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Deforestation pressure and biological reserve planning: a conceptual approach and an illustrative application for Costa Rica

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Abstract

An index of 'deforestation pressure' is suggested as useful for reserve planning alongside the currently used information on the species present at candidate sites. For any location, the index value is correlated with threats to habitat and thus also survival probabilities over time for members of species dependent on that habitat. Threats in the absence of reserves are key information for planning new reserves. The index is estimated using a regression approach derived from a dynamic, micro-economic model of land use, with data on observed clearing of forest over space and time as well as biophysical and socioeconomic factors in land returns. Applying an estimated threat (or probability of clearing) function for Costa Rica to locations of interest yields relevant estimates of sites' deforestation pressure, which are used to evaluate proposed reserves and to suggest other candidate sites.

JEL classification: Q15; R14; Q28; H4

Keywords: Deforestation; Reserve; Species; Conservation

1. Introduction

Most of the early literature on reserve-site selection focuses on the benefits of a reserve, in units of species. In particular, most analyses took a given number of sites or total area to be protected and then, through the choice of sites, tried to maximize species benefits,

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at times using a weighted objective to give greater weight to, for instance, rare species.[1] Alternatively, some analyses tried to minimize the total number of sites protected, i.e. minimize the cost of the reserves required to attain a given species benefit.[2] Even then, the informational focus was the benefits, i.e. species, in the sense that it was benefits that were differentiated across sites. In terms of the costs, all of the candidate sites were considered to be equal, so 'a site is a site'.[3]

Even a minimal review of such analyses of benefits from reserves should differentiate the 'scoring', 'iterative', and 'programming' approaches taken to the benefit maximization problem. The scoring approach values each site according to its contribution to the objective and then for new reserves proceeds down the ordered list of sites as the available resources permit.[4] This allows inefficient duplication of species which exist in multiple sites. Iterative analyses avoid that by ranking sites according to marginal gains, or gains conditional on sites already protected. Here the top-ranked site is protected but then other sites are re-ranked in light of all previous protection choices.[5] The programming approach replaces these 'heuristics' with standard operation research optimization techniques.[6] Unlike the sequential approaches, by comparing the whole sequence of site choices, this allows the earlier choices to be evaluated in light of the other (i.e. later) optimal site choices.

More recently, authors (often including economists) have started to focus more on costs, differentiating across candidate sites not only the benefits but also the costs of creating a reserve. For instance, Ando et al. (1998) and Polasky et al. (2001) demonstrate the potential empirical importance of the cost per unit area differing substantially across candidate reserve sites. Using a natural measure of per unit cost, land values, these papers show that focusing solely on benefits and ignoring reserve costs may waste significant resources in attaining a given species benefit or, alternatively, may fail significantly to maximize the total species benefit for a given level of cost.

All of the discussion above is deterministic. Uncertainty and probabilities have entered the site choice literature somewhat recently as well, not surprisingly concerning species benefits. Current species presence is often uncertain for a site, as field observations rarely exist for every species of interest for each candidate site. Species presence may be explicitly estimated, in fact. Thus, current presence data can be probabilities inferred from presence or absence observations, and at time predicted by measures of habitat suitability.[7] While such probabilities are sometimes converted by an arbitrary rule to binary 'presence/absence', Polasky et al. (2000) argue for instead being open about this uncertainty and then explicitly maximizing the expected species presence.

[1] The conclusion of Polasky et al. (2001), for instance, considers different rationales for such varied weights.

[2] This extensive and growing literature relies on two assumptions we will maintain: first, reserves are likely to be used within species conservation, perhaps alongside other approaches such as captive breeding and translocation; and second, given finite resources available for species conservation, it is pragmatic to get "most bang for the buck", i.e. to maximize conservation for a given cost or to minimize costs for a given set of benefits.

[3] Polasky et al. (2000) and Camm et al. (2002), among others, convey some history of this literature.

[4] See Tubbs and Blackwood (1971), Gehlbach (1975), and Williams (1980), for examples of this approach.

[5] Kirkpatrick (1983) and Saetersdal et al. (1993), among others, provide discussion of this approach.

[6] Cocks and Baird (1989), Church et al. (1996), and Csuti et al. (1997) all provide good examples of this approach.

[7] See the discussion within, for instance, Nicholls (1989) and Margules and Stein (1989).

This paper suggests and provides a second probability with a role to play in site selection: the probability of habitat vanishing over time, due to land-use choices, if a reserve is not created. All else equal, a higher threat that a site's habitat and thus also its species will vanish over time should indicate a higher priority site. If it were certain that all of a site's habitat would remain in the absence of a reserve, then there would be no benefit from a reserve at that site. But if habitat and species were sure to vanish within 5 years in the absence of a reserve, then there would be a clear gain from a reserve that reduced the chance that the habitat and all the species in question would vanish. Thus, candidate sites under greater 'pressure' should be higher reserve priorities.

A simple formalization is useful for organizing the discussion to follow. Taking as known the current presence at site i of species j ($B_{ij} \in \{0, 1\}$), consider the net benefit from a new reserve. There is a cost C_i of creating a reserve at site i to reduce the probability of habitat vanishing (P_{it}) from $P_{it\mathrm{N}}$ with no reserve ($R_{it} = 0$) to $P_{it\mathrm{R}}$ with a reserve ($R_{it} = 1$), where effective reserves would mean all $P_{it\mathrm{R}}$ equal zero. The reserve also affects future probabilities of species presence, EB_{ijt}, which fall with the probability of habitat vanishing at site i. For simplicity, $EB_{ijt} = B_{ij}\prod_t(1 - P_{it})$.

