Cattle, Clean Water, and Climate Change: Policy Choices for the Brazilian Agricultural Frontier

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In the Amazonian agricultural frontier, pasture for cattle ranching is an important and potentially hazardous form of land use because of sediment erosion as pastures degrade. This relationship between ranching, sediment load, and water quality is likely to further exacerbate environmental impacts, particularly in the context of climate change. We examine the role that river basin councils (RBCs) - a water governance option of Brazil's 1997 National Water Act - might play in managing this nonpoint-source pollution in the Amazônian state of Rondônia. We implement a simple coupled rancher-water system model to compare two potential governance options: a bulk water cleanup charge (BWC) implemented by RBCs and a land-use fine (LUF) for failing to maintain riparian buffers. We find no significant advantage of BWC over LUF in reducing sediment loading while keeping ranching profitable, under a changing climate. We also fail to find in Rondônia the important stake in water issues that has driven water reform elsewhere in Brazil. Moreover, the comparative success of reforestation programs suggests these programs may, in fact, have the potential to manage nonpoint-source agricultural pollution in the region.

Introduction

The Amazon region, or Amazônia, is commonly associated with abundant water; however, these resources are not uniformly distributed across space or time and are threatened by the joint stresses of agricultural/ranching development and climate change. In addition to slow changes in temperature and precipitation in the region, increases in the frequency of storms and droughts and increases in interannual and seasonal variability are expected (*1*). Coupled with uncertainty regarding climate expectations, these changes could enhance erosion and sedimentation and degrade water quality in agricultural landscapes.

Water stress and scarcity elsewhere in Brazil have driven water reform, punctuated by a new constitution in 1988 and a National Water Act in 1997, which together sought to reframe the idea of water as a resource with economic value (*2*). The new system creates structures for integrated governance of all water uses at the level of the hydrographic basin - river basin councils (RBCs) - that work in tandem with more traditional management such as municipal and state water and environmental agencies. These tripartite councils are composed of federal and state actors, water users and user groups, and representatives of organized civil society (*3*). Waters flowing entirely within the borders of a single state fall under state jurisdiction; those crossing state borders fall under federal jurisdiction, requiring greater federal representation on the basin councils (*3*). The RBCs have two central tools at their disposal to rationalize water use - *outorga* (water use permits) and *cobrança* (bulk water charges) with the revenues in principle to be reinvested in water projects within the basin (*4, 5*). To date, the water reform has created over 100 stakeholder-driven RBCs with mixed success (*6*); reform has advanced the furthest in the semiarid Northeast (*5*) and the highly industrialized South and Southeast (*2*). However, little has occurred in Amazônia where a single council has formed in the Tarumã-Açú River Basin in the state of Amazonas.

Despite this slow pace of progress, the exacerbation of water quality problems in Rondonia, especially under climate change, will soon push decision makers to either use available institutions or design new ones. In this study, we aim to inform this process by exploring, through modeling and institutional analysis, two potential policy choices. Our goals are 3-fold: 1) examine which policy is the best option to improve water quality; 2) inform the institutional design of RBCs by exploring whether pollution control instruments implemented in other parts of Brazil would work in Rondonia; and 3) inform policy makers of which options are likely to fare better under the threat of uncertain but expected climate stressors.

We use a simple model of a ranching property to compare the two policy options. The first is a bulk water charge (BWC) tool that incentivizes efforts to curb pollution by creating a price scale that punishes polluters and rewards cleaner forms of water use (*4*). The second is based on other environmental approaches in Amazônia that penalize farmers by levying a land-use fine (LUF) for failing to maintain adequate riparian buffer around watercourses.

The model is informed by conditions prevalent in Rondônia where the landscape is dominated by ranchland for cattle, making sedimentation and associated declines in water quality a potentially major future regional-scale water issue (*7, 8*). We use sediment load as a proxy for pollution because it is a key vector in the transport of nutrients, organic carbon, and other contaminants. Also importantly, the effects of sediment loading extend beyond the local area to have regional scale impacts (*7, 8*). However, most ranchers - the agents of land-use change - get water for domestic use from wells and have no stake in the condition of surface waters (*9*). In this sense, the landscape mirrors cases in other areas of the world where nonpoint source pollution has been of lower concern in deliberative water management processes (*10*). The most advanced work toward formation of a RBC in Rondônia has been a set of studies in the municipality of Ouro Preto d'Oeste (*11*), a small catchment (18,000ha) whose urban water demand is fed by the agriculturally developed Boa Vista River Basin; clearly, stake in water resource management is localized and isolated.

Compared to other regions of Brazil where the citizenry has been mobilized to initiate water reform, agriculture in Rondônia is less intensive, less dense, and less mechanized (*12*). This highlights the need for understanding the effect of public policy on livelihoods as well as impacts on water quality because much of the environmental degradation in Amazônia is related to the inability of poorer farmers to maintain their land's productivity (*13, 14*). In this context, policies that overly burden farmers in the interest of improving environmental quality may end up exacerbating environmental impacts.

