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THE ECONOMICS AND FINANCE OF NATURE LOSS

Emission Prices, Biomass, and Biodiversity in Tropical Forests†

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The prudent reforestation of tropical rain forests depends on alternative land uses, the dynamics of carbon accumulation of forest trees, and the implicit social cost of emissions. In this paper, we discuss and extend recent research by Assunção et al. (2023) showing that arguably modest foreign transfers per unit of carbon captured could incentivize substantial reforestation in areas currently used for low-productivity cattle ranching in Brazil. Building on this research, we initiate the study of the relationship between externally set emission prices and changes in biodiversity and biomass. The Amazon holds 10 percent of the world's vertebrate and plant species. There are more than 15,000 tree species, the vast majority of which are rare. Drawing on the scientific literature on rain forest biodiversity, we provide some preliminary estimates of how carbon pricing could affect biodiversity in the Brazilian Amazon.

I. Carbon Prices and Reforestation

Our recent research, Assunção et al. (2023), uses a rich dataset from the Brazilian Amazon to explore the robustly optimal deforestation/reforestation over space and time. The Assunção et al. (2023) computations rely on a formal model built to support the quantitative analysis and confront uncertainties of multiple types. Since we do not have direct measurements of alternative productivities for land usage, we rely on statistical models for extrapolation and interpolation supported by robust Bayesian inference. We go beyond typical forms of "uncertainty quantification" by using policy goals to assess where uncertainty matters. We demonstrate how uncertainty shapes our conclusions and challenge the simplistic notion that concerns about uncertainty result in climate policy inaction.

Empirical measurements show that subregions of the biome differ substantially in agricultural productivity and capacity to absorb carbon. Thus, we consider a partition of the Amazon into $I := 1043$ sites. While we use a geographic partitioning in our model to capture the heterogeneity in the productivities, we adopt a more global perspective on the overall land allocation because of well-established results on edge effects and the lower carbon capture productivity of forest fragments of smaller than 100 square kilometers (km).

This research recognizes that policy analysis of this nature requires a dynamic model. Keynes' dictum, "In the long run, we will all be dead," is particularly applicable to climate change. It is crucial to consider the dynamics of carbon accumulation, which are missing in static treatments or in "dynamic treatments" that omit the carbon accumulation dynamics of trees.

To make this investigation operational, we take as inputs alternative social costs of emissions and deduce the resulting implications for the allocation of land. We first use the model to set an initial cost and provide a narrative for the current state of the forest. Historically, most deforestation in the Amazon has been either the result of government incentives or illegal but tolerated activities. Past experience shows that a government is able to substantially control deforestation at low cost (Assunção, Gandour, and Rocha 2023). Thus, the current state of the forest is more likely to reflect the

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valuation of the forest, including its alternative uses, by the federal and local Brazilian governments than by decentralized occupiers of land in the Amazon. Additional transfers are needed to preserve the Amazon rain forest because Brazil alone is not sufficiently incentivized to engage in reforestation. We calculate the carbon capture potential of the biome by increasing the social cost of emissions assumed by a hypothetical planner with a more global perspective.

A. *Model*

For each site i, we consider two state variables: Z_t^i, the amount of land dedicated to agriculture, and X_t^i, the amount of CO_2 captured in the site. The planner controls $\dot{Z}_t^i$, and

$$\dot{X}_t^i = -\gamma^i\left(\dot{Z}_t^i \vee 0\right) - \alpha\left[X_t^i - \gamma^i\left(\bar{z}^i - Z_t^i\right)\right], \tag{1}$$

where γ^i denotes the maximum capacity to store carbon per unit of forest and α is the speed of convergence. Agricultural output in site i is given by $A^i = \theta^i Z^i$. The planner takes as given P^e, the price of emissions supported by transfers and the evolution of agricultural prices P_t^a.

The planner faces ex ante ambiguity in the parameters γ^i and θ^i that govern the land use productivity for each site i. Let φ denote the full vector of unknown parameters $\left(\gamma^i, \theta^i\right) : i = 1, 2, \ldots, I$. The planner takes as input the ex ante (to the decision problem) posterior distribution constructed from data with a conveniently chosen likelihood and prior distribution. This construction requires some cross-sectional extrapolation since we have limited direct evidence for some of the sites. The estimation and extrapolation induce dependencies in the posterior distribution for φ. The planner confronts the parameter ambiguity by performing a sensitivity analysis: minimizing the planner's objective by altering the posterior distribution of φ subject to a relative entropy or Kullback-Leibler penalty scaled by a parameter ξ. Larger values of ξ imply a larger penalty and, therefore, less aversion to ambiguity. Setting $\xi = \infty$ gives the planner full confidence in the baseline posterior distribution.

