

MIT Open Access Articles

*Tropical Forests, Tipping Points,
and the Social Cost of Deforestation*

The MIT Faculty has made this article openly available. **Please share**
how this access benefits you. Your story matters.

Citation: Franklin, Sergio L., Jr. and Robert S. Pindyck. "Tropical Forests, Tipping Points, and the Social Cost of Deforestation." *Ecological Economics* 153 (November 2018): 161-171 © 2018 Elsevier B.V.

As Published: <http://dx.doi.org/10.1016/j.ecolecon.2018.06.003>

Publisher: Elsevier BV

Persistent URL: <https://hdl.handle.net/1721.1/126850>

Version: Original manuscript: author's manuscript prior to formal peer review

Terms of use: Creative Commons Attribution-NonCommercial-NoDerivs License



NBER WORKING PAPER SERIES

TROPICAL FORESTS, TIPPING POINTS, AND THE SOCIAL COST OF DEFORESTATION

Sergio L. Franklin, Jr.
Robert S. Pindyck

Working Paper 23272
<http://www.nber.org/papers/w23272>

NATIONAL BUREAU OF ECONOMIC RESEARCH
1050 Massachusetts Avenue
Cambridge, MA 02138
March 2017

The authors acknowledge the support from MIT's Center for Energy and Environmental Policy Research, MIT's International Policy Lab, and Centro de Pesquisa e Economia do Seguro of Funenseg, Brazil. The views expressed herein are those of the authors and do not necessarily reflect the views of the National Bureau of Economic Research.

NBER working papers are circulated for discussion and comment purposes. They have not been peer-reviewed or been subject to the review by the NBER Board of Directors that accompanies official NBER publications.

© 2017 by Sergio L. Franklin, Jr. and Robert S. Pindyck. All rights reserved. Short sections of text, not to exceed two paragraphs, may be quoted without explicit permission provided that full credit, including © notice, is given to the source.

Tropical Forests, Tipping Points, and the Social Cost of Deforestation
Sergio L. Franklin, Jr. and Robert S. Pindyck
NBER Working Paper No. 23272
March 2017
JEL No. C6,Q5,Q57

ABSTRACT

Recent work has suggested that tropical forest and savanna represent alternative stable states, which are subject to drastic switches at tipping points, in response to changes in rainfall patterns and other drivers. Deforestation cost studies have ignored the likelihood and possible economic impact of a forest-savanna critical transition, therefore underestimating the true social cost of deforestation. We explore the implications of a forest-savanna critical transition and propose an alternative framework for calculating the economic value of a standing tropical forest. Our framework is based on an average incremental cost method, as opposed to currently used marginal cost methods, for the design of optimal land-use policy or payments for ecosystem services. We apply this framework to the calculation of the social cost of deforestation of the Amazon rainforest.

Sergio L. Franklin, Jr.
Superintendencia de Seguros Privados
Av. Presidente Vargas 730
Rio de Janeiro 20071-900
Brazil
sergio.franklin@uol.com.br

Robert S. Pindyck
MIT Sloan School of Management
100 Main Street, E62-522
Cambridge, MA 02142
and NBER
RPINDYCK@MIT.EDU

1 Introduction

A number of studies have assessed the economic benefits of a standing tropical forest by estimating the foregone economic benefits resulting from deforestation. The present value of the foregone economic benefits due to one hectare of deforestation has been compared to the present value of future economic benefits of alternative land uses (e.g., crops and cattle ranching) in order to determine the socially optimal land-use policy. To our knowledge, no studies have accounted for the likelihood and possible economic impact of a catastrophic forest-savanna transition.

Ecosystems are exposed to gradual changes in climate, nutrient loading, habitat fragmentation or biotic exploitation, and they are usually assumed to respond in a smooth way. However, studies of forests, lakes, coral reefs, oceans, and arid lands have shown that smooth change can be interrupted by sudden drastic switches to a contrasting state (Scheffer et al. 2001), i.e., a tipping point.

A tipping point can be defined as a situation in which an ecosystem experiences a drastic shift to a new state causing significant changes to its biodiversity and ecosystem services. Under certain environmental conditions, the ecosystem can have two or more alternative stable states, separated by an unstable equilibrium. Tropical forests and savannas represent alternative stable states, which are subject to drastic switches at tipping points in response to changes in rainfall patterns and other drivers (Lobo Sternberg 2001, Warman & Moles 2009, Staver et al. 2015).

We develop a new framework for calculating the marginal economic value of a standing tropical forest, and explore the implications of forest-savanna critical transitions on the design of optimal land-use policy and payments for ecosystem services. We show that marginal cost methods are not appropriate for the design of land-use optimal policy, or for the design of payments for ecosystem services, and propose the use of an average incremental cost method, with the increment properly defined. We also develop a definition of the average incremental social cost of deforestation that, to some extent, follows the approach used in Pindyck (2016) to measure an average social cost of carbon.

In the next section we discuss the social cost of deforestation of the Amazon, as measured by existing marginal cost models. Section 3 explains the nature of the forest-savanna tipping point, and provides evidence that the Amazon rainfall patterns are maintained, in part, by the forest itself. Section 4 proposes a new framework for calculating the marginal social cost of deforestation, taking into account changes in forest resilience. Section 5 introduces the average incremental cost method and shows how, with the in-

crement properly defined, it can be used for the design of optimal land-use policies and payments for ecosystem services. Section 6 concludes.

2 The social cost of deforestation as measured by existing marginal cost models

Other studies have tried to estimate the social cost of Amazon deforestation by applying the concept of total economic value (*TEV*) to assess the economic benefits of a standing tropical forest and the foregone economic benefits resulting from deforestation (Andersen et al. 2002, Margulis 2004, Torras 2000). The total economic value of a natural resource is the sum of its direct use, indirect use, option, and existence values,¹

$$TEV = \text{Direct use value} + \text{Indirect use value} + \text{Option value} + \text{Existence value}, \quad (1)$$

where:

- The direct use value of a standing tropical forest stems from sustainable harvesting of timber and non-timber products, such as nuts, fruits and latex, and from ecotourism.
- The indirect use value depends on the ecological functions performed by the forest, such as water recycling, soil and watershed protection, fire prevention, and carbon storage. Estimates of the indirect use values linked to water recycling, erosion control and watershed protection are rarely made, due to the lack of evidence of the ecological impact of a few hectares of deforestation. Estimates of the indirect use value linked to carbon storage are based on the estimates of net carbon emissions per hectare cleared, and cost of additional ton of carbon released into the atmosphere (i.e., marginal cost of carbon).
- The option value refers to uncertain benefits that can be realized at some point in the future, and reflects the willingness to preserve an option for the potential future use of the forest. Most studies estimate only the option value of biodiversity prospection, based on the prospects of forest biodiversity yielding new drugs, and their future medicinal benefits.
- The existence value is unrelated to both current and optional use, and arises because people are willing to pay for the existence of an environmental asset without ever directly using it. The existence value includes the value that society is willing to pay to secure the survival and well being of other species.

¹See, for example, Pearce (1993).



Figure 1: Map of the Amazon rainforest.

The Amazon rainforest, shown in Figure 1, covers around 530 million hectares of land (Soares-Filho et al. 2006),² and includes territory belonging to nine nations. Brazil holds about 60% of the forest area, followed by Peru with 13%, Colombia with 10%, and Venezuela, Ecuador, Bolivia, Guyana, Suriname and French Guiana with smaller amounts.

The range of ecosystem services and benefits provided by the Amazon rainforest can be classified as private, local/regional public, and global. Private benefits are always local and include, for example, the profits derived from timber and non-timber products that can be harvested from the forest. Local and regional public benefits include water recycling, nutrient recycling, fire control, erosion control and watershed protection. Global benefits include, for example, carbon storage and biodiversity protection.

Over the last few decades the Amazon forest has experienced rapid land use change, with 15% of the original area deforested by 2003 (Soares-Filho et al. 2006). Among the nine nations with forest territory, only Brazil generates and shares spatially detailed information on annual forest extent and change. In particular, the size of the Brazilian Amazon forest has decreased year by year and is now approaching 80% of its original area (INPE 2015). Although Brazil has substantially reduced deforestation rates, these rates are increasing in other Amazon countries (Hansen et al. 2013).