Various objectives can be proposed but a reserve-site-selection problem could be written as $\max_{it\mathrm{R}} \sum_{i,j,t} kEB_{ijt} - C_{it}(R_{it})$, where k is a parameter for species valuation, uniform here, and C_{it} the cost function includes not only cost C_i of creating a reserve but also costs of maintenance. The roles of the R_{it} are more explicit if $P_{it}(R_{it})$ within EB_{ijt} is written as $P_{it\mathrm{N}}(1 - R_{it}) + P_{it\mathrm{R}} R_{it}$.[8]

Creating a reserve R_{it} has costs C_{it} but generates expected benefits of the type $\sum_{j,t} (P_{it\mathrm{N}} - P_{it\mathrm{R}}) B_{ij}$. As the $P_{it\mathrm{R}}$ are bounded below by zero, the threats without reserves $P_{it\mathrm{N}}$ bound reserves' benefits. All else equal, sites with high probabilities of habitat vanishing ($P_{it\mathrm{N}}$) should be higher priorities. Alongside information on current species presence (B_{ij}), data on the $P_{it\mathrm{N}}$ can improve site choice.

Below a method is suggested for estimating an index of 'deforestation pressure', i.e. $P_{it\mathrm{N}}$, following Kerr et al. (2003). The index is generated using an econometric analysis of forest clearing observations over space and time, following a dynamic model of land-use choice. Explanatory variables include measures of biophysical and socioeconomic factors that affect the returns from land uses and are observable, as well as a set of observable proxies for factors that affect returns but are harder to observe, such as adjustment costs, the status of national economic and institutional development, and a local knowledge of soil quality and thus also output yields.

Next we apply the econometric analysis of Costa Rican deforestation provided by Kerr et al. (2003). We use their estimated deforestation pressure function to generate an index of $P_{it\mathrm{N}}$ for all candidate sites. Before using it to say sites are priorities, however, we must consider whether this index is also correlated with other elements of the site-selection problem. Specifically, the same observable site characteristics which affect $P_{it\mathrm{N}}$ may also affect the threat of habitat vanishing within a reserve, $P_{it\mathrm{R}}$, and the costs C_{it} of creating and maintaining a reserve. While our formulation above makes clear that data on these two

[8] The EB_{ijt} for any given site could also be affected by reserves at other sites ($R_{-i,t}$), for instance if each site faces an idiosyncratic loss of species presence independent of habitat but can be re-colonized from nearby reserves.

parameters would also be useful, alongside species information, we conclude that our index is best used as an estimate of $P_{it\mathrm{N}}$.

Then we consider specifically the GRUAS corridors proposed by a Costa Rican agency to conserve different types of ecosystems and thus biological diversity. Deforestation pressure varies across the proposed locations. All else equal, this suggests which should be the priorities. Surely, species benefits vary across these sites, but that pressure also varies should affect choices.

We also compare the GRUAS locations to other potential sites for reserves. The corridors are in areas with pressure slightly higher than average pressure. The lack of a big difference in pressure is consistent with ecological rationales previously posited for the corridor locations (Garcia, 1997). To demonstrate an alternative approach, or more precisely one additional rationale that could be used in future proposals, we also indicate some areas of relatively high deforestation pressure.

The paper proceeds as follows. Section 2 presents information on the Costa Rican setting and reserve network, including existing proposals for biodiversity corridors. Section 3 describes the estimation of a deforestation pressure function from observed forest transitions in light of our dynamic land-use model, following Kerr et al. (2003). Section 4 considers the interpretation of the estimated pressure index, compares deforestation pressure across the proposed corridor locations, and compares corridors to other forest areas that could be protected instead. Section 5 concludes.

2. Costa Rican setting and conservation policies

2.1. Historical background

From the arrival of the Spanish until the end of the 1950s and the beginning of the 1960s, Costa Ricans cleared thousands of hectares of forest for agriculture and for cattle production (see Sader and Joyce, 1988; Sanchez-Azofeifa et al., 2001). This resulted in part from official policies placing a priority on demographic growth and agricultural production (see, for example, Harrison, 1991; Solorzano et al., 1991; Rosero-Bixby and Palloni, 1998; Sanchez-Azofeifa et al., 1999). Where the clearing occurred appeared to depend upon biophysical features such as where coffee can and cannot grow, due to variations in soil quality and precipitation, or the natural location for a port.

Little clearing occurred outside of the central plateau and a western port until the 1950s and 1960s. Then trade between Costa Rica and the rest of the world started to rise, exposing the economy to changing international prices in various commodities. Expansion given strong prices occurred for cattle in the north, coffee in the center and, as multinationals expanded, also banana plantations in the Atlantic region. Sugar cane plantations geared to exports and internal markets also arose. Again, such production and its effect on forests varied over space within Costa Rica in part because of ecological constraints such as variations in precipitation and temperature.

The last two decades have brought a slowing of deforestation due in part to falling output prices inducing, for instance, significant abandonment of cattle across the Guanacaste Peninsula (Sanchez-Azofeifa, 2000). Relative returns have also changed because of

Table 1
Establishment dates and characteristics of 132 protected areas

Category	Number	Area (ha)	Number started per decade					National territory[a] (%)
			<1960s	1960s	1970s	1980s	1990s	
National parks	24	541,576	1	1	11	1	10	10.6
Biological preserves	9	39,644	–	–	5	2	2	0.8
National wildlife refuges	39	181,018	–	–	–	9	30	3.5
Forestry reserves	12	291,513	–	2	6	1	3	5.7
Protection zones	31	178,677	–	–	10	11	10	3.5
Wetlands	14	50,465	–	–	1	1	12	1.0
Special categories	3	1,650	–	1	1	–	1	<0.1
Total	132	1,284,543	1	4	34	25	68	25.1

[a] Percent of the national territory within these types of protected areas.

increased returns from forested land. One important change is the rise in ecotourism since the early 1990s. Government is the dominant provider of forest ‘eco-services’, and also has developed a program of payments for multiple environmental services which supports forested land uses. Forest returns also exist in ‘sustainable forestry’ and ‘shade coffee’, in part due to timber and coffee labeling programs.