To pose the robustly optimal planning problem for the Brazilian Amazon, we start by computing the intertemporal objective conditioned on the parameter vector φ, taking into account any pure risk considerations:

$$f(d, \varphi) = E\int_0^\infty e^{-\delta t}\left[P_t^a \sum_{i=1}^I A_t^i - P^e\left(\sum_{i=1}^I \kappa Z_t^i - \dot{X}_t^i\right) - LC_t\right]dt, \tag{2}$$

subject to (1) and $0 \leq Z^i \leq \bar{z}^i$, the total area of site i, where

$$LC_t = \frac{\zeta_1}{2}\left(\sum_{i=1}^I \dot{Z}_t^i \vee 0\right)^2 + \frac{\zeta_2}{2}\left(\sum_{i=1}^I \dot{Z}_t^i \wedge 0\right)^2$$

quantifies the asymmetric land-change costs, and κ measures CO_2 impact of agriculture. The expectation presumes a Markov process for the price of agricultural output. Here, d denotes the entire sequence of hypothetical decisions contingent on the relevant agricultural prices.

The planner criterion favors fully reforesting a plot before starting on another plot. Our plots are approximately 4,500 km^2 at the equator, guaranteeing that reforestation would not produce forest fragments. The objective takes the price P^e as input and implies that the planner is paid for net CO_2 capture—simple preservation is not rewarded. (See Assunção et al. 2023 for calibration and solution methods.)

We adopt an ex ante representation of the decision problem. Let π denote the baseline distribution over the parameter vector φ, constructed with a computationally tractable Bayesian method. The ambiguity-averse planner ranks alternative decision processes by solving the minimization problem:

$$\min_g \int \left[f(d, \varphi) + \xi \log g(\varphi)\right] g(\varphi) d\pi(\varphi), \tag{3}$$

Table 1—Present-Value Comparisons

	Agricultural output value ($\$10^9$)			Planner value ($\$10^9$)		
b (\$)	Ambiguity neutral	Ambiguity aversion	Percent change	Ambiguity neutral	Ambiguity aversion	Percent change
0	364	279	−23%	244	182	−25%
25	15	17	13%	526	453	−14%

subject to $\int g(\varphi)d\pi(\varphi) = 1$ where f is given in (2). In this formulation, $g(\varphi)d\pi(\varphi)$ represents an altered distribution over the parameter vector φ, and $\xi\int[\log g(\varphi)]g(\varphi)d\pi(\varphi)$ penalizes departures from the baseline posterior distribution $d\pi(\varphi)$. By maximizing (3), the planner constructs a robustly optimal allocation of land in the Brazilian Amazon over time and across space. As an illustration, the results reported here use a specific value of the penalty parameter, $\xi = 5$, to capture this aversion. Assunção et al. (2023) use insights from robust statistics to provide more meaning to this particular parameter setting and explore sensitivity to alternative configurations.

B. *Results*

We explore implications of changing P^e, the social price of emissions.[1] We first calculate a price P^{ee} that justifies observed deforestation between 1985, the date when reliable price data are available, and 2008, the year the Amazon Fund was announced to incentivize forest preservation with funding from Norway. Although the price is expressed per unit of CO_2, the price also includes other forest services. The price, P^{ee}, is model dependent since it must explain the same observed aggregate deforestation. For the case of no uncertainty aversion, $P^{ee} = \$6.6$, whereas under ambiguity aversion, $P^{ee} = \$4.5$. Our calibrations imply that the planner is more concerned about ambiguity in the productivity of cattle farming than about ambiguity in the maximum carbon content of forests under business-as-usual. This induces the planner to opt for less deforestation. Therefore, explaining the same observed deforestation under ambiguity aversion requires a lower valuation of net emissions.

Under business-as-usual computations, Assunção et al. (2023) show that biome area loss exceeds 21 percent, creating a scenario that Flores et al. (2023) say could "trigger unexpected ecosystem transitions and potentially exacerbate regional climate change." In addition, deforestation and degradation lower water recycling and cause downwind moisture loss, creating a cascading effect that doubles the impact of the initial damage.[2] In contrast, adding to P^{ee} transfers $\$b$s alters substantially the robustly optimal land allocation. For instance, adding transfers of $\$b = 25$ to P^{ee} reduces substantially the land allocated to agriculture through reforestation. For such transfers, the uncertainty concerns shift from the θ_i's to the γ_i's.

