennium Ecosystem Assessment. 2005. Ecosystems and Human Well-being: Wetlands and Water Synthesis. World Resources Institute. Washington, D.C., USA. http://www.millenniumassessment.org//proxy/Document.

Miller, P. M. 1999. Effects of deforestation on seed banks in a tropical deciduous forest of western Mexico. Journal of Tropical Ecology 15: 179-188.

Miranda, M., C. Dieperink, P. Glasbergen. 2006. Costa Rican environmental service payments: the use of a financial instrument in participatory forest management. Environmental Management 38: 562-571.

Mitsch, W.J. 2005. Applying science to conservation and restoration of the world's wetlands. Water Science & Technology 51(8): 13-26.

Mitsch, W.J., and J.G. Gosselink. 2000. The value of wetlands: importance of scale and landscape setting. Ecological Economics 35: 25-33.

Mitra, S., R. Wassman, and P.L. Vlek. 2005. An appraisal of global wetland area and its organic carbon stock. Current Science 88(1): 25-35.

Moran, E.F. 2004. Inferring the Behavior of Households from Remotely Sensed Changes in Land Cover: Current Methods and Future Directions. Pages 23-47 in M.F. Goodchild editor. Spatially Integrated Social Science. Oxford University Press, Oxford, UK.

Mulder, M., and P. Coppolillo. 2005. Conservation: Linking Ecology, Economics and Culture. Princeton University Press, Princeton, NJ, USA.

Murphy, P. G., and A. E. Lugo. 1986. Ecology of Tropical Dry Forest. Annual Review of Ecology and Systematics 17: 67-88.

Mustard J.F., R.S. Defries, T. Fisher, and E.F. Moran. 2004. Land-use and land-cover change pathways and impacts. Pages 411-429 in G. Gutma, A.C. Janetos, C.O. Justice, E.F. Moran, J.F. Mustard, R.R. Rindfuss, D. Skole, B.L. Turner II, and M.A. Cochrane, editors. Land change science: observing, monitoring and understanding trajectories of change on the earth's surface. Kluwer, Dordrecht, The Netherlands.

Myers, N. 1993. Tropical Forests - the Main Deforestation Fronts. Environmental Conservation 20: 9-16.

Nagelkerke, N.J. 1991. A note on a general definition of the coefficient of determination. Biometrika 78(3): 691-692.

Nagendra, H. 2001. Using remote sensing to assess biodiversity. International Journal of Remote Sensing 22: 2377-2400.

Naidoo, R., and T.H. Ricketts. 2006. Mapping the economic costs and benefits of conservation. PLOS Biology 4(11): 2153-2164.

Neupane, R. P., K. R. Sharma, and G. B. Thapa. 2002. Adoption of agroforestry in the hills of Nepal: a logistic regression analysis. Agricultural Systems 72: 177-196.

Norusis, M. 1990. SPSS Advanced Statistics User's Guide, SPSS Inc., Chicago, IL.

Notman, E., Gorchov, D.L., 2001. Variation in post-dispersal seed predation in mature Peruvian lowland tropical forest and fallow agricultural sites. Biotropica 33, 621-636.

Ochoa-Gaona, S., 2001. Traditional land-use systems and patterns of forest fragmentation in the highlands of Chiapas, Mexico. Environmental Management 27: 571-586.

Ogawa, H., and J.W. Male. 1986. Simulating the flood mitigation role of wetlands. ASCE Journal of Water Resources Planning and Management 112: 114-127.

O'Grady, M.A. 2006. Costa Rican poverty fighter. The Wall Street Journal. May 5, 2006.

Olson, M. 1965. The Logic of Collective Action. Harvard University Press, Cambridge, MA.

Oviedo, E. 2006. Estado paga poco por conservar bosques. La Nacion, 20 de noviembre.

Ozesmi, S.L., C.O. Tan, and U. Ozesmi. 2006. Methodological issues in building, training, and testing artificial neural networks in ecological applications. Ecological Modelling 195: 83-93.

Ozesmi, S.L., and M.E. Bauer. 2002. Satellite remote sensing of wetlands. Wetlands Ecology and Management 10(5): 381-402.