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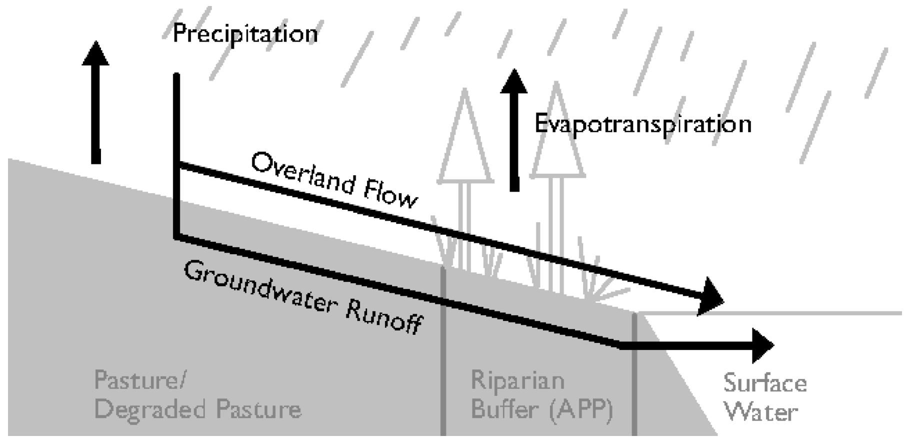

FIGURE 1. Ranch plot and model hydrology.

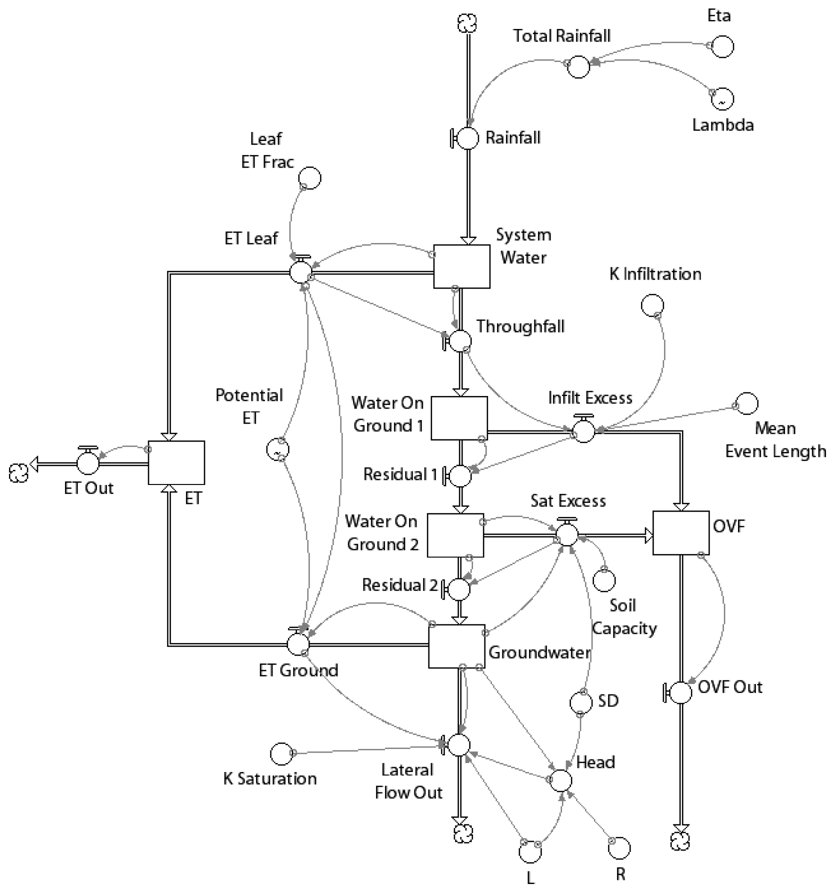

FIGURE 2. SDM implementation of hydrology.

Moreover, in the wake of projected climate change - and its potential negative impacts on livelihoods in the region, it is critical to better understand the role of alternative policy choices to achieve both environmental and socioeconomic goals.

Modeling Water Policy in Rural Landscapes

A causal loop diagram illustrating the assumptions of the ranching system upon which this research is based is included as Appendix A.

Modeling Hydrology. The hydrological submodel is nonspatial and excludes a base flow/shallow aquifer recharge component (Figures 1 and 2), whose primary role in the model would be to provide a constant diluting factor along the annual cycle (which would not affect the comparative analysis); thus, this component is excluded for simplicity.