Table 1 shows estimates for the present value of the foregone economic benefits from one hectare of Amazon deforestation. These are marginal values in that they represent

²One hectare is equal to 10,000 square meters, or roughly 2.47 acres.

the change in value for a small change in the forest area, at current deforestation levels. Most of the numbers in this table are derived from estimates from two deforestation cost studies of the Brazilian Amazon, Andersen et al. (2002) and Margulis (2004). In order to make these estimates comparable and accessible, the collected values were updated to 2016 US\$ values (i.e., adjusted for inflation), and converted into present values using a common discount rate, 2.5%, based on survey results in Pindyck (2016). In addition to the sources for each estimate, Table 1 also shows the method used for each calculation.

Table 1: Present value of the foregone economic benefits due to one hectare of deforestation (in US\$ per hectare).

	Present value (in 2016 dollars)	Sources and comments
DIRECT USE VALUE		
Timber products	1,478	Average of Andersen et al. (2002) and Margulis (2004).
Non-timber products	18	Average of Andersen et al. (2002) and Margulis (2004).
Ecotourism	273	Average of Andersen et al. (2002) and Margulis (2004).
Total	1,769	
INDIRECT USE VALUE^(*)		
Carbon storage	996	Average of Andersen et al. (2002) and Margulis (2004).
Water recycling	0	Andersen et al. (2002).
Nutrient recycling	0	Andersen et al. (2002).
Protection against fires	589	Andersen et al. (2002).
Watershed protection	0	Andersen et al. (2002).
Total	1,584	
OPTION VALUE		
Bioprospection	32	Andersen et al. (2002). Margulis (2004) provided an estimate for the average economic value.
Total	32	
EXISTENCE VALUE		
Existence value	54	Andersen et al. (2002). Margulis (2004) provided an estimate for the average economic value.
Total	54	
Grand Total (PV_O)	3,439	

^(*) Most studies say that one hectare of deforestation, at the current deforestation level, will have almost no impact on the ecological functions of water recycling, nutrient recycling and watershed protection.

The present value of the foregone economic benefits from one hectare of deforestation, $PV_{O,t}$, has been incorrectly interpreted as the marginal economic value of a standing tropical forest, and it has been compared to the present value of future economic benefits of alternative land uses (e.g., crops and cattle ranching), AU_t . Deforestation cost studies have shown that, at current deforestation levels, the foregone economic benefits due to deforestation are much lower than the future economic benefits of alternative land uses. Some have argued that the Amazon ecosystem is subject to non-linearities — i.e., sudden dramatic increases in the magnitude of damage once the forest area is reduced below some critical threshold — so that additional deforestation can result in rapid increases in the marginal economic value.³

Something is missing from these marginal economic value calculations. The greatest non-linearity in the total economic value of a tropical forest occurs at the deforestation threshold that triggers the forest-savanna critical transition, but no existing cost studies account for the likelihood and possible impact of a catastrophic shift to the savanna state. In fact, when the first economic impact of forest degradation appears, the forest ecosystem may have already started the self-propagating transition to the savanna state, which will almost certainly be irreversible. We turn to that next.

3 The forest-savanna tipping point

Recent studies have extensively investigated the tree cover distribution of tropical forests and savannas in Africa, Australia and South America. Hirota et al. (2011) analyzed the relationship between the distribution of tree cover and mean annual precipitation, and found that the frequency distribution of tree cover in the tropics has three distinct modes, corresponding to tropical forest, savanna, and treeless states. Additionally, tree cover does not increase gradually with rainfall, but is constrained by ranges which could be identified as treeless (0 to 5%), savanna (around 20%) or tropical forest (around 80%). The rarity of places with 5-10% or 50-60% tree cover suggests that these situations are unstable because of the positive feedbacks among tree cover, precipitation, fire and drought.

Precipitation is a major driver of past and recent shifts in the extension of tropical forests and savannas (Bowman et al. 2010, Mayle et al. 2007), though gradual changes in precipitation can have little apparent effect on tree cover. Staver et al. (2011) used

³See, for example, Torras (2000) and Andersen et al. (2002). According to Strand (2017), losses of rain-forest likely lead to less rainfall and increased forest fire risk, which in turn increase marginal forest value by making primary forest loss avoidance more valuable.

datasets on tree cover, rainfall, fire frequency, and soil categories to show that with intermediate rainfall levels having mild seasonality, both forest and savanna are common, and only fire feedbacks can explain the bimodalities in tree cover.⁴ In particular, they found that a large part of the Amazon forest supports biome bistability, i.e., although it is currently in the forest state, a sufficiently severe perturbation can induce a self-propagating shift to an open savanna.

Evidence is accumulating that the Amazon rainfall patterns are maintained in part by the forest itself, through contributions of water vapor to the atmosphere (Coe et al. 2009), and the precipitation patterns are bound to change with severe changes in forest cover (Malhi et al. 2007). According to Fearnside (1997), approximately half of the precipitation in the Amazon forest is derived from water that recycles through evapo-transpiration. This is because the Andes Mountains function as a six-kilometer high barrier that blocks the water vapor above the forest, and air currents carry the moisture across the Amazon region and towards the southeast and center of the South American continent.

A number of vegetation-climate models have suggested that the entire Amazon forest may cross a tipping point if deforestation exceeds about 40% of the original forest area, after which the ecosystem will experience a self-propagating transition to the savanna state (which can take several decades to a century to fully reach the new equilibrium). For example, Sampaio et al. (2007) assessed the climate impacts of converting the Amazon rainforest into pastures or soybean croplands, and found an accelerating decrease of rainfall for increasing deforestation for both classes of land use conversion, while the reduction in precipitation was more evident when deforestation exceeds 40% of the original forest cover. Nepstad et al. (2008) suggested that the economic, ecological and climatic systems of Amazonia may be interacting to move the forest towards a near-term tipping point, and predicted a large-scale substitution of the Amazon forest by savanna-like vegetation by the end of the twenty-first century. Nobre & Borma (2009) found that tipping points for the Amazon forest may exist for total deforested area greater than 40% and for global warming greater than 3 – 4°C.⁵

The risk of a forest-savanna critical transition should therefore be a major concern of

⁴Fire spread depends on a continuous grass layer, so that the lower is the tree cover, the higher is the fire spread, which causes further forest dieback and further reduction in tree cover.

⁵In addition, Davidson et al. (2012) showed that the changes in rainfall and discharge associated with deforestation already observed in the southern and eastern Amazon demonstrate potential for significant vegetation shifts and further feedbacks to climate and discharge, and Pires & Costa (2013) showed that while inner forest regions remain inside a rainforest bioclimatic envelope, outer forest regions may cross the forest-savanna bioclimatic threshold even at low deforestation levels.

policymakers, and provides an argument in favor of stringent reductions in deforestation.

4 The social cost of deforestation accounting for changes in forest resilience

The existence of a forest-savanna tipping point implies that changes in forest resilience affect the marginal economic value of a standing forest, and must be accounted for when calculating that marginal value. *Forest resilience* is the capacity of a forest to respond to a perturbation or disturbance by resisting damage and returning to its original condition. Such perturbation or disturbance can include stochastic events such as fire, drought and flooding, and human activities such as deforestation. There are large uncertainties about the effect of deforestation on the amount of in-forest precipitation, and the effect of forest precipitation on the ecosystem resilience. However, most scientists agree that deforestation reduces forest precipitation, and the greater the deforestation, the less resilient the forest will be.

One can think of forest resilience as a multidimensional function that depends on a number of ecosystem-wide variables. All other things held constant, forest resilience is a function of rainfall patterns. Because the Amazon rainfall patterns are in large part maintained by the forest itself, forest resilience can be represented as a function of the proportion of deforested area at any time.