Pagiola, S., A. Arcenas, and G. Platais. 2005a. Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. World Development 33: 237-253.

Pagiola, S., P. Agostini, J. Gobbi, C. de Haan, M. Ibrahim, E. Murgueitio, E. Ramirez, M. Rosales, and J. P. Ruiz. 2005b. Paying for biodiversity conservation services - Experience in Colombia, Costa Rica, and Nicaragua. Mountain Research and Development 25: 206-211.

Pagiola, S. 2006. Payments for environmental services in Costa Rica. MPRA Paper No. 2010. Revised version of a paper presented at the ZEF-CIFOR workshop on Payments for environmental services: Methods and design in developing and developed countries, Titisee, Germany, July 15-18 2005.

Perez-Salicrup, D. R., S. Schnitzer, and F. E. Putz. 2004. Community ecology and management of lianas. Forest Ecology and Management 190: 1-2.

Perz, S.G. 2007. Grand theory and context-specificity in the study of forest dynamics: Forest transition theory and other directions. Professional Geographer 59: 105-114.

Peters, G. 2001. La cuenca del Tempisque: una perspective historica. Pages 1-21 in J. Jimenez and E. Gonzalez, editors. La Cuenca del Rio Tempisque: Perspectivas para un Manejo Integrado. Organizacion para Estudios Tropicales, San Jose, Costa Rica.

Pijanowski, B.C., D.G. Brown, B.A. Shellito, and G.A. Manik. 2002. Using neural networks and GIS to forecast land use changes: a land transformation model. Computers, Environment and Urban Systems 26: 553-575.

Plaza, C.R. 2000. Gender roles, inheritance patterns, and female access to land in an ejidal community in Veracruz, Mexico. Pages 161-173 in A. Zoomers, G. van der Haar, editors. Current land policy in Latin America: regulating land tenure under neo-liberalism. Royal Tropical Institute, Amsterdam, The Netherlands.

Polasky, S., J.D. Camm, and B. Garber-Yonts. 2001. Selecting biological reserves cost-effectively: An application to terrestrial vertebrate conservation in Oregon. Land Economics 77(1): 68-78.

Pontius, R.G., and L.C. Schneider. 2001. Land cover change model validation by an ROC method for the Ipswich watershed, Massachusetts, USA. Agriculture, Ecosystem and Environment 85: 239-248.

Pringle, C.M. 2001. Hydrologic connectivity and the management of biological reserves: a global perspective. Ecological Applications 11(4): 981-998.

Pringle, C. 2003. What is hydrologic connecitivty and why is it ecologically important? Hydrological Processes 17(13): 2685-2689.

Ramankutty N, H.K. Gibbs, and F. Achard. 2007. Challenges to estimating carbon emissions from tropical deforestation. Global Change Biology 13: 51-66.

Ramankutty, N., L. Graumlich, F. Achard, D. Alves, A. Chhabra, R. DeFries, J. Foley, H. Geist, R. Houghton, K. Goldewijk, E.F. Lambin, A. Millington, K. Rasmussen, R. Reid, and B.L. Turner II. 2006. Global Land Cover Change: Recent Progress, Remaining Challenges. Pages 9-40 in E. Lambin, and H. Geist, editors. Land Use and Land Cover Change: Local Processes, Global Impacts. Springer Verlag, NY, NY, USA.

Randall, A. 1993. The Problem of Market Failure. In: Economics of the Environment. R. Dorfman and N. Dorfman, eds. Norton, New York. Rodriguez, J. P., Beard, T. D.,

Bennett, E. M., G. S. Cumming, S. J. Cork, J. Agard, A. P. Dobson, and G. D. Peterson. 2006. Trade-offs across space, time, and ecosystem services. Ecology and Society 11: 28.

Ravan, S., A.M. Dixit, and V.B. Mathur. 2005. Spatial analysis for identification and evaluation of forested corridors between two protected areas in Central India. Current Science 88(9): 1441-1448.