Equations for the state variables (stocks - rectangular boxes in Figure 2) in this submodel are not defined explicitly in the STELLA platform but can be inferred by the sum of flows entering and leaving them. Equations for the flows are given by

$$ET_{Leaf} = \min(\text{Leaf ET Frac} \cdot \text{System Water}, ET_{potential}) \quad (1)$$

$$ET_{Ground} = \min(\text{Groundwater}, ET_{potential} - ET_{Leaf}) \quad (2)$$

$$\text{Throughfall} = \text{System Water} - ET_{Leaf} \quad (3)$$

$$\text{Infilt Excess} = \max(0, \text{Throughfall} - \text{Mean Event length} \cdot K_{\text{Infiltration}}) \quad (4)$$

$$\text{Residual 1} = \text{Water on Ground 1} - \text{Infilt Excess} \quad (5)$$

$$\text{Sat Excess} = \max(0, \text{Water on Ground 2} - (\text{Soil Capacity} \cdot \text{SD} - \text{Groundwater})) \quad (6)$$

$$\text{Residual 2} = \text{Water on Ground 2} - \text{Sat Excess} \quad (7)$$

$$\text{Lateral Flow Out} = \min\left(\frac{\text{Head}}{L} \cdot K_{\text{Saturation}}, \text{Groundwater} - \text{ET}_{\text{Ground}}\right) \quad (8)$$

$$\text{Head} = \frac{R}{\sqrt{R^2 + L^2}} \cdot \frac{\frac{\text{SD} \cdot L}{2 \cdot R} - \text{SD} + \sqrt{\text{SD}^2 + 2 \cdot L \cdot \left(\frac{L}{R} + \frac{R}{L}\right) \cdot \text{Groundwater}}}{\frac{L}{R} + \frac{R}{L}} \quad (9)$$

where ET_{Leaf} and $\text{ET}_{\text{Ground}}$ are evapotranspiration from water on above-ground biomass and from water drawn from the ground, respectively; $K_{\text{Infiltration}}$ and $K_{\text{Saturation}}$ are the rates of infiltration excess flow and saturation excess flow, respectively; SD is the soil depth; and L and R are the length and rise of the property. Residuals 1 and 2 denote the residual water after accounting for infiltration and saturation excess flow, respectively. The quantities ET_{Out} and OVF_{Out} denote the total evapotranspiration and overland flow leaving the system.

Potential evapotranspiration ($\text{ET}_{\text{potential}}$) is defined for both pasture and forest to be around 4 mm/day, though pasture evapotranspiration dips significantly during the dry season (Appendix B). The shape of this annual pattern is based on modeled and measured results in ref *15*.

Modeling Climate. In the model, daily precipitation is drawn from the distribution

$$\text{Total Rainfall} = [X \sim \exp(\lambda(1 - \eta) + X \sim \exp(\eta\lambda))] \cdot (1 + \nu) \quad (10)$$

Here, $X \sim \exp(a)$ denotes an exponential distribution with mean a, commonly applied to model precipitation (*16, 17*) as they have the property of being highly skewed toward 0 as in real distributions of precipitation. Integer values for λ for each month preserve rainy-dry season structure (Table 1) and an average rainfall for the region of about 1800−2200 mm per year (*18*). The parameter η scales the variance of λ to increase the frequency of extreme precipitation events while maintaining the same annual precipitation. The parameter ν scales the overall precipitation without affecting distribution. Thus, the modified three-parameter (λ,η,ν) model allows us to capture several of the major anticipated impacts of climate change in the region - increases in wet and dry season precipitation and increases in the frequency of droughts and extreme storms - in a simple and straightforward way. Illustrations of the distributions generated by this model as well as monthly values for λ are included as Appendix B.

TABLE 1. Policy Parameters in Experimental Design

	parameter values
S_{BW} ($/m^3)	0, 0.04, 0.08, 0.12, 0.16, 0.2, 0.24
L_{R} (t/m$_3$)	0.0001, 0.00005, 0.000001, 0.000005, 0.000001, 0.0000005, 0.0000001
S_{LU} ($/ha)	0, 100, 200, 1000, 1500, 2000, 3000, 4500, 6000, 7500, 9000
W_{R} (m)	0, 30, 60, 90, 120, 150, 180

Modeling Erosion. The erosion model used here is simple. Soil erodes from pasture and degraded pasture at rates proportional to the amount of overland flow and is retained by the buffer at a rate proportional to the width of the buffer

$$\begin{aligned} e_{\text{net}} &= e_{\text{total}} - e_{\text{atten,Buffer}} \\ &= e_{\text{total}} - \min(e_{\text{total}}, e^{\text{potential}}_{\text{atten,APP}}) \end{aligned} \quad (11)$$

where

$$\begin{aligned} e_{\text{total}} &= \left(\frac{H_{\text{OVL}}}{H_{\text{OVL},0}}\right)^c \cdot (e_{0,\text{past}} \cdot A_{\text{past}} + e_{0,\text{dpast}} \cdot A_{\text{dpast}}) \\ e^{\text{potential}}_{\text{atten,APP}} &= e_{0,\text{atten,Buffer}} \cdot \left(\frac{A_{\text{Buffer}}}{A_{0,\text{Buffer}}}\right)^a \cdot \left(\frac{H_{\text{OVL},0}}{H_{\text{OVL}}}\right)^b \end{aligned} \quad (12)$$

Here, H_{OVL} is the overland flow in mm and $H_{\text{OVL},0}$ is the nominal overland flow that under normal ($\eta = 0$) climate conditions would lead to annual average erosion values equal to the defined nominal erosion values $e_{0,\text{past}}$ and $e_{0,\text{dpast}}$ per unit area; $A_{0,\text{Buffer}}$ is the width of buffer for which the nominal erosion attenuation $e_{0,\text{atten,Buffer}}$ is defined. The exponents a, b, and c allow the relationship between erosion or sediment trapping and buffer depth or overland flow to be nonlinear or linear; the definition of these exponents is discussed in the Experiments section.