2.2. Reserves network

Costa Rica has created a significant system of protected areas (see Table 1). Between 1974 and 1978 the areas protected expanded from 3 to 12% of the national territory, and now stand at 25% (Castro-Salazar and Arias-Murillo, 1998).

Innovation has been ongoing. Since 1979, three Forestry Laws (1979, 1986, and 1996) were enacted, and a legal definition of conservation was written in the 1998 Biodiversity Law. Further, agency structure has changed. Prior to 1995, three agencies (Forestry, National Parks, and Wildlife) were responsible for achieving conservation. In 1995, a SINAC (National System of Conservation Areas) was created. Consolidation involved the agencies and also the reserves. SINAC placed all reserves into 11 conservation areas which form the reserve structure.[9]

The proposed GRUAS biodiversity corridors (Fig. 1) resulted from government efforts to identify national priorities for sites on which the state could invest in biodiversity protection, including as part of the Mesoamerican Biological Corridor (Powell et al., 2000). The status of the existing protected areas was reviewed and then modifications of the current set of

[9] Not all reserves enjoy the same protection status. “Level 1”, to this point a status applied to the national parks and biological reserves, indicates that no land-cover change can occur within the reserve. A “Level 2” status, in contrast, permits land-cover changes of some types within reserves. It has been applied to forest reserves and wildlife refuges. Note that within our estimation below we drop the Level 1 areas from the candidate sites for clearing. Even though these lands have not all been paid for after being designated as reserves, i.e. government owes private land owners (see Segnini, 2000) and further a 1994 Supreme Court ruling upheld the need for compensation (Busch et al., 2000), it appears that these have been treated as reserves. Sanchez-Azofeifa et al. (2003) shows that little forest in reserves has been cleared.

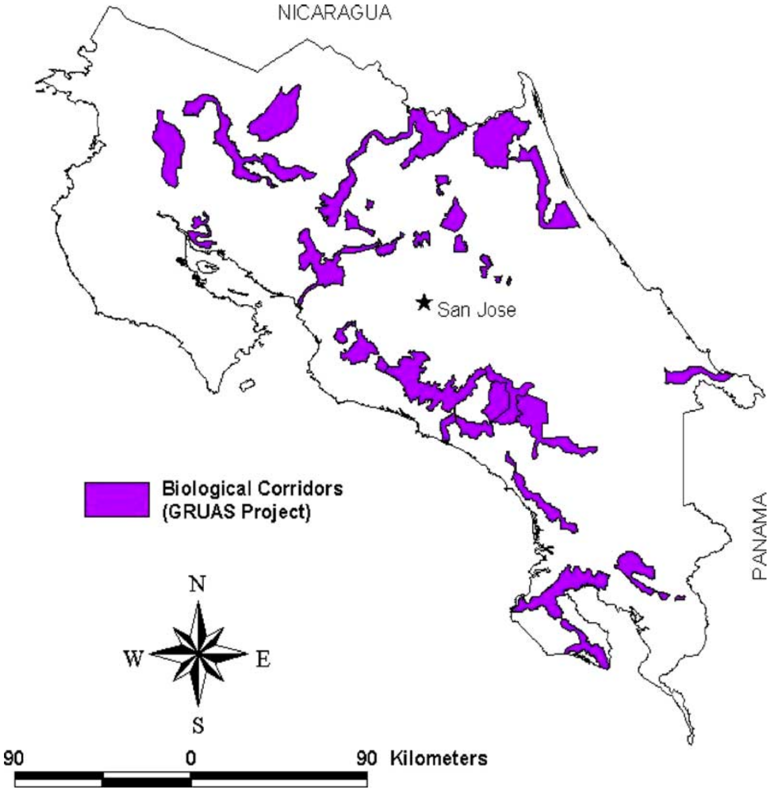

Fig. 1. Locations of the proposed GRUAS reserve sites.

protected areas were also considered, including for feasibility of implementation. It has been suggested that the network of protected areas does not currently but should represent all of the different ecological conditions present in Costa Rica. Along these lines, the expert consultant group tried to identify the minimum area necessary to conserve each of the 'vegetation macro-types' within the country (Garcia, 1997). Thus, the GRUAS network of proposed conservation corridors is now expected to cover 9% of Costa Rica and to protect new ecosystems to promote protection of biodiversity.

3. Estimating deforestation pressure

This section summarizes work in Kerr et al. (2003), an economic analysis of forest transitions over time in Costa Rica. Like many others (e.g. Stavins and Jaffe, 1990), we employ a dynamic theoretical model, but we include empirical implications of such a model that are not in previous work. We feel this is important for projecting land use along a development path.

3.1. Model and derived regression equation

For further details we refer the reader to the paper cited, and try to provide just enough information for the analyses to be understandable. In the model below, a risk-neutral landowner selects when to clear land now forested. Decisions are taken in order to maximize the expected present discounted value of returns. Relevant returns are from the potential forest and non-forest land uses, and are affected by the costs of clearing (including option values) and the interest rate:

$$\max_T \int_0^T (S_{it})\, \mathrm{e}^{-rt}\, \mathrm{d}t + \int_T^\infty (R_{it})\, \mathrm{e}^{-rt}\, \mathrm{d}t - C_T\, \mathrm{e}^{-rt} \tag{1}$$

where S_{it} is the expected return to forest uses of the land on plot i in year t, R_{it} the expected return to non-forest land uses on plot i in year t, C_T the cost of clearing net of obtainable timber value and including lost option value, and r the interest rate.