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

Reyes, E., M.L. White, J.F. Martin, G.P. Kemp, J.W. Day, and V. Aravamuthan. 2000. Landscape modeling of coastal habitat change in the Mississippi delta. Ecology 81: 2331-2349.

Rogerson, P.A. 2001. Statistical Methods for Geography. Sage Publications. Thousand Oaks, CA, USA.

Rojas, M., and B. Aylward. 2003. What are we learning from experiences with markets for environmental services in Costa Rica? A review and critique of the literature. International Institute for Environment and Development, London, UK.

Rouget M, D.M. Richardson, and R.M. Cowling. 2003. The current configuration of protected areas in the Cape Floristic Region, South Africa - reservation bias and representation of biodiversity patterns and processes. Biological Conservation 112: 129-145.

Rudel T.K., O.T. Coomes, E. Moran. 2005. Forest transitions: towards a global understanding of land use change. Global Environmental Change 15:23-31.

Rudel, T.K., O. Coomes, E.F. Moran, A. Angelsen, F. Achard, E. Lambin, and J. Xu. 2005. Forest transitions: towards an understanding of global land use change. Global Environmental Change 15(1): 23-31.

Salzman, J., and J.B. Ruhl. 2002. Paying to protect watershed services: Wetland banking in the United States. In S. Pagiola, and J. Bishop, editors. Selling Forest Ecosystem Services: Market-based mechanisms for conservation and development. Earthscan, London, UK.

Salzman, J. 2005. Creating markets for ecosystem services: Notes from the field. New York University Law Review 80: 870.

Samuelson, P.A. 1954. The pure theory of public expenditure. The review of Economics and Statistics 36(4): 387-389.

Sanchez-Azofeifa, G.A., G.C. Daily, A. Pfaff, and C. Busch. 2003. Integrity and isolation of Costa Rica's nacional parks and biological reserves: examining the dynamics of land-cover change. Biological Conservation 109: 123-135.

Sanchez-Azofeifa G.A., A. Pfaff, J.A. Robalino, et al. 2007. Costa Rica's payments for environmental services program: intention, implementation and impact. Conservation Biology 21: 1165-1173.

Sanchez-Azofeifa, G., R. C. Harriss, and D. L. Skole. 2001. Deforestation in Costa Rica: A quantitative analysis using remote sensing imagery. Biotropica 33: 378-384.

Schaafsma, W. and G.N. van Vark. 1979. Classification and discrimination problems with applications. Part IIa. Statistica Neerlandica 33: 91-126.

Semlitsch, R.D. 2002. Critical elements for biologically based recovery plans of aquatic-breeding amphibians. Conservation Biology 16(3): 619-629.

Sierra, R., E. Russman. 2006. On the efficiency of environmental service payments: A forest conservation assessment in the Osa Peninsula, Costa Rica. Ecological Economics 59(1): 131-141.

Sills, E., G. Hartshorn, P. Ferraro, and B. Spergel. 2005. Evaluation of the World Bank-GEF ecomarkets project in Costa Rica. Blue Ribbon Panel Report. North Carolina State University, Durham, NC, USA.

Skole D.L., W.H. Chomentowski, W.A. Salas et al. 1994. Physical and Human Dimensions of Deforestation in Amazonia. Bioscience 44: 314-322.

Sloan, S. 2007. Fewer people may not mean more forest for Latin American forest frontiers. Biotropica 39(4): 443-446.

Southworth, J. 2004. An assessment of Landsat TM band 6 thermal data for analysing land cover in tropical dry forest regions. International Journal of Remote Sensing 25: 689-706.

Southworth J., D.K. Munroe, and H. Nagendra. 2004. Land cover change and landscape fragmentation—comparing the utility of continuous and discrete analyses for a western Honduras region. Agriculture, Ecosystems and Environment, 101(2-3): 185-205.

Southworth J, H. Nagendra, L.A. Carlson et al. 2004. Assessing the impact of Celaque National Park on forest fragmentation in western Honduras. Applied Geography 24: 303-322.

Staaland H., O. Holand, C. Nellemann, et al. 1998. Time scale for forest regrowth: Abandoned grazing and agricultural areas in southern Norway. Ambio 27: 456-460.