4.1 Analytical framework

Let A denote the Amazon's original forest area, which at time t is the sum of the forest area, F_t , and the deforested area, D_t , and let d_t denote the proportion of total area that is deforested. Forest resilience can be expressed as $R_t = f(d_t)$, where, $f(0) = \alpha, \alpha \leq 1$, and $f(h) = 0$, with h the deforestation threshold that triggers the forest-savanna transition. Assume that $f(d)$ is monotonically decreasing and strictly concave in the interval $[0, h]$, $f(d) = 0$ for $d \in [h, 1]$, and once the deforestation threshold is reached, system dynamics change irreversibly and the ecosystem never gets back to the forest state.⁶ The functional form of the forest resilience function can depend on a number of ecosystem-wide variables, such as precipitation volume and seasonality, soil characteristics, and temperature.

⁶In fact, to get the ecosystem back to the forest state, it would be necessary to reduce the size of the deforested area far below the deforestation threshold, due to the hysteresis effect. By "irreversibly", we mean it would be extremely costly to get the ecosystem back to the original state of tropical forest.

Its concavity can be inferred from the positive feedbacks among tree cover, precipitation, fire and drought.

The total ecosystem area is divided into a number of grid cells, of one hectare each, centered around the latitude and longitude coordinates x and y . Let $V_\psi(S_t)$ denote the economic value of a one-hectare grid cell at time t , which is a function of a vector of state variables, $S_t(x, y)$. The subscript ψ indicates whether the grid cell is in forest state ($\psi = F$), savanna state ($\psi = S$), or treeless state ($\psi = O$), so that $V_F(S_t)$, $V_S(S_t)$ and $V_O(S_t)$ denote the economic value of a one-hectare grid cell in each state.

The marginal economic value of a standing forest, $V_F(S_t)$, is the marginal social cost of deforestation, i.e., the total cost to society of an additional hectare of deforested area. Let $MSCD(S_t)$ denote the marginal social cost of deforestation, so that $MSCD(S_t) = V_F(S_t)$, where $S_t(x, y) = [E_t(x, y), N_t(x, y), R_t]$ is a vector of state variables. Here $E_t(x, y)$ and $N_t(x, y)$ denote the set of hectare-wide economic and nature state variables, identified by the grid coordinates (x, y) , and R_t is a measure of forest resilience (i.e., an ecosystem-wide state variable):

- $E_t(x, y)$ includes a number of economic state variables, such as price of timber and non-timber products, price of energy, price of carbon, logistics and extractive costs;
- $N_t(x, y)$ includes nature state variables, such as vegetation type, average tree cover, soil characteristics, temperature, day length hours, evapo-transpiration rate, biodiversity; and
- R_t is the measure of forest resilience, expressed as a function of the proportion of deforested area, $R_t = f(d_t)$. The forest-savanna tipping point is characterized when the forest resilience reaches zero, and that happens when the proportion of deforested area, d_t , reaches the deforestation threshold h .

For simplicity, assume that the economic and nature state variables do not depend on the geographic location of the grid cell inside the forest area, i.e., $E_t(x, y) = E_t$, $N_t(x, y) = N_t$, and the vector of nature state variables can only take two values: $N_t = N_F$ if the ecosystem is in the forest state, and $N_t = N_S$ if the ecosystem is in the savanna state.⁷ Suppose the ecosystem is currently in the forest state (i.e., $\psi = F$) so that the total economic value

⁷ E_t and N_t can be interpreted as the geographic averages of the economic and nature state variables inside the forest area, respectively. In a more general case, the forest area could be split into a number of regions, with similar economic and nature state variables.

of the forest area is $TEV_t = \overline{V}_F F_t$, where \overline{V}_F is the average economic value of a representative hectare of forest, and F_t is the size of forest area at time t . The average economic value of a representative hectare of forest represents the present value of the foregone economic benefits resulting from a large deforestation area divided by the total size of deforested area.

Suppose that at time t^* forest resilience is zero, so one additional hectare of deforestation makes the ecosystem cross the forest-savanna tipping point and experience a self-propagating shift to an open savanna. The total economic value of the forest area is $TEV_{t^*} = \overline{V}_F F_{t^*}$. After crossing the forest-savanna tipping point, the total economic value of the forest area changes to the value of an equivalent savanna area, at time $t^* + \tau$, $TEV_{t^*+\tau} = \overline{V}_S F_{t^*}$, where τ is the time period until the ecosystem reaches the new equilibrium state. Assume that the forest-savanna transition occurs at a constant rate of time.

The marginal social cost of deforestation must be calculated for two different regimes. In the pre-threshold regime (i.e., $R_t > 0$):

$$MSCD = V_F(E_t, N_F, R_t) = \text{Change in } TEV \text{ due to an additional hectare of deforestation,} \quad (2)$$

which is a function of the economic and nature state variables, and the forest resilience, at time t . At the threshold (i.e. $R_{t^*} = 0$):

$$MSCD = V_F(E_{t^*}, N_F, 0) = PV_{O_{t^*}} + \left(\frac{1}{r\tau}\right) [\overline{V}_F - \overline{V}_S] F_{t^*} (1 - e^{-r\tau}), \quad (3)$$

where $PV_{O_{t^*}}$ is the present value of the foregone economic benefits due to one hectare of deforestation at time t^* , r is the discount rate, and τ is the time until the ecosystem reaches the savanna state.

4.2 Economic impact of a forest-savanna transition

In order to estimate the economic loss from a forest-savanna transition, it is necessary to estimate the change in average economic value of a representative hectare of forest that undergoes the state transition, $\Delta \overline{V}_{F,S} = [\overline{V}_F - \overline{V}_S]$.

The estimates for the average direct use values of forest are the same as the ones for the direct use values shown in Table 1. Given that the characteristic tree cover of savanna is approximately 25% of the characteristic tree cover of forest (Hirota et al. 2011), and sustainable timber harvesting is responsible for a great share of the forest total direct use value, the estimates for the average direct use values of savanna are 25% of the average direct use values of tropical forest.

The average indirect use value from carbon storage is the product of the average carbon stock per hectare of forest/savanna and the average social cost of carbon. For forest, the average carbon stock was obtained by averaging the rates indicated in Andersen et al. (2002) and Margulis (2004), of 150 tC/ha and 100 tC/ha respectively, yielding 125 tC/ha. For savanna, the average carbon stock was assumed to be 25% of the average carbon stock of forest, that is 31.25 tC/ha. For the average social cost of carbon, we use US\$80/tC, as suggested in Pindyck (2016).

The indirect use values linked to water resources can be consolidated into a single economic benefit, water resource values, including the regulation of water flows, precipitation and river discharge. These ecosystem services provide a series of economic benefits to the region, in large measure related to agricultural outputs and electric power generation, mostly in the vicinity of the forest, but also in distant areas by cycling atmospheric water in the form of aerial rivers to the southeast and center of the South American continent (Fearnside 2003). For forest, the average indirect use value linked to water resources was drawn from the average estimate in De Groot et al. (2012). Here we assume that the forest-savanna transition would reduce evapotranspiration by 75%, further reducing forest precipitation by 37.5%, so that the estimate for the average indirect use value linked to water resources after the forest-savanna transition is 62.5% of the estimated value for forest. With regards to hydropower generation, Stickler et al. (2013) analyzed the effect of large-scale deforestation on the Amazon region's water cycle, and concluded that a loss of 40% of the Amazon rainforest would reduce the mean annual energy generation of the Belo Monte energy complex by 38%. We estimate that if the forest-savanna transition causes a 60% reduction in the annual hydroelectric generation in the Amazon and Tocantins basins, and a 20% reduction in the Paraná basin, the loss of hydroelectric potential will account for approximately 44% of the total change in the indirect use value linked to water resources.⁸

The average option value was obtained from Margulis (2004). We used information in De Groot et al. (2012) to calculate the ratio between the median values of the gene pool protection/conservation service provided by tropical forests and woodlands. This ratio was multiplied by the estimate for the average option value of forest, in order to find the estimate for the average option value of savanna, and the change in average economic value of a representative hectare of forest that undergoes the state transition.

⁸The Belo Monte complex, located in the eastern Amazon, will be one of the world's largest hydropower plants after construction is concluded. The Amazon, Tocantins and Paraná basins are respectively responsible for 40.5%, 10.3% and 23.5% of Brazilian hydroelectric potential.