Table 1 summarizes statistics on present-value over the next 200 years. Absent ambiguity aversion, when $b = 25$ the value of agricultural output almost vanishes. This loss is more than compensated for by \$422 billion in transfers along with the net gain in forest economic services such as natural rubber, nuts, fruits, cosmetics, and tourism. The economic value of the biome more than doubles. Under ambiguity aversion, the present value for $b = 0$ drops by 25 percent, while for $b = 25$ it is reduced by 14 percent. Thus, under ambiguity aversion, the advantage of accepting a contract in which net emissions are paid \$25 relative to the status quo increases. The difference declines as ambiguity aversion diminishes, but it stays positive.

Land conversions under $b = 0$ or $b = 25$ are front-loaded. As a consequence, emissions under $b = 0$ are 16 gigatons (Gt) in the first 30 years, and carbon capture under $b = 25$ is about 18 Gt if we abstract from ambiguity aversion. Hence, the total change in emissions in the next 30 years is 34 Gt. This

[1] For simplicity, in this paper, we set $P_t^a = 41$, the mean of the stationary distribution estimated in Assunção et al. (2023).
[2] Araujo et al. (2023).

Table 2—Percent Biodiversity Change per HA

	Mean	Min	10%	20%	50%	80%	90%	Max
$b = 0$	−13.7	−100.0	−100.0	0.0	0.1	2.5	11.8	290.1
$b = 25$	31.1	0.0	0.0	0.1	1.8	46.5	105.0	515.3

is a significant amount during a time when substantial technological change in carbon capture is not likely to occur. Since Brazil would be paid for the net carbon capture, the effective cost per ton would be $\$(25 \times 18)/34 \approx \$13$ in the next 30 years. When the planner is ambiguity averse, the difference in CO_2 net capture across trajectories increases slightly, modestly reducing the effective cost.

The computations we report are ex ante in nature, but this raises a question about incentives to depart from forward-looking commitments. Table 1 shows that the planner would agree to sign an agreement to receive (pay) \$25 per ton of CO_2 captured (emitted) in the Brazilian Amazon. However, since mature forests reach an equilibrium, the value of future payment flows declines, and the planner may find it optimal to defect and revert to an optimal trajectory for $b = 0$. Calculations in Scheinkman (2024) show that for $b = 25$, defection can be avoided for 50 years at a cost of a *carrot*—a bond payable in 50 years if there are no prior deviations from planned land use changes—that increases the cost to buyers by less than 55 cents per ton. Alternatively, one can choose a *stick* that punishes deviations in planned land use changes. Boycotting Brazilian exports, contingent on deviations, lowers the cost of these schemes.

II. Biodiversity

Incorporating biodiversity into the analyses of rain forests creates modeling and measurement challenges. For instance, there is no agreed-upon scientific model that connects biodiversity measures to economic performance. Instead, there are many different ways to measure biodiversity such as species count, Hill indices (which assign alternative weights to rare species versus abundant ones), functional diversity, and genetic diversity. In this essay, we narrow our focus by concentrating on three aspects of biodiversity: (i) the impact of policy-induced emission prices on species counts; (ii) the multiplier effect on carbon losses induced by losses of biodiversity; and (iii) the spatial priorities for biodiversity protection in tropical forests and the ramifications of future climate change.

The computations that we report are intended to be suggestive, building on our earlier work. They open the door to future research that captures modeling interactions between biodiversity and deforestation while incorporating uncertainties and their ramifications for policy.

A. *Emission Price Impacts on Biodiversity*

Although we do not account for the impact of biodiversity on the planner's problem, we use estimates of the η^i's to measure the potential tree biodiversity per hectare (ha) of each of our 1,043 sites to examine how biodiversity is altered by changing the land allocation after 30 years. The construction of η^i's is detailed in the Supplemental Appendix. To simplify our computations, we suppose that if a ha is deforested, it has zero tree biodiversity. For reforested areas, we use the estimate of Rozendaal et al. (2019). They find that for natural reforestation of tropical forests, the median time to recover 90 percent of the undisturbed tree species richness is 32 years.

The next table displays statistics on the impact on biodiversity/ha of the business-as-usual trajectory and the optimal trajectory when transfers are \$25/ton. These calculations abstract from ambiguity aversion.