Modeling Ranching. The ranch in this model is a 1-dimensional plot, with three land types - pasture, degraded pasture, and forest buffer (Figure 1). The rancher is a profit maximizer, with two decisions to make in each period - whether and how to stock cattle and how to change land use. All cattle on the land that reach three years of age are slaughtered, and this is the sole source of revenue for the rancher. Ranching costs include diet supplements during drought periods when grass growth is not sufficient to support cattle growth, costs for land-use changes, and costs incurred through sanctions.

Cattle Stocking. The rancher decides whether to stock the land with cattle based on the present value (PV) of beef (over a 3-year cattle lifetime) on a mass basis

$$\text{PV}_{\text{beef}} = \frac{p - \frac{c_{\text{annual}}}{T}}{(1 + d)^2} - \frac{\frac{c_{\text{annual}}}{T}}{(1 + d)} - \frac{c_{\text{annual}}}{T} \quad (13)$$

where p is the market price for beef, c_{annual} is the total annual cost for the ranching operation, d is the discount rate, and T is the current cattle stock in kg. When this value is positive, the rancher performs a simple estimate of the rate at which the land can be stocked

$$\text{d}T = f \cdot \frac{k_{\text{grass,max,obs}} \cdot A_{\text{Past}}}{U_{\text{daily,kg}} \cdot (W_{\text{calf}} + W_{\text{2year}} + W_{\text{3year}})} \cdot \text{d}t \quad (14)$$

where W_{calf}, $W_{\text{2 year}}$, and $W_{\text{3 year}}$ are the weights of calves, 2-year, and 3-year old heads of cattle; $k_{\text{grass,max,obs}}$ is the maximum observed grass growth rate per hectare; $U_{\text{daily,kg}}$ are the daily nutrient requirements of cattle per kg of body weight; A_{Past} is the area of pasture; dt is the cattle stocking interval; and f is a unitless scalar term. The scalar f is an important part of the rancher decision-making process, providing a means both to correct for imperfections in this stocking rate estimate and to distinguish behaviors among ranchers. A higher f implies riskier behavior with respect to exceeding the production capacity of the land; a lower f implies more conservative, risk-averse behavior.

Grass grows according to simple logistic growth, scaled by the availability of water

$$ds_{grass} = k_{grass,max} \cdot \frac{s_{grass}}{S_{grass}} \cdot \left(1 - \frac{s_{grass}}{S_{grass}}\right) \cdot \left(\frac{ET_{out}}{ET_{potential}}\right) dt \quad (15)$$

where s_{grass} and S_{grass} are the grass stock and grass capacity per hectare, respectively, and $k_{grass,max}$ is the intrinsic growth rate. Grass is grazed by cattle at a rate proportional to their body weight, $U_{daily,kg}$, except when grass stocks are low (below half-capacity in these experiments) when they are fed diet supplements.

Land-Use Change. At each decision interval the rancher is able to change up to a_{change} hectares of land by 1) restoring degraded pasture to pasture, 2) restoring pasture to forest, or 3) clearing new forest. The rancher looks at the present value (PV) per hectare of each of these decisions and undertakes them in order of decreasing PV until a_{change} ha have been changed or there is no further change that would yield a positive PV, where

$$PV_{r,dp}(\$/ha) = -C_{r,dp} + PV(I_{e,p}) - PV(C_{e,n}) + PV(C^{dp}_{e,wc}) \quad (16)$$

$$PV_{r,fb}(\$/ha) = -C_{r,fb} - PV(I_{e,p}) + PV(C_{e,n}) + PV(C^{p}_{e,wc}) + PV(C_{e,fb}) \quad (17)$$

$$PV_{c,fb}(\$/ha) = -C_{c,fb} + PV(I_{e,p}) - PV(C_{e,n}) - PV(C^{p}_{e,wc}) - PV(C_{e,fb}) \quad (18)$$

C indicates cost, and I indicates income. The subscripts e, r, and c denote expected, restore, and clear, respectively. The sub- and superscripts p, dp, fb, n, and wc denote pasture, degraded pasture, forest buffer, nutrient, and water charge, respectively. The individual present value terms are given by

$$PV(I_{e,p}) = \sum_{i}^{n} \frac{w_{3year} \cdot S_{cattle} \cdot p}{(1+d)^i} \quad (19)$$

$$PV(C_{e,n}) = \sum_{i}^{n} \frac{S_{cattle} \cdot c_{nutrients}}{(1+d)^i} \quad (20)$$

$$PV(C^{p}_{e,wc}) = \sum_{i}^{n} \frac{q_{eros} \cdot e_{past}}{(1+d)^i} \quad (21)$$