Clearing must be profitable and, so waiting is not preferred,[10] an arbitrage condition must hold:

$$R_{it} - S_{it} - r_t C_t + \frac{\mathrm{d}C_T}{\mathrm{d}t} > 0 \tag{2}$$

Our empirical approach is based on this condition,[11] assuming a second-order condition.[12]

From Eq. (2), for any given plot, and assuming that our explanatory variables enter in a linear way, the implications of this model can be expressed in terms of a latent variable y^*_{it}, a binary y_{it} dependent variable indicating whether the plot is cleared, and y^*_{it} linked with y_{it} as follows:

$$y^*_{it} = R_{it} - S_{it} - r_t C_t + \frac{\mathrm{d}C_{it}}{\mathrm{d}t} = \beta' X_{it} + \varepsilon_{it},$$

$$y_{it} = \begin{cases} 1 & \text{if } y^*_{it} > 0 \\ 0 & \text{otherwise} \end{cases} \quad \text{s.t. probability}(y_{it} = 1 = \text{cleared}) = \text{probability}(\varepsilon_{it} > -\beta' X_{it}) \tag{3}$$

Plots with higher values of unobservable factors ε_{it} are more likely to be cleared, conditional on the observed factors X_{it}. If the distribution of ε_{it} is logistic, then we have a logit model:

[10] Clearing costs may fall and future returns may depend upon the time of clearing, e.g. land may degrade with use.

[11] We are not estimating structural parameters, though. Our coefficients could represent the roles of variables in this arbitrage condition alone, but are reduced form for two reasons. First, our explanatory factors may affect more than one of these variables. Ecological constraints, e.g. may affect both returns and conversion costs. Second, our coefficients may blend factors' effects in this condition with effects on profitability of clearing (see next footnote).

[12] This condition is satisfied if the relative returns to non-forest uses are increasing over time while the rate of fall in costs is decreasing. Population and economic growth plus improved infrastructure may make this the case. Then the arbitrage condition is necessary and sufficient for clearing (implying that clearing is profitable, i.e. discounted rents from non-forest uses more than compensate for lost returns from forestry uses and net cost of land clearing). But at a certain stage of development, this may be reversed. Environmental protection can become more stringent, returns to ecotourism can rise over time, and agriculture can become more capital intensive and require less land. If in that case the second-order condition does not hold, then both the arbitrage condition and the profit condition must both hold. Our coefficients may be an amalgam of explanatory factors' effects within the arbitrage and the profit conditions.

$$F(\beta' X_{it}) = \frac{1}{1 + \exp(\beta' X_{it})} \tag{4}$$

Given that we have grouped data rather than plot-level data, we estimate this model using the minimum logit chi-square method also known as 'grouped logit'.[13] The equation we estimate is

$$\log\left(\frac{\hat{h}_{it}}{1 - \hat{h}_{it}}\right) = \beta' X_{it} + \mu_{it} \tag{5}$$

and the equation is estimated by weighted least-squares. $\hat{h}_{it}$ is the deforestation rate for a given sub-district (land unit), a simple estimate of all of the plot-level hazard rates in that unit.

3.2. Data and variables

We use forest observations at five points in time (1963, 1979, 1986, 1997, and 2000). In a given year, the area of forest that is in a district (436 in Costa Rica) and a given "life zone" (12 in Costa Rica using Holdridge's delineation of these combinations of precipitation and temperature) constitutes one forest observation. On average, there are about three such life zones in a district. This forest-cover data is from several sources, and is described in more detail in Pfaff et al. (2000). Table 2 gives variables' descriptive statistics, weighted by the forested area for each observation.

From this data, by district life zone we calculate changes over time, the deforestation rate or hazard rate.[14] The intervals between our forest observations vary in length, and thus to make the estimated full-interval hazard rates comparable we convert them to annualized hazard rates.[15]

Explanatory variables include GDP per capita, or time, with positive expected effects as they stand in for unobserved aspects of development. We include squared terms for these proxies with if anything negative expected effects in accord with rationales for 'environmental Kuznets curves'. This empirical approach also permits direct testing of time trends in the hazard rates.

We also include the percentage of forest previously cleared. To the extent that clearing suggests that the best land has been taken, it should have a negative effect. But a positive effect would be expected if clearing spurs investments within a process of endogenous development.[16]

Standard relative returns' variables and proxies include an effort to directly measure the monetary returns to non-forest land use for each of the four major export crops: coffee,

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[13] Berkson (1953) cited in Maddala (1983). See also Greene (1990) for explicit discussion of heteroskedasticity.

[14] We assume no clearing before 1900. In fact some areas had been cleared by that time, and long before. However, population was only around 200,000. We compare these results to use of 1850 as the date and find little difference.

[15] By looking only at deforestation of land forested at the start of the period, we acknowledge that irreversibility in clearing may introduce a significant asymmetry between factors' effects on clearing and effects on reforestation.

[16] We use the lagged hazard as well in later cross-sections in which it differs from the total previous clearing. As an indicator of partial adjustment, or of persistence from endogenous unobservable development, it has a positive prior.