Notice that when $b = 0$, the average site loses 14 percent of biodiversity/ha in 30 years. The biodiversity loss is left skewed with a median percent change close to zero. This outcome is because deforestation in the first 30 years in most sites would be either total or zero. In contrast, when $b = 25$, the average site gains 31 percent of biodiversity/ha in 30 years. The biodiversity gain is now right

Table 3—Percent Biodiversity Change

	Mean	Min	10%	20%	50%	80%	90%	Max
$b = 0$	−17.1	−100.0	−100.0	0.0	0.0	0.7	3.4	45.1
$b = 25$	6.3	0.0	0.0	0.0	0.5	11.6	22.7	62.7

Table 4—Percent Change in Site Biomass Mediated by Biodiversity (1985–2017)

	Mean	Min	10%	20%	50%	80%	90%	Max
%Δ in diversity	−3.9	−31.0	−14.0	−8.6	−0.2	0.0	0.0	10.1
%Δ in mass	−12.6	−77.3	−45.4	−30.1	−0.9	0.0	0.1	47.0
Extra %Δ in mass	−0.5	−2.3	−1.7	−1.3	−0.1	0.0	0.0	2.3

skewed with median change that is approximately 2 percent. This happens because the optimal policy favors reforesting a full site before starting on another site.

Biodiversity per ha cannot be scaled up to total biodiversity of a plot because species overlap. Instead, to calculate the distribution of changes over sites, we use the relationship first postulated by Arrhenius (1921) between the species count, S, and the area, A, $S = \mathsf{c}A^{\mathsf{a}}$, for constants c and a. The value $\mathsf{a} = 0.25$ is a common value used for tropical forests.

Table 3 displays the distribution of species across sites using this Arrhenius formula and demonstrates qualitatively similar forms of skewness. Under business as usual, the sites lose 17 percent of biodiversity on average, but this includes total deforestation of some sites with no losses or small gains in many others. For $b = 25$, most sites experience very small, if any, gains, but the average biodiversity still increases by about 6 percent.

The estimates in Table 2 and Table 3 show that emission prices' effect on local biodiversity can be substantial. We next discuss how this change in biodiversity may produce additional loss of biomass.

B. *Biodiversity and Biomass*

O'Connor et al. (2017) estimates a relationship $P_{bm} = (P_{bd})^{\mathsf{d}}$ between the proportional loss of species, P_{bm}, and the proportional loss of biomass, P_{bd}, with a point estimate for d of 0.26 and a 95 percent confidence interval of 0.16–0.37. This relationship captures the idea that more species with different (functional) traits lead to more efficient resource use.[3] This biomass loss is in addition to any direct loss from, for example, land use change and affects the remaining biomass.

Leveraging estimates in Assunção et al. (2023), during 1985–2017, on average, plots in the Amazon lost 12.6 percent of biomass due to deforestation. Table 4 reports statistics on additional biomass losses induced by biodiversity changes using the point estimate of d and biodiversity estimates as in 2.1. We estimate that in the Brazilian Amazon, 0.8 Gt of additional emissions were induced by biodiversity changes in 1985–2017.[4]

C. *Preserving Biodiversity*

We now consider a stylized model of diversity priorities. Suppose there are two territories of equal area with different numbers of unique species, $U_1 > U_2$, and C species that are common to both territories. A fictitious social planner allocates a fraction λ of a budget B to territory one and a fraction $1 - \lambda$ to territory two in order to maximize the number of species. We assume that it costs

[3] Since this relationship is nonlinear, we implicitly assume that our plots are the correct scale to apply it.
[4] This amounts to 15 percent of current annual US emissions.

$\mathsf{b}f$ to protect a fraction f of a territory, that the Arrhenius species-area relationship holds for $\mathsf{a} < 1$ for each territory, and that the proportion of common and idiosyncratic species saved is identical to the initial proportion. If the budget is relatively scarce, *that is*, $B \leq \mathsf{b}$, the objective of the planner, scaled by B/b, is

$$\max_{0 \leq \lambda \leq 1} \left\{ \left[\lambda^{\mathsf{a}} \vee (1 - \lambda)^{\mathsf{a}} \right] C + U_1 \lambda^{\mathsf{a}} + U_2 (1 - \lambda)^{\mathsf{a}} \right\}.$$

In the Supplemental Appendix, we show that, while it is optimal to invest more in territory one than in the less biodiverse territory two, some resources would still be allocated to territory two. This positivity result generalizes to $n > 2$ territories. If, instead, we had assumed a constant cost per unit of probability ($\mathsf{a} = 1$), the planner would allocate the entire budget to territory one. Such a result is analogous to one in Weitzman (1998).

We view this model as one of long-run biodiversity. Thus, each $U_i + C$ should refer to potential biodiversity. One important caveat, however, is that future climate change will affect biodiversity. Flores et al. (2024) predict that, by 2050, human activity and global warming will cause cross-sectionally heterogeneous changes in the state of the forest. The consequences of global warming include changes in mean temperature during the dry season and changes in the frequency of extreme droughts (see Flores et al. 2024, Figure 1). Presumably, areas that have suffered critical losses of biodiversity between 1985 and 2017 should be prioritized less than other plots because their potential future biodiversity may be limited.

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