Table 2 summarizes the estimates for the change in average economic value of a representative hectare of forest that undergoes the forest-savanna transition. These are average values in the sense that they represent the change in value for a large change in the forest area. The numbers in this table are derived from estimates provided by Andersen et al. (2002) and Margulis (2004), the social cost of carbon in Pindyck (2016), averages of ecosystem service values summarized in De Groot et al. (2012), and our calculations. All of these estimates were updated to 2016 US\$ values and converted into present values using a 2.5% discount rate.

Table 2: The change in average economic value of a representative hectare of forest that undergoes the forest-savanna transition (in US\$ per hectare).

	$\Delta \overline{V}_{F,S} = [\overline{V}_F - \overline{V}_S]$ (in 2016 dollars)	Sources and comments
DIRECT USE VALUE		
Timber products	1,109	Average of Andersen et al. (2002) and Margulis (2004).
Non-timber products	13	Average of Andersen et al. (2002) and Margulis (2004).
Ecotourism	205	Average of Andersen et al. (2002) and Margulis (2004).
Total	1,327	
INDIRECT USE VALUE		
Carbon storage	7,500	Andersen et al. (2002), Margulis (2004), Pindyck (2016) and authors' calculations.
Water recycling	6,374	De Groot et al. (2012) and authors' calculations.
Total	13,874	
OPTION VALUE		
Bioprospection	985	Margulis (2004), De Groot et al. (2012) and authors' calculations.
Total	985	
EXISTENCE VALUE		
Existence value	1,596	Margulis (2004). Andersen et al. (2002) provided an estimate for the marginal economic value.
Total	1,596	
Grand Total ($[\overline{V}_F - \overline{V}_S]$)	17,782	

4.3 The marginal cost model

In what follows, we make two simplifying assumptions. First, we ignore the opportunity cost of depletion, and assume that the marginal economic value of a standing tropical forest, i.e., the marginal social cost of deforestation, equals the present value of the foregone economic benefits from an additional hectare of deforestation, taking into account the likelihood and possible impact of a forest-savanna transition. Second, we assume that the economic and nature state variables remain constant until the ecosystem crosses the forest-savanna tipping point. In other words, there is no degradation of the remaining forest area (e.g., no changes to the average tree cover and biodiversity) and no changes in the relevant economic variables (e.g., prices of timber and non-timber products).

Uncertainty over the deforestation threshold that triggers the forest-savanna transition, H , can be described by a probability distribution, $f_H(h)$. The Beta distribution is a family of continuous probability distributions defined on the interval $[0, 1]$, parameterized by two positive shape parameters, a and b , that is often used to model random variables limited to intervals of finite length. We model the deforestation threshold as a linear transformation of a Beta distribution, $H = l + (u - l)X$, where $X \sim \text{Beta}(a, b)$, and l and u denote the lower and upper bounds of the threshold. The two shape parameters, a and b , and the lower and upper bound parameters, l and u , provide flexibility in modeling the threshold. Thus the probability density function of H is

$$f_H(y) = \begin{cases} \frac{1}{(u-l)} \frac{1}{B(a,b)} \left(\frac{y-l}{u-l}\right)^{a-1} \left(\frac{u-y}{u-l}\right)^{b-1}, & \text{for } y \in (l, u), \text{ and} \\ 0, & \text{otherwise,} \end{cases} \quad (4)$$

where $B(a, b)$ is the beta function, a normalized constant to ensure that $f_H(y)$ integrates to one, and the parameters a , b , l and u reflect our uncertainty about the deforestation threshold. The higher are the values of a and b , the more concentrated is the probability distribution around the mean. If $a = b$ the probability distribution is symmetric around the mean (and $E[H] = (l + u)/2$); if $a \neq b$ it is asymmetric. The closer are the values of l and u , the more confident we are about the exact threshold.

The range of studies suggesting that the Amazon forest may cross a tipping point if deforestation exceeds 40% suggests that $E[H]$ is close to 40% and $f_H(H)$ is slightly concentrated around the mean. The uncertainty about the deforestation threshold seems to be symmetric so that we assume $a = b > 1$. The fact that the Amazon rainforest has experienced severe droughts in the last few years suggests that the current deforestation level (of approximately 20%) may be close to the lower bound of the deforestation threshold,

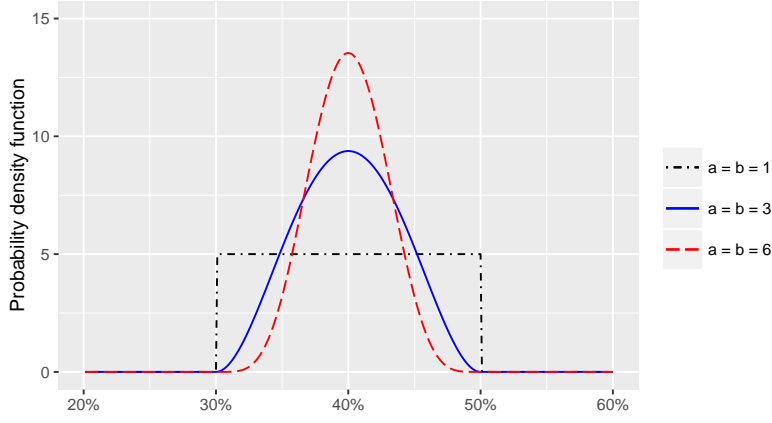


Figure 2: Probability density function of the deforestation threshold for different shape parameters.

Figure 2 illustrates the probability density function of the deforestation threshold for different values of $a = b > 1$, when $l = 30\%$ and $u = 50\%$ ($E[H] = 40\%$). In what follows, we assume that $3 \leq a = b \leq 6$.

Policymakers do not know the precise location of the deforestation threshold that will trigger the forest-savanna transition. Crossing that threshold will shift the ecosystem from the pre-threshold regime to a post-threshold regime with permanently altered ecosystem dynamics. If the proportion of deforested area is currently d , the probability that an additional hectare of deforestation will bring the ecosystem to the forest-savanna tipping point is $P(d < H < d + \delta \mid H > d)$, where δ denotes the change in the proportion of deforested area due to an additional hectare of deforestation.

The marginal social cost of deforestation of the Amazon rainforest, $MSCD_{Am}$, is the expected value of the change in TEV due to an additional hectare of deforestation. From Eqs. (2), (3) and (4), it can be shown that for $d < l$, $MSCD_{Am} = PV_O$, and for $l \leq d < u$,

$$MSCD_{Am} = PV_O + \left(\frac{1}{r\tau}\right) [\overline{V}_F - \overline{V}_S] A (1-d) (1 - e^{-r\tau}) P(d < H < d + \delta \mid H > d), \quad (5)$$

where A , r and τ respectively denote the original size of the Amazon forest, the long-term discount rate and the time period to fully reach the savanna state, $\delta = 1/A$ denotes the change in the proportion of deforested area due to an additional hectare of deforestation, and estimates for PV_O and $[\overline{V}_F - \overline{V}_S]$ are shown on Tables 1 and 2.⁹

⁹To keep the notation simple, the index t has been omitted when no confusion arises. Take $PV_O = \text{US\$3,439}$, $[\overline{V}_F - \overline{V}_S] = \text{US\$17,782}$, $A = 620$ million hectares, $\tau = 75$ years, and $r = 2.5\%$.