$$PV(C^{dp}_{e,wc}) = \sum_{i}^{n} \frac{q_{eros} \cdot (e_{dpast} - e_{past})}{(1+d)^i} \quad (22)$$

$$PV(C_{e,fb}) = \begin{cases} \sum_{i}^{n} \frac{q_{fb}}{(1+d)^i} \\ 0 (A_{fb} > A_{fb,req}) \end{cases} \quad (23)$$

where p is the unit price for beef, S_{cattle} is the pasture capacity, and $w_{3\ year}$ is the weight fraction of 3-year old cattle in the stock; q_{eros} is the average sanction (collected per unit contaminated water) that accrues per unit erosion, and q_{Buffer} is the unit sanction collected per hectare of land below the required $A_{Buffer,\ req}$; e_{past} and e_{dpast} are the observed erosion loads from pasture and degraded pasture, respectively; $c_{nutrients}$ is the average annual cost of nutrient supplement; and d is the discount rate. The number of periods n for the PV calculation is the expected lifetime of the pasture before it degrades, which is a function of the rate at which it is grazed

$$n = \left(\frac{G_0}{G} \cdot n_0\right) \quad (24)$$

where G is the grazing rate, and G_0 is the nominal grazing rate for which the nominal pasture lifetime n_0 is defined (Appendix C).

This model does not evaluate processes at a landscape scale. The rancher has access only to the plot of land in the model and cannot expand his holdings. Sills et al. (2008) found that in Ouro Preto do Oeste, a typical established ranching region in Rondônia, relatively secure land tenure and low rates of absentee ownership suggest that land speculation is not important in the area (*19*); therefore, it is not included as a mechanism in this model.

Policy Scenarios. *1. Bulk Water Charges (BWC).* One instrument of the Brazilian water reform laid out in the National Water Act is the BWC levied against consumptive water use, including extraction and pollution (*3*), where large users are charged for contaminated water cleanup. Here, we examine a BWC levied against rural nonpoint source polluters for contamination of river water by sediment. While, from a monitoring and implementation perspective, charges for nonpoint source polluters might be challenging and costly (*20*), they are not impossible (*21*). In addition, the trend of land aggregation along the frontier (*22*) will reduce the difficulty in attribution. As well, use of BWC in conjunction with water quality criteria laid out by the National Environment Council CONAMA (*23*) are an example of Brazilian enforcement institutions tied directly to water quality. Implementation of BWC in Brazil has been challenging, with significant resistance both from users (who have generally had water for free) and some state agencies (who are unwilling to cede control over water resources) (*24, 25*). A small number of basins have successfully negotiated BWC, such as in the Jaguaribe Basin in Ceará (*25*) and the federal basin of the Paraiba do Sul River, whose BWC formula allows collection for consumptive, withdrawal, and effluent dilutive uses of water resources (*4*).

2. Land Use Fines (LUF). The 1965 Brazilian Forest Code mandates maintenance of riparian buffer zones along watercourses on rural properties, with required widths dependent on the width of the river. For example, watercourses up to 10 m across must be bounded by 30 m of riparian vegetation (*26*). Current attempts to enforce this rule in Rondônia include a new environmental licensing scheme, under which producers provide a management plan showing progress toward compliance with the Forest Code to access rural credit programs (*27*). Noncompliance results in farmers fined for not maintaining adequate riparian vegetation (*28*). Results from licensing schemes elsewhere in Amazônia have been mixed (*29*), attributed mostly to poor enforcement by environmental agencies that lack resources and capacity (*30*). As a proxy for enforcement (e.g., maintenance of forest cover), in this paper we introduce a land-use fine (LUF) levied against each unit area of deficient riparian forest below the regulation width, charged once yearly.

Model Experiments. Each policy option is defined along two dimensions. For the LUF, these dimensions are required buffer width (W_R) and the monetary sanction per unit deficient width (S_{LU}). For BWC, the dimensions are the threshold sediment load (L_R), and the monetary sanction per unit volume to clean the water (S_{BW}). A factorial experiment (Table 1) was carried out for each policy option along each of its two dimensions, as well as for the climate scenarios. Response surfaces were generated for the total revenues obtained, total costs incurred, and total sediment eroded over the period of the policy. Where possible, values for model parameters have been drawn from the literature

TABLE 2. Ranges for Monte Carlo Analysis

variable	range (uniform distribution)
soil depth, SD (m)	0.5−1
nominal overland flow, $H_{OVF,0}$ (mm)	10−20
nominal buffer width, $A_{Buffer,0}$ (m)	20−40
exponent for trap efficiency - overland flow	0.75−1.25
exponent for trap efficiency - buffer width	0.75−1
exponent for erosion - overland flow	0.75−1.25

(Appendix C); other values were selected to simulate a ranch supporting 1.5−2 head of cattle per hectare.