Table 2
Variable definitions and descriptive statistics (weighted by forest area)[a]

Variable	Name	Mean	S.D.	Minimum	Maximum
Deforestation rate	Dep.Var.	0.016	0.048	0.00001	1
Returns					
Returns/ha (US$, 1997)	Return	659	1154	0	5047
Distance to major markets (km)	Distance	73	38	0	186
Roads density	Roadsden	0.0026	0.0022	0	0.049
Population density (no./ha)	Popden	0.089	0.38	0	107
Ecology					
Dummy for humid life zones	Good lz	0.24	0.43	0	1
Dummy for very humid (pre-montane, lower montane) and montane life zones	Medium lz	0.23	0.42	0	1
Dummy for very humid (tropical), dry (tropical), and rainy life zones	Bad lz	0.54	0.50	0	1
Proportion of known soil types that are entisol	Badsoil	0.11	0.23	0	1
Dynamics					
Time (midpoint of period)	Time	66	26	33	100
Proportion of forest cleared	%Cleared	0.21	0.26	0	0.99996
Lagged hazard (1979 onward)	Prevhazard	0.0080	0.017	0	0.46

[a] This is Table 1 from Kerr et al. (2003).

bananas, sugar, and pasture. These were calculated by estimating the yield, multiplying by the export price, and subtracting an estimated production cost. These measures were also all combined for a single summary measure, although the four may not have been measured in perfectly consistent ways.

The returns' proxies include ecological and economic variables.[17] These include the combinations of humidity, bio-temperature, and elevation represented by Holdridge Life Zones. We have grouped our life zone variables into productivity classes: good, medium, and bad. We also have data on soil type. Better soil should lead to higher hazards. We proxy for access with the minimum linear distance from the center of the district to the closest of three key cities and ports: San Jose, Puntarenas, and Limon. We also include this distance interacted with time, as development may include progress in the transport sector that diminishes the effect of distance. In the most recent years (1986, 1997), we use density of roads to proxy for access to markets.

3.3. *"Deforestation pressure" function*

Table 3 presents our core specification, a grouped logit explaining the hazard rates.[18] In terms of the dynamics intended to aid in projection of clearing rates along the development path, these results suggest that hazard rates initially rise with development over time and

[17] Pfaff et al. (2000) discusses additional ecological data and analyses necessary to project carbon sequestration.
[18] This treatment of an aggregation of individual plot-clearing choices follows, for instance, Greene (1990, p. 670).

Table 3
Pooled regression results[a]

Explanatory variables	Coefficients (*t* statistics)	Defaults	Marginal Δs	1986 Marginal effects (1986 default hazard = 0.048)
Return	4.7E−06 (0.26)	1232	1685	0.00036 (<1% of default)
Distance	−0.020 (−16)	71 km	38 km	0.011 (23%)
Distance × time	3.0E−04 (16)	(implied by the above)		(joint with above)
Good lz	0.21 (6.0)	0	1	0.011 (23%)
Bad lz	−0.46 (−11)	0	1	−0.017 (35%)
Badsoil	−0.13 (−1.9)	0	1	−0.0057 (12%)
Time	0.13 (21)	86	10	−0.026 (54%)
Time^2	−0.0012 (−25)	(implied by the above)		(joint with above)
%Cleared	1.8 (7.4)	37%	28%	0.019 (40%)
%Cleared^2	−0.71 (−2.6)	(implied by the above)		(joint with above)
Constant	−6.1 (−36)			
R^2	0.36			
N	4343			

Dependent variable: annualized deforestation probability (see Section 4.1). Years: 1900–2000, pooled transitions. Grouped logit with spatial error correction over 5 km (see Section 3.3). Weighted OLS addresses heteroskedasticity.

[a] This is Table 2 from Kerr et al. (2003).

then may fall. While this fits our priors, it must be noted that purely temporal stories rely on few observations.

Previous local clearing of forest has a positive effect. This may indicate fixed unobserved characteristics that support higher productivity and clearing rates. It may also reveal endogenous dynamics in which unobserved investments follow clearing and encourage future deforestation.[19]

These results suggest that in choosing among sites for new reserves, all else equal it is more important to save the sites near existing clearing. Those sites are more likely to face high clearing pressure in the future, and thus the net benefits from protecting them will be higher.

Higher productivity life zones tend to have higher deforestation, and lower productivity life zones have lower pressure, as expected.[20] Thus, higher productivity ecological conditions indicate priority sites for new reserves, all else equal. Also, low-quality soil has lower clearing, although this is not a very robust result. Market access matters, as greater distances to market lower clearing. Adding a dynamic twist, for a given distance this effect diminishes with time. At any point in time, though, candidate sites closer to markets should be higher reserve priorities.

Note that we considered the potential for spatial autocorrelation in the unobserved factors to affect estimation results. We present corrected results, as the estimated lambda is

[19] The lagged hazard is not found in Table 3, only in later-year cross-sections when it is distinct from total clearing (see Table 4 of Kerr et al. (2003)). In those analyses, it has a positive estimated coefficient, in keeping with the priors.

[20] Our direct measure of monetary crop returns raises clearing in most cross-sections and is not the main focus here. It is worth noting that when other proxies for land-use returns are not included, it has a significant effect.

significant. Both the coefficients and their significance are little affected by correcting over other distances.

In sum, we confirm the expected importance of market access, or distance, in prioritizing reserve sites. However, policymakers appear to often choose the farthest possible distance, while our perspective suggests the opposite if maximizing the net benefits of effective species reserves. Further, we highlight the importance of spatial persistence in clearing rates when considering the clearing pressure across the candidate sites, and we demonstrate a role for ecological conditions.

4. Evaluating candidate reserve sites

4.1. Interpreting the 'pressure' estimates

The coefficients in Table 3 can be applied to observed characteristics of candidate sites, yielding projected pressure values. These coefficients are estimated using observed deforestation within areas of Costa Rica that lacked reserves. Thus, their most natural interpretations are as the marginal contributions of the observed characteristics for a candidate site to the probability that species habitat will be cleared in the absence of a reserve, i.e. as marginal contributions to $P_{it\mathrm{N}}$.

If so, then projected pressure values should be interpreted precisely as estimates of $P_{it\mathrm{N}}$.

Recall, the net benefits from a new reserve at site *i* are of the type $\sum_{j,t} (P_{it\mathrm{N}} - P_{it\mathrm{R}}) B_{ij} - C_{it}$. Thus, sites with high estimated pressure should be high priorities for new reserves, all else being equal.