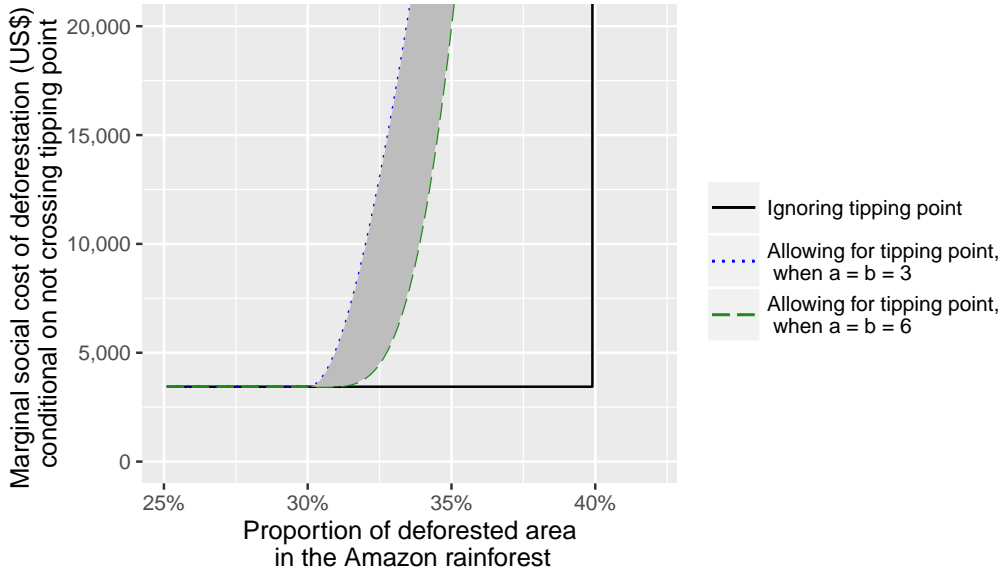


Figure 3: Marginal social cost of deforestation observed by the Amazon region.

Figure 3 shows, from the perspective of the Amazon region, how the marginal social cost of deforestation varies with the proportion of deforested area, ignoring and then taking into account a tipping point. First, ignoring the tipping point (as current cost studies do), the present value of the foregone economic benefits from one hectare of deforestation remains constant until the ecosystem reaches the forest-savanna tipping point, at which point the foregone economic benefits jump to \$3.0 trillion. In this case we might estimate a marginal social cost of deforestation that is always lower than the marginal economic benefits of alternative land uses, so that the forest is continually converted into agricultural land, until there is a sudden transition to the savanna state, causing dramatic losses to all Amazon countries (and the whole world). On the other hand, taking into account the existence of a forest-savanna tipping point, the marginal social cost of deforestation starts rising when the proportion of deforested area surpasses the lower bound of the deforestation threshold. We might then observe rapid increases in damages once the forest area is reduced below some critical point, which may function as an early warning to stop deforestation. In the figure, the probability distribution of the deforestation threshold is modeled with $l = 30\%$, $u = 50\%$ and $3 \leq a = b \leq 6$, and the gray area illustrates the range of possible values for the marginal social cost of deforestation.

Now take the perspective of an individual country, Brazil, where 60% of the Amazon forest is located. Let $d_{B,t}$ denote the proportion of deforested area of the Brazilian Ama-

zon, and $d_{O,t}$ denote the proportion of deforested area of the other Amazon countries, so that $d_t = .6d_{B,t} + .4d_{O,t}$. If H denotes the (unknown) deforestation threshold of the Amazon forest, then for any given values of H and $d_{O,t}$, the deforestation threshold of the Brazilian Amazon that triggers the forest-savanna transition, H_B , is such that $H_B = (H - .4d_{O,t})/.6$. Thus, the deforestation threshold of the Brazilian Amazon, H_B , is a linear transformation of the random variable H , and its probability density function is

$$f_{H_B}(y) = \begin{cases} \frac{1}{(u_{B,t} - l_{B,t})} \frac{1}{B(a, b)} \left(\frac{y - l_{B,t}}{u_{B,t} - l_{B,t}} \right)^{a-1} \left(\frac{u_{B,t} - y}{u_{B,t} - l_{B,t}} \right)^{b-1}, & \text{for } y \in (l_{B,t}, u_{B,t}), \\ 0, & \text{otherwise,} \end{cases} \quad (6)$$

where $l_{B,t} = (l - .4d_{O,t})/.6$, $u_{B,t} = (u - .4d_{O,t})/.6$, and $B(a, b)$ is the beta function.

If the proportion of deforested area of the Brazilian Amazon is d_B , the probability that an additional hectare of deforestation will bring the ecosystem to the forest-savanna tipping point is $P(d_B < H_B < d_B + \delta_B \mid H_B > d_B)$, where δ_B denotes the change in the proportion of deforested area of the Brazilian Amazon from an additional hectare of deforestation. The marginal social cost of deforestation observed by Brazil, $MSCD_{Br}$, is the expected value of the change in TEV of the Brazilian Amazon due to an additional hectare of deforestation. From Eqs. (2), (3) and (6), it can be shown that for $d_B < l_B$, $MSCD_{Br} = PV_O$, and for $l_B \leq d_B < u_B$,

$$MSCD_{Br} = PV_O + \left(\frac{1}{r\tau} \right) [\overline{V_F} - \overline{V_S}] A_{Br} (1 - d_B) (1 - e^{-r\tau}) P(d_B < H_B < d_B + \delta_B \mid H_B > d_B), \quad (7)$$

where A_{Br} , r and τ denote the original size of the Brazilian Amazon forest, the discount rate, and the time to fully reach the savanna state, $\delta_B = 1/A_{Br}$ denotes the change in proportion of deforested area of the Brazilian Amazon from an additional hectare of deforestation, and estimates for PV_O and $[\overline{V_F} - \overline{V_S}]$ are in Tables 1 and 2.¹⁰

Figure 4 shows, from the Brazilian perspective, how the marginal social cost of deforestation varies with the proportion of deforested area, when the other Amazon countries convert 20%, 40% and 60% of their forest areas to alternative land uses. The existence of a forest-savanna tipping point implies that each Amazon country observes a marginal social cost of deforestation that depends on the land-use policy adopted by the others. This gives rise to a coordination problem in which all parties can realize mutual gains, but only by making mutually consistent decisions. (Even if all Amazon countries decide

¹⁰To keep the notation simple, the index t has been omitted when no confusion arises. Take $PV_O = \text{US\$}3,439$, $[\overline{V_F} - \overline{V_S}] = \text{US\$}17,782$, $A_{Br} = 372$ million hectares, $\tau = 75$ years, and $r = 2.5\%$.

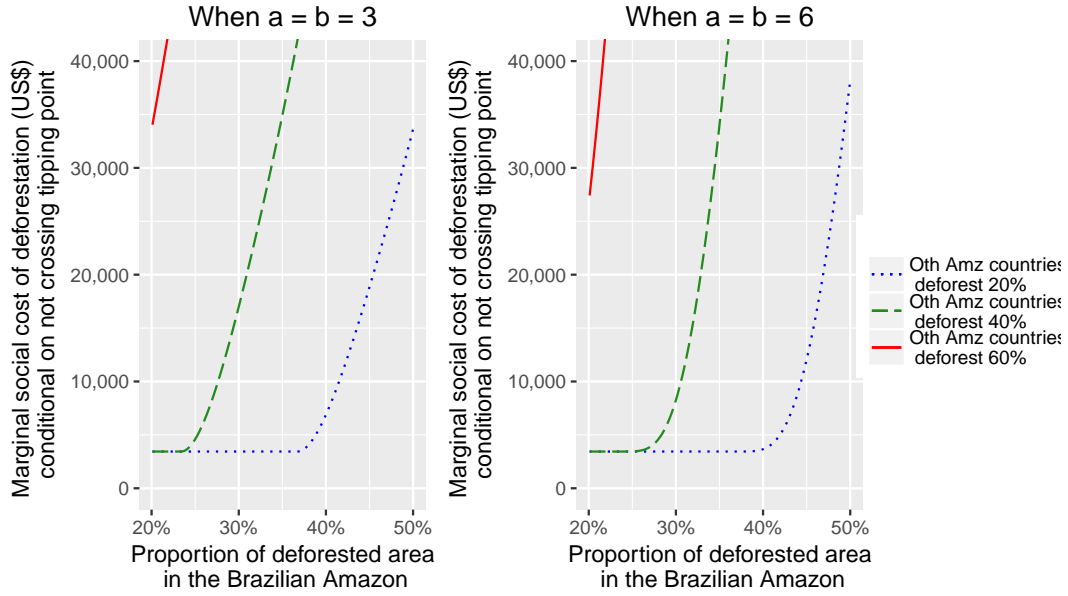


Figure 4: Marginal social cost of deforestation observed by Brazil.

to cooperate, it will be very difficult to monitor compliance because only Brazil generates and shares spatially detailed information on annual forest extent and change.) Here, the probability distribution of the deforestation threshold has been modeled with $l = 30\%$, $u = 50\%$, and two different sets of shape parameters, $a = b = 3$ and $a = b = 6$.