There are several physical relationships in the model that have strong influences on the role each policy will play - the relationships between precipitation and overland flow, between overland flow and erosion and sediment trapping by buffers, and between buffer width and sediment trapping. A detailed calibration along these physical dimensions is not possible with this low-fidelity modeling approach, nor can it represent the strongly heterogeneous relationships across a landscape. Therefore, each experimental run was a Monte Carlo simulation with parameters governing the above relationships (from eqs 6, 9, and 12) drawn from the uniform distributions described in Table 2. The results can be thought of as analogous to simulations across a rugged landscape of physical relationships between water, soil, vegetation, and erosion.

Results

Our experimental design generated response surfaces across S_{BW} and L_R in the case of BWC and across S_{LU} and W_R in the case of LUF. A set of runs for each of the BWC and LUF cases is shown in Appendix D along with complete response surfaces showing outcomes across both unit sanction and sanction threshold.

We focus on a set of two representative 'slices' across these surfaces. For BWC, we look first along a curve of decreasing pollutant threshold (L_R) at constant unit water charge (S_{BW}) and then along a curve of increasing S_{BW} with constant L_R. For LUF, we look at a curve generated by increasing the required buffer width (W_R) at constant unit area fine (S_{LU}) and then a curve of increasing S_{LU} with constant W_R. To compare results, we present results in 'sanction response curves' (described in Appendix E) where sediment loading and profitability are plotted against each other. Increasing sanction strictness follows the curve from right to left. Therefore the slope of the curve is analogous to a measure of cost effectiveness - the steeper the curve, the greater improvement in water quality per unit economic burden on the rancher.

These sanction response curves typically contain up to three distinct regions: an initial flat 'weak' region, in which increasing sanction costs have not yet encouraged the rancher to change behavior; a declining 'effective' region where increments in sanction strength induce responses by the rancher and lead to better environmental outcomes; and finally a flat 'burden' region where the rancher is unable to make further improvements to environmental outcomes.

The responses to BWC and LUF policies share a number of features. First, making requirements stricter (requiring cleaner water, or wider buffers moving from right to left in Figure 3A,C) under a stiff unit sanction brings immediate results from both policies. In contrast, when the rancher is given a strict requirement to meet, his behavior does not begin to shift until the unit sanction increases to a point that, from his rational decision making point and level of information, warrants it - a 'weak' region preceding the 'effective' region (Figure 3B,D). In all cases, the curves flatten into a 'burden' region when the rancher is no longer able to make improvements in environmental performance, and increasingly stringent requirements or sanctions simply cost the rancher more money. Both policies respond similarly to climate change - increases in climate variability and in overall precipitation lead to decreased environmental performance and greater costs to the rancher (shifts up and to the left in Figure 3).

However, there are several important differences between BWC and LUF. First, across all climate scenarios, BWC appears to achieve better environmental outcomes. This is due in part to the push by the BWC not only to maintain riparian cover but also to reduce exposed soils by restoring degraded pasture; LUF, in contrast, does not make it more or less profitable to restore degraded pasture. Second, these environmental gains under BWC come at much greater cost to the rancher. Much of the improvement in sediment loading occurs after profits for the rancher have dropped below 0. In our simple model, the rancher is able to continue to operate at a loss and make the best of a bad situation, leading to these eventual improvements in water quality. However, it is more likely that onerous sanctions such as this would force the rancher off the landscape or, perhaps more likely given the low capacity for monitoring and enforcement in Amazônia (*30, 31*), would simply be ignored. The reason for these incredible costs to the rancher in the model is that, where climate is uncertain and where the relationship between riparian cover and sediment loading is not perfectly known, it is difficult to make a good decision about how much riparian cover to maintain to avoid incurring a BWC. In contrast, when the required width of the buffer is specified, as in the LUF, decision-making is much simpler for the rancher, reflected by the comparatively minor drops in profitability that the rancher incurs under LUF.

Looking only at improvements to water quality where profits remain positive (and where we could more reasonably expect some degree of compliance), BWC no longer appears to outperform LUF. Also, from a 'cost-effectiveness' perspective (the slopes of the curves in this range), LUF brings about greater improvement to water quality per unit drop in rancher profitability.

Discussion

Model Limitations. This simple coupled human-natural ranching environment model necessarily abstracts many aspects of the real system. For example, ranching decisions are likely to be more sophisticated and more continuously spread over time than represented in the model. Hydrology will vary within and across farm plots, leading to great variation in erosion potential across the landscape (*32, 33*). The sediment trapping efficiency of riparian buffers may be modeled as a more detailed function of overland flow rate and buffer width (*34*) and of vegetation, and agricultural performance is subject to a number of natural and environmental factors not addressed by this model. Finally, differences among ranchers' decision-making as well as their cost structures and land endowments will lead to quantitatively different outcomes at the landscape scale than predicted by our model at the ranch scale.

However, our purpose was not to simulate detailed ranching operations and predict specific economic and environmental performance. Rather, it was to explore reasonable scenarios of ranching operations and look for patterns that offer insight into the combined impacts of climate and policy. Construction of such coupled models to better understand impacts and response to climate change has been identified as a critical priority of climate change research in the U.S. and abroad (*35*).