Steed, B. 2003. Completing the mosaic: the conservation of private lands in Costa Rica. Journal of Land, Resources, and Environmental Law 173: 173-180.

Stern P.C., O.R. Young, and D. Druckman. 1992. Global environmental change: understanding the human dimensions. National Academy Press, Washington, D.C., USA.

Stoms, D.M., K.M. Chomitz, and F.W. Davis. 2004. TAMARIN: a landscape framework for evaluating economic incentives for rainforest restoration. Landscape and Urban Planning 68: 95-108.

Stoneham, G., V. Chaudri, A. Ha, and L. Strappazon. 2003. Auctions for conservation contracts: An empirical examination of Victoria's BushTender trial. Australian Journal of Agricultural and Resource Economics 47(4): 477-500.

Szentandrasi, S., S. Polasky, R. Berrens, J. Leonard. 1995. Conserving biological diversity and the conservation reserve program. Growth and Change 26: 383-404.

Summerville, K. S., C. J. Conoan, and R. M. Steichen. 2006. Species traits as predictors of lepidopteran composition in restored and remnant tallgrass prairies. Ecological Applications 16: 891-900.

Taylor J.E. 1980. Peripheral capitalism and rural-urban migration: a study of population movements in Costa Rica. Latin American Perspectives 7(2-3): 75-90.

Terborgh, J. 1999. Requiem for Nature. Island Press. Washington, D.C., USA.

Teran, S., and C. H. Rasmussen. 1995. Genetic Diversity and Agricultural Strategy in 16th-Century and Present-Day Yucatecan Milpa Agriculture. Biodiversity and Conservation 4: 363-381.

Tietenberg, T.H. 1989. Economic instruments for environmental regulation. Oxford Review of Economic Policy 6(1): 17-33.

Tiner, R.W. 2005. Assessing cumulatie loss of wetland functions in the Anticoke River Watershed using enhanced national wetlands inventory data. Wetlands 25(2): 405-419.

Toner, M., and P. Keddy. 1997. River hydrology and riparian wetlands: a predictive model for ecological assembly. Ecological Applications 7(1): 236-246.

Trebitz, A.S., J.A. Morrice, D.L. Taylor, R.L. Anderson, C.W. West and J.R. Kelly. 2005. Hydromorphic determinants of aquatic habitat variability in Lake Superior coastal wetlands. Wetlands 25(3): 505-519.

Turner, M.G., R.H. Gardner, R.V. O'Neill. 1995. Ecological Dynamics at Broad Scales. BioScience 45: S29-S35.

Turner II, B.L., J. Geoghegan, D. Foster. 2004. Integrated Land-change Science and Tropical Deforestation in the Southern Yucatán: Final Frontiers. Clarendon Press, Oxford, UK.

Universidad de Costa Rica. 1976. La poblacion de Costa Rica. Instituto de Investigaciones Sociales. Editorial Universidad de Costa Rica: 90-91. San Jose, Costa Rica.

Unruh, J. D. 1988. Ecological Aspects of Site Recovery under Swidden-Fallow Management in the Peruvian Amazon. Agroforestry Systems 7: 161-184.

Unruh, J.D. 1990. Iterative increase of economic tree species in managed swidden-fallows of the Amazon. Agroforestry Systems 11(2): 175-197.

Urban, D.L. 2005. Modelling ecological processes across scales. Ecology 86(8): 1996-2006.

Vance, C. and J. Geoghegan. 2004. Modeling the determinants of semi-subsistance and commercial land uses in an agricultural frontier of southern Mexico: a switching regression approach. International Regional Science Review 27(3): 326-347.

Vogeler, I.K. 1970. Frontier settlements in south-eastern Campeche: A report for the 1970 National Geographic Society-Tulane University Archaelogical Project at Becan, Campeche, Mexico.

Vogeler I.K. 1976. The Dependency Model Applied to a Mexican tropical frontier region. Journal of Tropical Geography 43: 63-68.