The marginal social cost of deforestation is not well-suited to the design of optimal land-use policy. First and most importantly, the marginal social cost derived above is conditional on the ecosystem not having crossed the forest-savanna tipping point. However, when the proportion of deforested area is d_{t_0} , we cannot know whether the ecosystem will cross the tipping point in the future. Additionally, individual countries do not devise optimal land-use policies for each hectare of forest. Instead, they analyze alternative land uses of much larger areas in order to calculate the present values of foregone and potential future economic benefits. As shown in the next section, an incremental cost measure is thus better suited for the design of land-use policy.

5 The average incremental social cost of deforestation

We propose an average incremental cost measure as an alternative measure of the social cost of deforestation. In general, the incremental cost of a product or service is the increase in total cost following the introduction of an additional increment of production.

That increment can take several forms: one additional unit of the product or service (in which case the incremental cost is the marginal cost), the entire output over some period, output for a group of products or services, etc. In our case, the increment is the size of an additional area that is deforested. The average incremental social cost of deforestation is the change in total economic value due to that incremental deforestation, ΔTEV , divided by the change in the forest area, ΔF , i.e., $AISCD = \Delta TEV / \Delta F$. How large that increment should be depends on whether the cost study will be used for the design of optimal land-use policy or payments for ecosystem services.

If the objective of the cost study is the design of optimal land-use policy, it is important to understand how the average incremental social cost of deforestation changes as the planned/target deforestation level changes. To see this, take the perspective of an individual country, say Brazil, and let $\Delta F = (d_B - d_{B,t_0})A_{Br}$ denote the size of an additional area of the Brazilian Amazon that is deforested. The average incremental social cost of deforestation for Brazil at time t_0 , $AISCD_{Br}$, is given by $E\left[\frac{\Delta TEV}{\Delta F} \mid H_B > d_{B,t_0}\right] = \frac{1}{\Delta F}E[\Delta TEV \mid H_B > d_{B,t_0}]$. From Eqs. (2), (3) and (6), it can be shown that for $d_B < l_B$, $AISCD_{Br} = PV_O$, and for $l_B \leq d_B < u_B$,

$$AISCD_{Br} = PV_O + \left(\frac{1}{r\tau}\right)[\overline{V}_F - \overline{V}_S] \left(\frac{1 - d_B}{d_B - d_{B,t_0}}\right) (1 - e^{-r\tau}) P(H_B < d_B \mid H_B > d_{B,t_0}) \quad (8)$$

where d_{B,t_0} denotes the proportion of deforested area of the Brazilian Amazon at time t_0 , r and τ denote the discount rate and time to fully reach the savanna state, and estimates for PV_O and $[\overline{V}_F - \overline{V}_S]$ are in Tables 1 and 2.¹¹ Again, each Amazon country observes an average incremental social cost of deforestation that depends on the land-use policies adopted by the others. Therefore, prior to designing its own policy, each country should carefully assess the planned land-use policies of the other Amazon countries, and then compare the average incremental social cost of deforestation with the future economic benefits of alternative land uses.

On the other hand, if the objective of the cost study is the design of payments for ecosystem services, the choice of increment should be the additional deforested area that will bring the entire ecosystem to the forest-savanna tipping point, i.e., the random variable $\Delta F = (H - d_{t_0})A$, where H is the (unknown) deforestation threshold, d_{t_0} is the proportion of deforested area at time t_0 , and A is the original size of the Amazon forest. Using the assumptions set forth on subsection 4.3, it can be shown that

¹¹To keep the notation simple, the index t has been omitted when no confusion arises. Take $PV_O = \text{US\$3,439}$, $[\overline{V}_F - \overline{V}_S] = \text{US\$17,782}$, $d_{B,t_0} = 20\%$, $\tau = 75$ years, and $r = 2.5\%$.

$\Delta TEV = PV_O(H - d_{t_0})A + \left(\frac{1}{r\tau}\right)[\overline{V}_F - \overline{V}_S](1 - e^{-r\tau})(1 - H)A$, and the average incremental social cost of deforestation for the incremental deforested area that will bring the ecosystem to the tipping point is

$$\begin{aligned} AISCD_{tip} &= E \left[\frac{\Delta TEV}{\Delta F} \mid H > d_{t_0} \right] \\ &= \int_{d_{t_0}}^u \frac{PV_O(y - d_{t_0}) + \left(\frac{1}{r\tau}\right)[\overline{V}_F - \overline{V}_S](1 - e^{-r\tau})(1 - y)}{(y - d_{t_0})} f_{H|H>d_{t_0}}(y) dy, \end{aligned} \quad (9)$$

where $f_{H|H>d_{t_0}}(y) = \frac{d}{dy}P(H \leq y, H > d_{t_0})/P(H > d_{t_0})$, and $f_H(y)$ is the probability density function of H , given by Eq. (4).

As shown in Section 3, the ecosystem services and benefits provided by the Amazon rainforest can be classified as private, local/regional public, and global. The local and regional benefits are enjoyed by all Amazon countries, while the global benefits can be seen as positive externalities enjoyed by the rest of the world as a result of the Amazon countries' efforts to preserve the rainforest. Escalating global demands for new agricultural land should create incentives for alternative land uses. It seems reasonable for the international community to provide appropriate incentives, in the form of payments for ecosystem services, to ensure the continued provision of global ecosystem services and benefits such as carbon storage, biodiversity protection and existence value. To see this, imagine a situation where global demands for new agricultural land keeps increasing over time, so that the potential benefits from alternative land uses are always higher than the local and regional benefits from a standing tropical forest. Then, in the absence of appropriate incentives, one should expect that the forest area will keep being converted into agricultural land until the ecosystem crosses the forest-savanna tipping point.

Let $AISCD_{glb}$ denote the average incremental social cost of deforestation due to the loss of the global ecosystem services and benefits (e.g., carbon storage, biodiversity protection and existence value). A risk-neutral global party may be willing to pay up to that value to ensure the continued provision of such services/benefits, because this is how much it expects to lose on average, per hectare of deforested area. Similarly, let $AISCD_{lo/re}$ denote the average incremental social cost of deforestation due to the loss of the local and regional ecosystem services and benefits. The Amazon countries may be willing to pay up to that value to ensure the continued provision of such services/benefits, because this is how much they expect to lose on average, per hectare of deforested area.

Both $AISCD_{glb}$ and $AISCD_{lo/re}$ can be found by solving Eq.(9) for the respective ranges of ecosystem services and benefits. Table 3 shows a sensitivity analysis of $AISCD_{glb}$

Table 3: Sensitivity analysis of the average incremental social cost of deforestation due to the loss of global and local/regional ecosystem services and benefits (in US\$ per hectare).

$AISCD_{glb}$			
Parameter values	$l = .30$ and $u = .50$ ($E[H] = 40\%$)	$l = .35$ and $u = .55$ ($E[H] = 45\%$)	$l = .40$ and $u = .60$ ($E[H] = 50\%$)
$a = b = 3$	15,450	11,448	8,869
$a = b = 6$	15,107	11,282	8,774
$AISCD_{lo/re}$			
Parameter values	$l = .30$ and $u = .50$ ($E[H] = 40\%$)	$l = .35$ and $u = .55$ ($E[H] = 45\%$)	$l = .40$ and $u = .60$ ($E[H] = 50\%$)
$a = b = 3$	13,333	10,277	8,306
$a = b = 6$	13,071	10,149	8,234

and $AISCD_{lo/re}$ with respect to the probability distribution parameters of the deforestation threshold, l , u , a and b . The results show that $US\$8,234 \leq AISCD_{lo/re} \leq US\$13,333$, and $US\$8,774 \leq AISCD_{glb} \leq US\$15,450$. In fact, $AISCD_{glb}$ should be interpreted as a ceiling price. The international community may be willing to pay any amount equal to or below this ceiling price, which is enough to make the value of the local and regional ecosystem services/benefits provided by the Amazon rainforest match the potential economic benefits of alternative land uses.