Implementing BWC and LUF in Practice. The implication of our results is that, from the perspective of the tools available

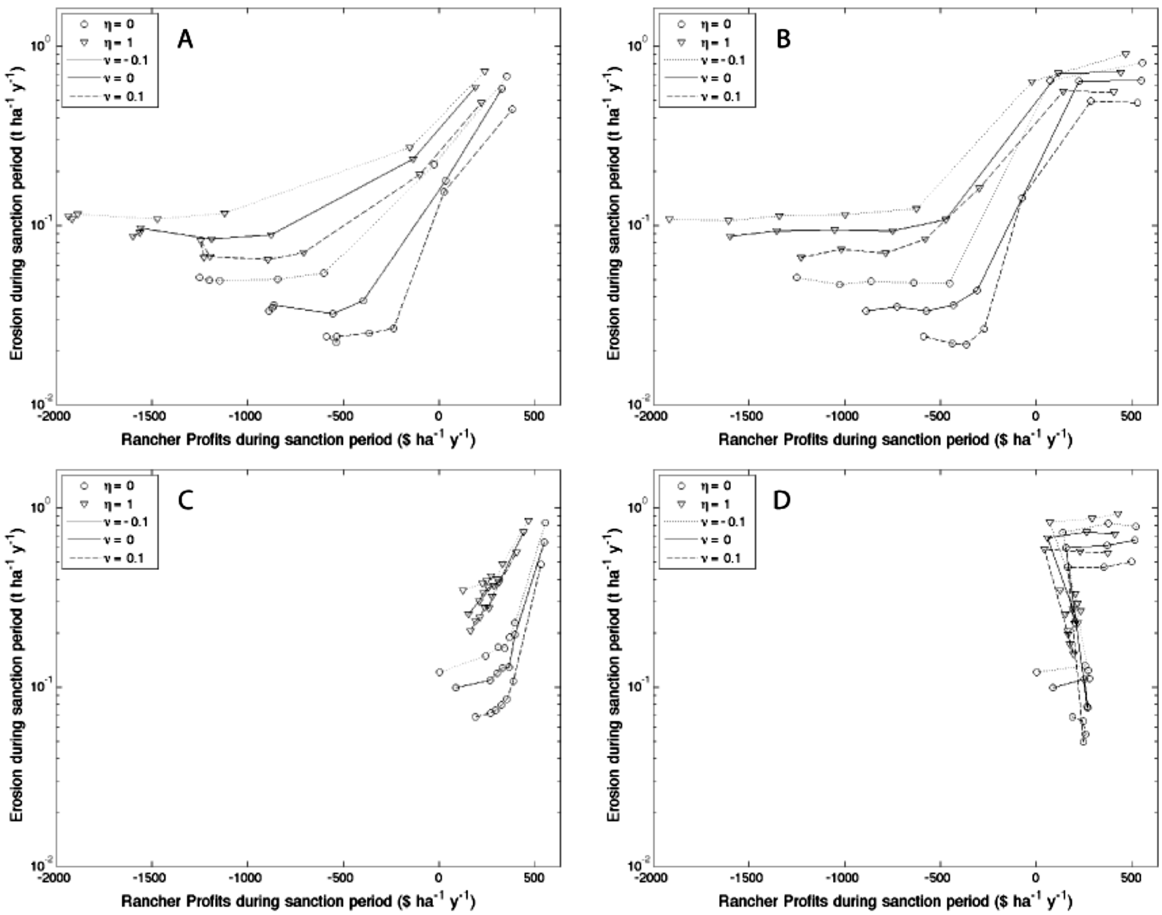

FIGURE 3. Sanction response curves over climate scenarios across A) constant unit sanction S_{BW} = $0.24/m^3 and B) constant threshold of $L_R = 10^{-6}$ t/m^3 for BWC, and C) constant unit sanction of S_{LU} = $9000/ha and D) constant target buffer width of W_R = 180 m for LUF. Curves associated with low climate variability are marked with circles; those with high climate variability are marked with triangles. Increased precipitation is marked by a dashed line; decreased precipitation is marked by a dotted line.

to basin councils as part of the water reform, BWC does not offer any particular advantage over approaches tied directly to land use (LUF), assuming rational behavior and perfect compliance. A standard for land use is much easier to understand and thus comply with than is a standard of water quality. This should be even truer in a drier, less certain climate, where droughts and extreme precipitation events make erosion more severe and difficult to plan for, and this is borne out in our model results (Figure 3). But how might the two approaches compare where compliance is less than perfect? That is, what is the comparative ability of RBCs (who would implement the BWC) and environmental agencies (who have begun implementing incentive and regulations programs tied to land use, like the LUF) to be effective?

One part of the answer lies in how aware land users are of regulation. Our own research in Rondônia demonstrated relatively low awareness of policies relating to water or water quality but relatively widespread understanding of the requirement to use buffers on rural properties (*9*), suggesting a stronger understanding of LUF practices over those for BWC. A second part of the answer lies in that while both approaches have similar requirements for enforcement (in the form of site visits with the land owner), monitoring requirements differ considerably. Brazil is relatively well-positioned to perform near-real-time satellite monitoring of land use and deforestation (*36*), but cost-effective monitoring and attributing nonpoint-source pollution remain elusive in agricultural landscapes worldwide (*20, 37*–*39*). Attempts to levy BWC against agricultural users elsewhere in Brazil have met with much resistance. Farmers in the Paraiba do Sul Basin agreed to pay only symbolic levels (*4*), while in Ceará, farmers have refused to pay - only large, visible users such as electric utilities have agreed to participate (*25*).