Vorosmarty, C.J., and D. Sahagian. 2000. Anthropogenic disturbance of the terrestrial water cycle. BioScience 50(9): 753-765.

Walker, R.T. 1999. The Structure of Uncultivated Wilderness: Land Use beyond the Extensive Margin. Journal of Regional Science 39(2): 387-410.

White, A. and A. Martin. 2002. Who owns the world's forests? Forest tenure and public forests in transition. Forest Trends. Washington, D.C., USA.

Watson, V., S. Cervantes, C. Castro, L. Mora, M. Solis, I.T. Porras, and B. Cornejo. 1998. Making Space for Better Forestry: Costa Rica Country Study. Policy that Works for Forests and People Series No. 6. International Institute for Environment and Development, London, UK.

White, D., P.G. Minotti, M.J. Barczak, J.C. Sifneos, K.E. Freemark, M.V. Santelmann, C.F. Steinitz, A. Ross Kiester, and E.M. Preston. 1997. Assessing risk to biodiversity from future landscape change. Conservation Biology 11: 349-360.

World Bank. 1995. Mexico resource conservation and forest sector review. Natural Resoruces & Rural Poverty Operations Division Country Department II Latin America & the Caribbean Regional Office, Washington, D.C., USA.

World Bank. 2000. Costa Rica Ecomarkets Project. Project Appraisal. Environmentally and Socially Sustainable Development, Latin America and the Caribbean Regional Office, Washginton, D.C., USA.

World Bank. 2006. Scaling up and mainstreaming payment for environmental services in Costa Rica. Project Appraisal. Environmentally and Socially Sustainable Development, Latin America and the Caribbean Regional Office, Washginton, D.C., USA.

World Bank. 2007. Key Development Data & Statistics. Available at http://web.worldbank.org/WBSITE/EXTERNAL/DATASTATISTICS/, accessed May 2, 2007.

Wunder, S. 2005. Payments for environmental services: Some nuts and bolts. CIFOR Occasional Paper No. 42. Center for International Forestry Research, Jakarta, Indonesia.

Wunder, S. 2007. The Efficiency of Payments for Environmental Services in Tropical Conservation. Conservation Biology, 21(1): 48-58.

Wünscher, T., S. Engel, and S. Wunder. 2007. Spatial targeting of payments for environmental services: A tool for boosting conservation benefits. Presented at the Workshop on Carbon Sequestration in Agriculture and Forestry conference, Thesaloniki, Greece, June 27, 2007.

Zbinden, S., and D. R. Lee. 2005. Paying for environmental services: An analysis of participation in Costa Rica's PSA program. World Development 33: 255-272.

Zedler, J.B., and S. Kercher. 2005. Wetland resources: status, trends, ecosystem services and restorability. Annual Review of Environmental Resources 30: 39-74.

# BIOGRAPHICAL SKETCH

Amy Daniels grew up in Apalachicola, FL, graduating from Apalachicola High School in 1995. She went on to Wesleyan College to earn her B.A. in biology in 1999. In the years between college and graduate school, Amy worked as a resource management biologist for the Apalachicola National Estuarine Research Reserve (ANERR) in Florida and as a counselor for a non-profit AIDS-care foundation in New York. She also traveled independently throughout Central America, learning Spanish and volunteering for a grassroots conservation organization in Guatemala. During her master's degree Amy received a Fulbright Fellowship to investigate landscape transformation in northwest Costa Rica and a Coca-cola World Citizenship Fellowship to work as an agricultural development consultant for a United States Agency for International Development-funded project in El Salvador. Amy graduated with an M.S. in Interdisciplinary Ecology in May 2004. Deciding that the graduate-school lifestyle was too great to give up, Amy began her PhD the following year. Amy had the privilege of independently teaching Physical Geography two semesters during her doctoral studies, a rewarding experience that helped her discover a great love for teaching. After graduating Amy received a Presidential Management Fellowship to work in the Global Change Program of the United States Forest Service's Research and Development Office.

Amy is married to a wonderful attorney-turned-diplomat, Paul Ghiotto. Together they enjoy traveling, canoeing, spending time with family and friends, and trying just about anything new.