4.1.1. Correlation with C_{it}

However, all else may not be equal across sites whose projected levels of pressure differ. Specifically, the observed characteristics (X_{it}) in Table 3 may be correlated with the costs (C_{it}) of creating and maintaining a reserve. If the dominant threat to habitat results from agricultural use, then higher projected pressure suggests that the observed characteristics also imply high returns and profit in agriculture, and thus also a high cost to an agency of outbidding farmers for the site. Should that be the case, the high-pressure sites would not necessarily yield higher net benefits. Positive correlation of threat and cost would mean that reserves' costs could offset their benefits.

There are settings in which a positive correlation between threat and cost would not arise. The highest possible land returns could result from activities that do not require clearing, such as gathering of non-timber forest products, like fruits, or hunting of animals that live in forest areas. Should this be the case, then a low clearing threat and high cost of land purchase could coincide. Alternatively, the pressure on a forest may result from people squatting on and deriving a living from forest land to which they have no or weak property rights. Then a high clearing threat and a low cost of land purchase could coincide. However, while both of these settings are observed, generally the threat of habitat clearing and cost of a reserve may well be positively correlated.

Nonetheless, in practice our projected pressure index remains useful as an indicator of threat to species habitat. If seriously considering the purchase of candidate sites, in the process of negotiations an agency will undoubtedly obtain cost information more precise than estimates from our regression, leaving the index to play the role of threat. Further, the index of threat may still be useful concerning cost, as the search for cost information is itself costly. Thus, using high projected pressure to prioritize candidate sites could help to target that search and thus save cost. Some high-pressure areas will be of high cost and thus unattractive, but others will be good choices.

4.1.2. Correlation with P_{itR}

The observed characteristics (X_{it}) may also be correlated with the threat of forest clearing in the presence of a reserve, $P_{it\mathrm{R}}$. If the pressure $P_{it\mathrm{N}}$ results from any reward to habitat clearing, then high projected pressure may also indicate a high chance a reserve is invaded (e.g. high $P_{it\mathrm{R}}$), since the net benefit of invasion will equal the reward from habitat clearing minus certain costs.[21] Note that with an estimate in hand of $P_{it\mathrm{N}}$, i.e. the level of threat that bounds a reserve's benefits, an agency needs to know not $P_{it\mathrm{R}}$ itself but rather $(P_{it\mathrm{N}} - P_{it\mathrm{R}})$, i.e. the effectiveness of the reserve. Were the $P_{it\mathrm{N}}$ and $P_{it\mathrm{R}}$ always close to equal, that would mean that reserves are not very effective.

Many reserves are in remote, low-pressure (low $P_{it\mathrm{R}}$) sites. These may result from a belief that reserves can be effective there. Unfortunately, it seems likely that these sites were chosen for low $P_{it\mathrm{R}}$ and that they feature low $P_{it\mathrm{N}}$ as well, i.e. the reserve effectiveness may well be minimal. Again, more generally, threats of habitat clearing with and without a reserve may be correlated.

However, in many settings, reserves will be somewhat effective, i.e. feature $P_{it\mathrm{R}} < P_{it\mathrm{N}}$. Factors such as sociocultural norms, monitoring expertise, and the historical strength of property rights all have effects. Further, they differ across sites such that effectiveness differentiates sites.

But if so, is effectiveness correlated with $P_{it\mathrm{N}}$? Sites with high pressure should not be the priorities if they also happen to be where reserves would be least effective. To the extent that $P_{it\mathrm{N}}$ and $P_{it\mathrm{R}}$ differ due to these factors, though, the gap between them need not correlate with the $P_{it\mathrm{N}}$. Thus, where norms, incentives, monitoring expertise and property rights generally make $P_{it\mathrm{R}}$ low, agencies can focus on B_{ij}, $P_{it\mathrm{N}}$, and C_{it} for all sites. But where their estimate of the status of these factors suggests that $P_{it\mathrm{R}}$ will in some locations be significant, agencies can search high-$P_{it\mathrm{N}}$ sites for when the local information on these factors (distinct from $P_{it\mathrm{N}}$) suggests reserve effectiveness.

4.2. Priorities within proposed locations

In light of the conceptual discussion above, we will interpret our projections based on the sites' observed characteristics (X_{it}) plus the estimated coefficients (β) in Table 3 as projections of future $P_{it\mathrm{N}}$. Speaking more concretely, we project deforestation rates using an iterative process. To project forward our dependent variable, we must first estimate the future values of all of the explanatory variables (i.e. $X_{i,t+1}$) for every site in Costa Rica for

[21] This presumes imperfect enforcement, which seems realistic. If we assume sufficient effort to prevent invasion, then higher projected pressure may instead correlate with another reserve cost, the cost of this enforcement effort.

every future period of interest. Then using these underlying predictions for each site we can evaluate the following equation:

$$\ln\left(\frac{h_{t+1}}{1-h_{t+1}}\right) = \beta_2\,\mathrm{dist}_{t+1} + \beta_3(t+1)\mathrm{dist}_{t+1} + \beta_4\,\mathrm{good\,lz}_{t+1} + \beta_5\,\mathrm{bad\,lz}_{t+1} + \beta_6\,\mathrm{badsoil}_{t+1} + \beta_9\,\%\mathrm{cleared}_{t+1} + \beta_{10}\,\%\mathrm{cleared}^2_{t+1} + \mathrm{constant} + \mathrm{effect\ of\ development}_{t+1} \quad (6)$$

The distance and three natural productivity variables are assumed to remain constant over time in the projections used below. Previous clearing of course changes with time, and it can be calculated iteratively, starting with the hazard rate for the previous period (e.g. h_t), for instance:

$$\mathrm{Total\ forest\ stock}_{t+1} = (1 - h_t) \times (\mathrm{total\ forest\ stock})_t$$