Existing cost studies have compared the present value of the foregone economic benefits due to one hectare of deforestation, PV_O (equal to US\$3,439, as shown in Table 1), with the present value of future economic benefits of alternative land uses, AU (in the range between US\$10,000 and US\$36,000),¹² and they have found that, at current deforestation levels, the foregone economic benefits due to deforestation are much lower than the potential economic benefits of alternative land uses. The framework proposed in this paper shows that when tipping points are taken into account, the social cost of deforestation can be much higher than the foregone economic benefits due to one hectare of deforestation, and payments for ecosystem services may be necessary to ensure the continued provision of global ecosystem services and benefits.

In fact, the present value of future economic benefits of alternative land uses, AU , should be compared to how much each party expects to lose on average if forest area is converted into agricultural land (i.e., how much each party would pay to ensure the

¹²See, for example, Andersen et al. (2002), Margulis (2004) and Andersen (2015).

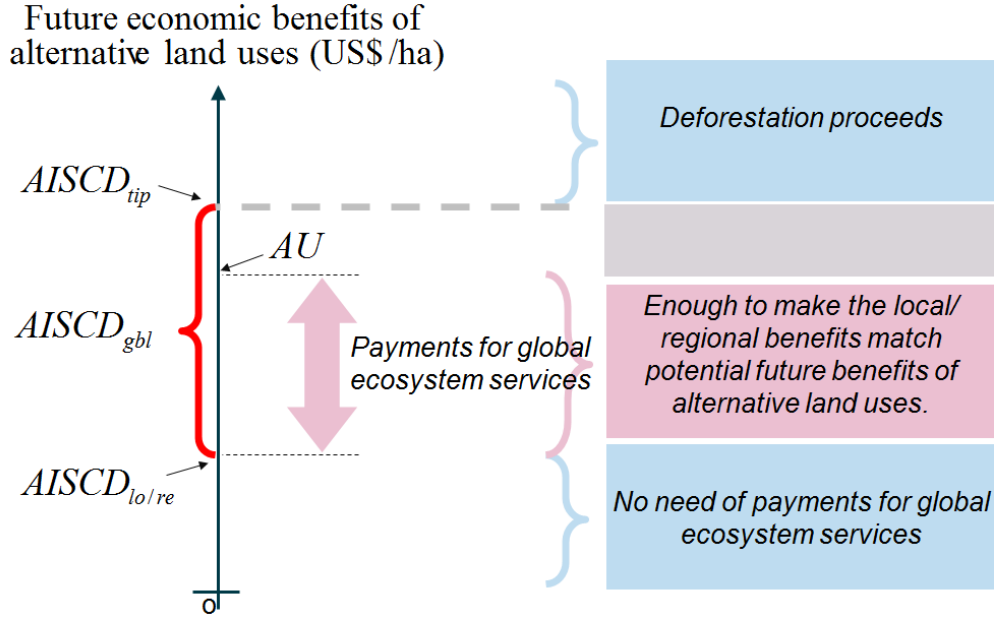


Figure 5: Relationships among average incremental social cost of deforestation ($AISC D$), future economic benefits of alternative land uses (AU), and payments for ecosystem services.

continued provision of the applicable ecosystem services and benefits):

- If $AU \leq AISC D_{lo/re}$, payments for global ecosystem services are not necessary, because the Amazon countries will find it is in their best interest to protect the rainforest so as to ensure the continued provision of the local and regional ecosystem services/benefits.
- If $AISC D_{lo/re} < AU \leq AISC D_{tip}$, the international community may be willing to pay any amount equal or below $AISC D_{gbl}$ which is enough to make the value of the local and regional ecosystem services/benefits match the future economic benefits of alternative land uses.
- If $AU > AISC D_{tip}$, one should expect the Amazon deforestation to proceed up to the point where the non-linearities in the Amazon ecosystem, now captured by the proposed framework, make it economically efficient to protect the rainforest.

Figure 5 illustrates the relationships among the distinct pieces of the average incremental social cost of deforestation, the future economic benefits of alternative land uses, and payments for global ecosystem services, for the case when $AISC D_{lo/re} < AU \leq AISC D_{tip}$.

6 Concluding remarks

If it were certain that the worst outcomes from deforestation could be addressed successfully in the future, either through natural regeneration or reforestation, so that the forest-savanna tipping point would never be reached, then we could rely on current marginal cost calculations that ignore forest resilience. However, if plausible scenarios exist in which the forest ecosystem undergoes a transition, the marginal economic value of a standing forest can be much higher than the present value of the foregone economic benefits from one hectare of deforestation.

Existing cost studies tell us nothing about the likelihood and possible economic impact of a catastrophic forest-savanna transition, which should be a major policy concern. Ignoring the existence of a tipping point means underestimating the social cost of deforestation, favoring the adoption of alternative land uses, and facing the risk of an unexpected shift to the savanna state, which will result in dramatic losses of ecosystem services and benefits.

This paper proposes an alternative framework for calculating the economic value of a standing tropical forest, and explores the implications of a forest-savanna critical transition for the design of optimal land-use policy and payments. The economic value of a one-hectare grid cell of forest is modeled as a function of a number of hectare-wide (economic and nature) state variables and one ecosystem-wide state variable, forest resilience. This framework allowed us to disentangle the impact of deforestation on the hectare-wide state variables from that on the ecosystem-wide state variable. The marginal economic value of a standing tropical forest is then measured by the change in total economic value from an additional hectare of deforestation. We applied this framework to the estimation of the social cost of deforestation of the Amazon rainforest.

We have shown advantages to using an average incremental cost method for the design of optimal land-use policy and payments for ecosystem services. For land-use policy, the increment should be the size of an additional area that is deforested. For the design of payments for ecosystem services, the increment should be the additional deforested area that will bring the entire ecosystem to the forest-savanna tipping point (i.e., a random variable). Current marginal cost models do not provide an advance warning of an approaching tipping point, so that Amazon countries must take into account the likelihood and possible impact of a catastrophic shift to the savanna state. Additionally, the social cost of deforestation observed by one Amazon country depends on the land-use policies of other countries, and payments for ecosystem services may be necessary to ensure the

continued provision of global ecosystem services such as carbon storage and biodiversity protection. The average incremental social cost of deforestation is a single number that provides relatively long-term guidance for the design of payments for ecosystem services. That number can be used by all Amazon countries, and it is not expected to change much over time, while the marginal social cost of deforestation faced by an individual country may change from year to year.

Future research can address the assumptions of this paper and the uncertainties related to forest resilience and the forest-savanna tipping point. Resilience is highly site- and scale-specific, and is difficult to quantify.¹³ It can depend on a number of ecosystem-wide variables, most importantly on precipitation volume, but also on environmental conditions such as climate and soil. The exact forest-savanna tipping point is unknown. Hirota et al. (2011) suggest that deforestation to the unstable threshold of 60% tree cover might induce a self-propagating shift to an open savanna over a range of rainfall levels. Nobre & Borma (2009) suggest that tipping points for the Amazon forest may exist for total deforested area greater than 40% and for temperature increase greater than 3-4 °C. We treated forest resilience as a function of the proportion of deforested area, and modeled the uncertainty over the threshold by a linear transformation of a beta distribution. (We ignored any possible effect of global warming on the threshold.) The impact of forest resilience on the economic value of a standing forest is also largely unknown. Additionally, the interdependence of land-use policies across countries (that results from the existence of a tipping point) leads to coordination problems that we largely ignored.

A final caveat: Our focus has been the appropriate method for calculating the social cost of deforestation. For that purpose, most of the data employed in the calculations were drawn from other studies, some of which related to other tropical forests besides or in addition to the Amazon rainforest. A more robust cost study would require a thorough assessment of the input data used in this model.