Finally, the likelihood of either approach to be put into legislation and implemented must be compared. For RBCs, evidence from other regions in Brazil suggests several important drivers of successful water reform. First, having a strong stake in how water is allocated is critical in mobilizing participation in water governance. In the state of Ceará in Northeast Brazil for example, perennial drought has driven allocation of scarce water resources among a diverse set of user stakeholders - irrigated agriculture, industry, and the metropolitan area of the state capital in Fortaleza (*5*). In the Southeast of Brazil, drought is more the exception than the rule, but dense urban populations, intensive agriculture, and powerful industries make access to water of sufficient quality an issue. Urban demand in the Alto Tietê basin (which supplies the metropolitan area of São Paulo) already outstrips supply, and with access to water resources at stake, a broad stakeholder base has been mobilized to support and participate in integrated water management (*24*). Whether driven by aridity or by user density, water scarcity creates incentive for all users to join the deliberative process and ensure that they have access to their share of water resources or that the resources are of sufficient quality for them to use.

Where water quality issues have made it onto the agenda of basin councils, they have focused largely on larger point sources, such as in the Alto Tietê Basin where bulk water charges cover effluents such as BOD and COD, inorganic effluents, and sediment residues, but are limited to waters for which users have been issued permits (*24*); nonpoint-source pollution from agricultural landscapes has not been a driver of the reform process in other areas in Brazil.

The successful cases in the Northeast and Southeast of Brazil involved groups of technical experts working over an extended period (since as early as the 1970s) who capitalized on the process of democratization in Brazil in the 1980s to advance water reform (*5*). For Paraibá do Sul, Formiga-Johnsson et al. (2007) drew on Kingdon's policy streams model to explain the progress of water reform in Brazil (*4*): particular focusing events (drought and water conflict) combined with political moods or climates (the 1980s shift to democracy) to create 'policy windows' through which policy entrepreneurs (the technical experts in Ceará and Alto Tietê) could advance an agenda (*40*). The lack of similar focusing events in Rondônia, the lack of large point-source water users, and the lack of stake in water allocation do not preclude water reform from being driven by some other actor or issue but do lead us to suggest that implementation of RBCs in Rondônia may be slow.

Looking now to LUF, evidence is mixed but suggests that there is capacity for different incentives and regulations to rationalize land use in Amazônia (*30, 41*). The Proambiente project has established a pilot ecosystem services payment program involving about 4000 households in 12 different regions throughout Amazônia (*30*), and several Amazônian states including Rondônia have begun to implement environmental licensing schemes for rural properties in which farmers and ranchers must demonstrate plans to bring their land use in alignment with the Brazilian Forest Code to gain access to rural credit (*27*). While these programs are distinct both from each other and from the stylized fine for land use implemented in our model, they share common goals and mechanisms - farmers are rewarded, or fail to be punished, by restoring and maintaining riparian cover along watercourses, with the underlying goal of preserving ecosystems services.

Thus far, these projects have only been partially implemented and have met the common barriers of a lack of implementation and monitoring capacity in Amazônia (*30, 42*). However, a focus on incentives tied to land-use practices to improve water quality has precedent elsewhere. In the United States, the difficulty in measuring or attributing nonpoint source pollutants, the expense associated with nonpoint source pollutant control, and the resistance generated by nonpoint source polluters to regulation have led the Federal government and many states to adopt voluntary best management programs (BMPs) to encourage land use that preserves water quality (*37*). The jury is still out as to what factors, in the US, lead to BMP adoption, but these programs have had some success in rationalizing land use (*43*). Success in Amazônia could point toward a generalization of the BMP approach across different institutional and cultural contexts.

The results of our simple model suggested that policy tied to land use rather than pollutant loads could more effectively manage rural water quality; the predictable sanctions from the LUF could be followed more easily and at lower cost by the rancher than the BWC. This result was even clearer under drier and less predictable climates. Institutionally, evidence is building that policies like the LUF are a better fit for the Amazonian context - examples of water quality issues driving the formation of basin councils are yet absent in Brazil, while encouraging results from projects regulating land use are already emerging in Amazônia. Tying together these threads, we find that the inherent measurability of land-use criteria (through near-real-time satellite monitoring, for example) coupled with a simpler set of objectives for farmers to follow (a width of riparian buffer versus a water quality outcome) and greater mobilization around the issue of deforestation and environmental services suggest that the future of water quality in Amazônia may be better served through these innovations in environmental legislation.

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Supporting Information Available

Detailed information on the causal and climate models, the literature values used for parametrization, additional model results, and an explanation of the sanction response curve are included as Appendices A-E. This material is available free of charge via the Internet at http://pubs.acs.org.

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