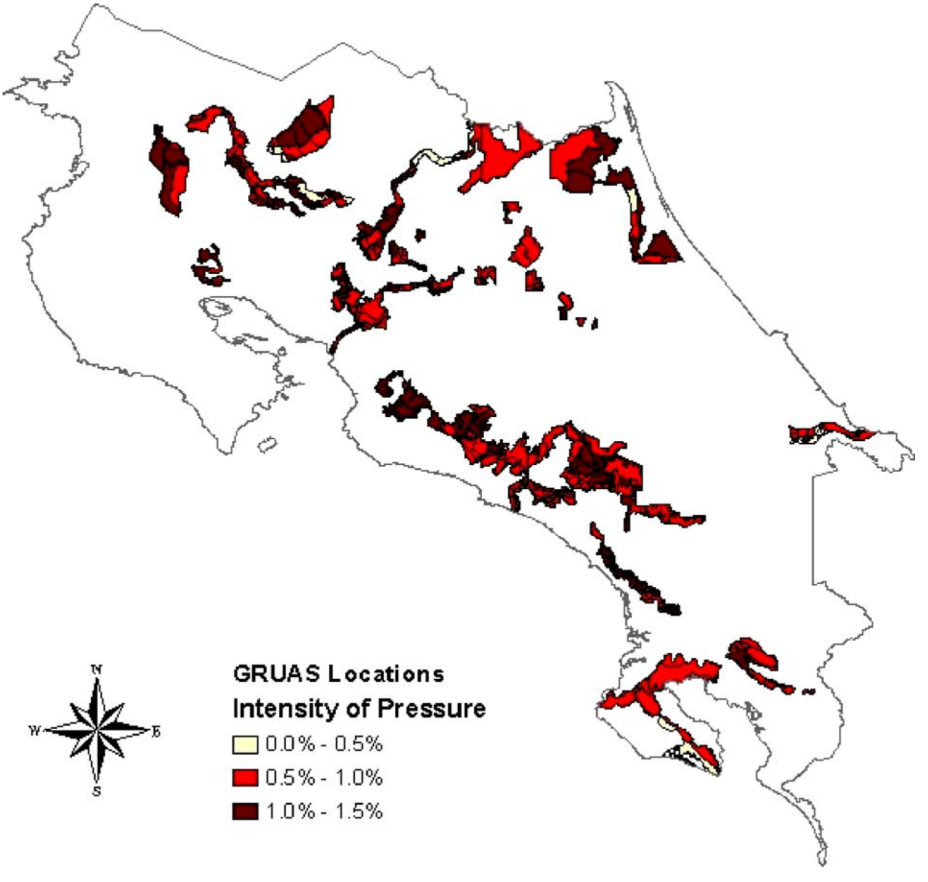

Fig. 2. Projected pressure across the proposed GRUAS sites.

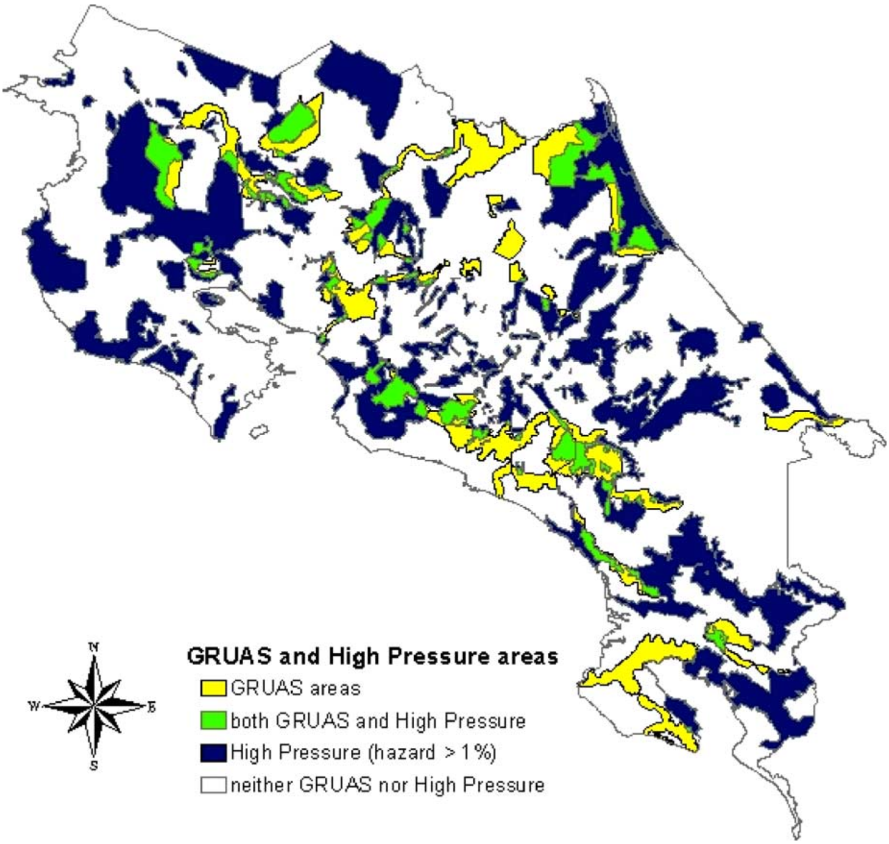

Fig. 3. Comparing GRUAS to sites of highest projected pressure.

and then

$$\%\text{cleared}_{t+1} = \left(1 - \frac{(\text{total forest stock})_{t+1}}{\text{total potential forest area}}\right) \times 100 \tag{7}$$

The development process will also continue over time. From Table 3, we could use the quadratic form of time to predict forward, but this would quickly yield irrelevant negative values. Thus instead we have estimated the regression in Table 3 again (other effects are robust to this), using dummy variables for the different time periods to control for the effects of development. Next, rather than simply using the coefficient for the final time dummy for the period 1997–2000, we fit a curve to these time dummies to estimate a path of effects of development over time.[22]

Using projections for 2002, Fig. 2 shows projected pressure for the GRUAS locations. The colors alone convey the main point that pressure is not uniform across these candidate sites; the projected values for future rates of deforestation can easily differ by a factor of 3 or more. This suggests which GRUAS sites should be higher priority, based on the $P_{it\text{N}}$

[22] For more discussion of projections, see Pfaff (2003).

input to decisions, although of course current species presence (B_{ij}), reserve costs (C_{it}), and effectiveness matter too.