References

- Andersen, L. E. (2015), A cost-benefit analysis of deforestation in the Brazilian amazon, Working paper, IPEA.
- Andersen, L. E., Granger, C. W. J., Reis, E. J., Weinhold, D. & Wunder, S. (2002), *The Dynamics of Deforestation and Economic Growth in the Brazilian Amazon*, Cambridge

¹³For a recent research on the quantification of ecological resilience see, for example, Mitra et al. (2015).

University Press.

URL: <http://doi.org/10.1017/CBO9780511493454>

Bowman, D. M. J. S., Murphy, B. P. & Banfai, D. S. (2010), 'Has global environmental change caused monsoon rainforests to expand in the Australian monsoon tropics?', *Landscape Ecology* **25**(8), 1247–1260.

URL: <http://doi.org/10.1007/s10980-010-9496-8>

Coe, M. T., Costa, M. H. & Soares-Filho, B. S. (2009), 'The influence of historical and potential future deforestation on the stream flow of the Amazon river - Land surface processes and atmospheric feedbacks', *Journal of Hydrology* **269**(1-2), 165–174.

URL: <http://doi.org/10.1016/j.jhydrol.2009.02.043>

Davidson, E. A., De Araújo, A. C., Artaxo, P., Balch, J. K., Brown, I. F. C., Bustamante, M. M. & Wofsy, S. C. (2012), 'The Amazon basin in transition', *Nature* **481**(7381), 321–328.

URL: <http://doi.org/10.1038/nature10717>

De Groot, R., Brander, L., Van Der Ploeg, S., Costanza, R., Bernard, F., Braat, L. & Van Beukering, P. (2012), 'Global estimates of the value of ecosystems and their services in monetary units', *Ecosystem Services* **1**(1), 50–61.

URL: <http://doi.org/10.1016/j.ecoser.2012.07.005>

Fearnside, P. M. (1997), 'Environmental services as a strategy for sustainable development in rural Amazonia', *Ecological Economics* **20**(1), 53–70.

URL: [http://doi.org/10.1016/S0921-8009\(96\)00066-3](http://doi.org/10.1016/S0921-8009(96)00066-3)

Fearnside, P. M. (2003), A floresta Amazônica nas mudanças globais, Technical report, Instituto Nacional de Pesquisas da Amazônia (INPA), Manaus, Brazil.

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., Thau, D., Stehman, S. V., Goetz, S. J., Loveland, T. R., Kommareddy, A., Egorov, A., Chini, L., Justice, C. O. & Townshend, J. R. G. (2013), 'High-resolution global maps of 21st-century forest cover change', *Science* **342**(6160), 850–853.

URL: <http://science.sciencemag.org/content/342/6160/850>

Hirota, M., Holmgren, M., Van Nes, E. H. & Scheffer, M. (2011), 'Global resilience of tropical forest and savanna to critical transitions', *Science* **334**(6053), 232–235.

URL: <http://science.sciencemag.org/content/334/6053/232.abstract>

- INPE (2015), Program for the estimation of Amazon deforestation (Projeto PRODES Digital), Technical report, Instituto Nacional de Pesquisas Espaciais.
URL: Available at http://www.obt.inpe.br/prodes/prodes_1988_2015n.htm, Accessed 24 July 2016
- Lobo Sternberg, L. D. S. (2001), 'Savanna-forest hysteresis in the tropics', *Global Ecology and Biogeography* **10**(4), 369–378.
URL: <http://doi.org/10.1046/j.1466-822X.2001.00243.x>
- Malhi, Y., Roberts, J. T., Betts, R. A., Killeen, T. J., Li, W. & Nobre, C. A. (2007), 'Climate change, deforestation, and the fate of the Amazon', *Science* .
URL: <http://science.sciencemag.org/content/early/2007/11/29/science.1146961.abstract>
- Margulis, S. (2004), Causes of deforestation of the Brazilian amazon, Working paper 22, World Bank.
URL: <http://doi.org/10.1596/0-8213-5691-7>
- Mayle, F. E., Langstroth, R. P., Fisher, R. A. & Meir, P. (2007), 'Long-term forest-savannah dynamics in the Bolivian Amazon: Implications for conservation', *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences* **362**(1478), 291–307.
URL: <http://doi.org/10.1098/rstb.2006.1987>
- Mitra, C., Kurths, J. & Donner, R. V. (2015), 'An integrative quantifier of multistability in complex systems based on ecological resilience', *Scientific reports* **5**.
- Nepstad, D. C., Stickler, C. M., Filho, B. S. & Merry, F. (2008), 'Interactions among Amazon land use, forests and climate: Prospects for a near-term forest tipping point', *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences* **363**(1498), 1737–1746.
URL: <http://doi.org/10.1098/rstb.2007.0036>
- Nobre, C. A. & Borma, L. D. S. (2009), 'Tipping points for the Amazon forest', *Current Opinion in Environmental Sustainability* **1**(1), 28–36.
URL: <http://doi.org/10.1016/j.cosust.2009.07.003>
- Pearce, D. W. (1993), Economic values and the natural world, Technical report, London: Earthscan Publications Limited.
- Pindyck, R. S. (2016), The social cost of carbon revisited, Working paper 22807, NBER.

- Pires, G. F. & Costa, M. H. (2013), 'Deforestation causes different subregional effects on the Amazon bioclimatic equilibrium', *Geophysical Research Letters* **40**(14), 3618–3623.
URL: <http://doi.org/10.1002/grl.50570>
- Sampaio, G., Nobre, C., Costa, M. H., Satyamurty, P., Soares-Filho, B. S. & Cardoso, M. (2007), 'Regional climate change over eastern Amazonia caused by pasture and soybean cropland expansion', *Geophysical Research Letters* **34**(17), 1–7.
URL: <http://doi.org/10.1029/2007GL030612>
- Scheffer, M., Carpenter, S., Foley, J. A., Folke, C. & Walker, B. (2001), 'Catastrophic shifts in ecosystems', *Nature* **413**(6856), 591–596.
URL: <http://doi.org/10.1038/35098000>
- Soares-Filho, B. S., Nepstad, D. C., Curran, L. M., Cerqueira, G. C., Garcia, R. A., Ramos, C. A. & Schlesinger, P. (2006), 'Modelling conservation in the Amazon basin', *Nature* **440**(7083), 520–523.
URL: <http://doi.org/10.1038/nature04389>
- Staver, A. C., Archibald, S. & Levin, S. (2011), 'The global extent and determinants of savanna and forest as alternative biome states', *Science* **334**(6053), 230–232.
URL: <http://doi.org/10.1126/science.1210465>
- Staver, A. C., Archibald, S., Levin, S., Stayer, A. C., Archibald, S. & Levin, S. (2015), 'Tree cover in sub-Saharan Africa: Rainfall and fire constrain forest and savanna as alternative stable states', *Ecology* **92**(5), 1063–1072.
URL: <http://doi.org/10.1890/i0012-9658-92-5-1063>
- Stickler, C. M., Coeb, M. T., Costac, M. H., Nepstada, D. C., McGrath, D. G., Diasc, L. C. P., Rodrigues, H. O. & Soares-Filho, B. S. (2013), 'Dependence of hydropower energy generation on forests in the Amazon basin at local and regional scales', *PNAS* **110**(23), 9601–9606.
URL: <http://doi.org/10.1073/pnas.1215331110>
- Strand, J. (2017), 'Modeling the marginal value of rainforest losses: A dynamic value function approach', *Ecological Economics* **131**(C), 322–329.
URL: <http://dx.doi.org/10.1016/j.ecolecon.2016.09.019>

Torras, M. (2000), 'The total economic value of Amazonian deforestation, 1978-1993', *Ecological Economics* 33(2), 283–297.

URL: [http://doi.org/10.1016/S0921-8009\(99\)00149-4](http://doi.org/10.1016/S0921-8009(99)00149-4)

Warman, L. & Moles, A. T. (2009), 'Alternative stable states in Australia's wet tropics: A theoretical framework for the field data and a field-case for the theory', *Landscape Ecology* 24(1), 1–13.

URL: <http://doi.org/10.1007/s10980-008-9285-9>