4.3. Proposed versus other locations

However, the new reserves have not yet been created, so we should also ask whether the proposed locations are the best ones for Costa Rica. To start, we should ask whether they are the highest-pressure or relatively high-pressure sites. The area-weighted average deforestation rate in the proposed sites is 0.93% (i.e. $<$1%), compared with an average for all the sites of 0.90%. That the difference is not drastic is consistent with the ecological rationale for the choice of sites. Since a main goal was to cover many ecosystems, if anything we might expect average pressure. Thus, overall, the GRUAS locations are not those we would prioritize for pressure reasons alone.

Which areas would be ‘pressure priorities’? Fig. 3 shows the set of locations for which projected future pressure would yield deforestation rates of greater than 1% per year. That is not an extremely high rate of clearing, reflecting the whole nation’s situation at this point along its chosen development path, although even 1% per year will compound over time, in particular relative to the close-to-zero projected pressure levels for other locations in Costa Rica, including some of the proposed sites for biodiversity corridors. The point here is that areas with higher $P_{it\mathrm{N}}$ differ significantly from, but overlap, the proposed GRUAS sites.

Fig. 4 suggests a quite different, ecological rationale for the choice of sites concerning EB_{ijt}. Above, other than in one footnote, future presence (EB_{ijt}) followed directly from

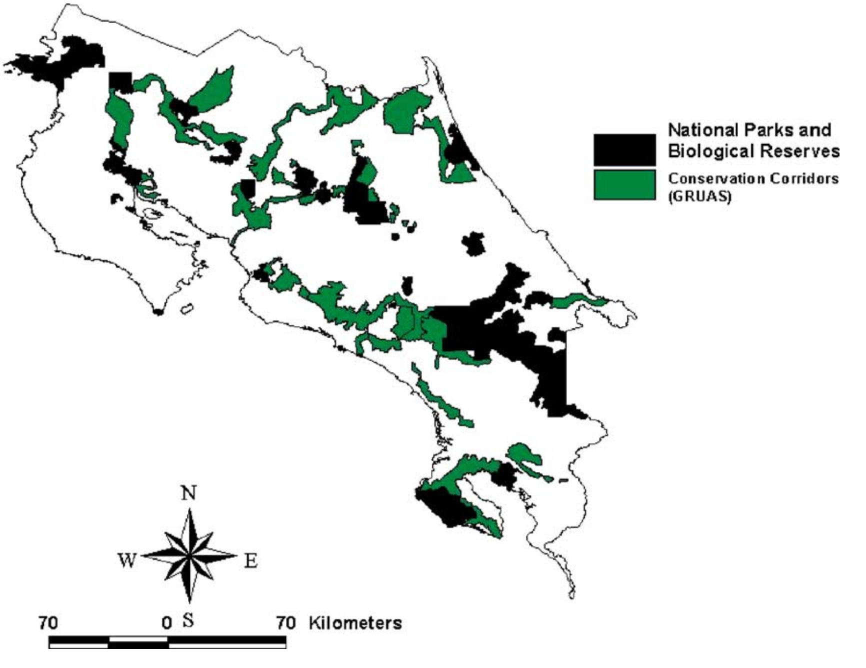

Fig. 4. Existing national parks and biological reserves and the GRUAS corridors.

current presence (B_{ij}) plus the threat to habitat at that site (P_{it}), which could be lowered by creating a reserve at the site. However, because species require ranges of a certain size, and also nearby fragments may help to re-colonize areas in which a species is threatened or gone, expected future species presence may increase with nearby forest sites. Both the term 'corridor' and Fig. 4 suggest that policymakers are accepting the logic that, when linked, forested areas can provide better habitat services. Such ecological rationales may well remain dominant in initial identification of forest sites of interest. Still, as suggested by Figs. 2 and 3, the level of projected pressure ($P_{it\mathrm{N}}$) suggested and provided in this paper could affect the prioritization of locations, once a set of forested sites featuring both current species presence and proximity or linkages to other forested sites is found.

5. Conclusion

This paper suggested a index of deforestation pressure as a tool for reserve planning, alongside data on species and costs. As it represents the threats to habitat, the index correlates negatively with species survival. We discussed how such an index is estimated and its proper interpretation, then applied a regression for deforestation rates in Costa Rica to project future levels of forest pressure. This suggested priorities among proposed sites for biodiversity corridors and other forested sites.

A focus on pressure raises the issue of a reserve's future benefits. In most existing work, future species presence can be guaranteed by current presence plus plans for new future reserves. That assumption may be incorrect if habitat is threatened and annual reserve spending is limited. Our work notes that land use can affect future presence, while Costa Rican agencies' clear goal of putting new sites near to existing reserves assumes that spatial linkages affect future presence.

For this analysis, we took existing plans for reserves as given and pointed out that adding our projections of habitat pressure can affect site choice. More generally, and in our future work, one should integrate species presence, cost data and projected pressure in establishing priorities.

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