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Ecosystem Service Value and Agricultural Conversion in the Amazon: Implications for Policy Intervention

Michael L. Mann · Robert K. Kaufmann · Dana Marie Bauer · Sucharita Gopal · James G. Baldwin · Maria Del Carmen Vera-Diaz

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Abstract We explore the welfare implications of agricultural expansion in the Brazilian Amazon by comparing spatially explicit estimates of soybean rents and the value of ecosystem services. Although these estimates are generated from different datasets, models, and estimation techniques, the values are comparable, such that the value of ecosystem services is greater than soybean rents for about 61 % of the total area and 24 % of the area where soybean rents are positive if protected areas are well enforced. Based on the balance between the benefits and costs of conversion, failure to value ecosystem services reduces total social welfare by 7.13 billion dollars annually relative to an optimum. Policy instruments that internalize the value of ecosystem services via protected lands, land conversion taxes, conservation subsidies, or excise taxes can avoid much of this loss. Regardless of intervention regime, policy makers should be cognizant of the diminishing net benefits of converting natural ecosystems to agriculture. Realizing the final 3.8 % of total social welfare requires the conversion of an additional 15 % of natural ecosystems to soybean production.

Keywords Ecosystem services · Agriculture · Land use policy · Tax policy

M. L. Mann (✉) · R. K. Kaufmann · D. M. Bauer
Center for Energy and Environmental Studies, Boston University, 675 Commonwealth Ave.,
Boston, MA 02215, USA
e-mail: mmann1123@gmail.com

M. L. Mann · R. K. Kaufmann · D. M. Bauer · S. Gopal · J. G. Baldwin
Department of Geography and Environment, Boston University, 675 Commonwealth Ave.,
Boston, MA 02215, USA

M. L. Mann
Department of Environmental Science, Policy and Management, University of California-Berkeley,
130 Mulford Hall #335, Berkeley, CA 94720, USA

M. Del Carmen Vera-Diaz
GDAE, Tufts University, 44 Teele Avenue, Medford, MA, 02155, USA

1 Introduction

The Amazon basin provides environmental services at several spatial scales. The broader basin covers approximately 5.4 million km^2, and supports one of the world's greatest assemblages of terrestrial and aquatic biological diversity (Dirzo and Raven 2003). It is also a major engine of global atmospheric circulation and accounts for 15–20 % of global river flows (Chagnon 2005; Malhi et al. 2008). It is not easy to overstate this area's importance considering that the Amazon as a whole constitutes forty percent of all remaining tropical rainforest (Rodrigues et al. 2009).

Ecosystem services can be classified among four general types (MEA 2005). Supporting services include nutrient cycling and primary productivity, provisioning services include food, fiber, fuel and water, regulating services include climate regulation and flood control, and cultural services include recreational, spiritual and aesthetic uses. Each service works at one or more geographic scales. At the local level, ecosystem services support the production of cattle, soy and timber, which provide livelihoods for agrarian workers and constitutes six percent of Brazil's GDP (twenty five percent including agribusiness) (CRS 2006; Fearnside 1989). The basin also provides a number of other services such as recreational opportunities, transportation via river ways, and access to freshwater. At the regional scale, the basin provides several services such as hydroelectric power, evapotranspiration, which provides approximately 35 % of rainfall in the basin and strongly influences precipitation throughout South America, pollination, wildlife habitat, fire regulation, and flood control (Eltahir and Bras 1996; Foley et al. 2007). At the global scale, the Amazon's evaporation and condensation drive global atmospheric circulation, while vegetation stores approximately 120 Pg C with an annual uptake of nearly 0.6 Pg C per year (Malhi et al. 2008). The Amazon basin also hosts some of the largest stores of genetic material (Malhi et al. 2008).

Despite the clear benefits of ecosystem services, the 'invisible hand' of the market will not guide the efficient allocation of these resources. This is because the majority of benefits derived from ecosystem services are public goods, which has profound implications for both their conservation and use. By definition, the existence of a public good creates a 'market failure', or the inefficient allocation of a resource in an economy due to their non-rival and non-exclusive characteristics (Cole 2002). The public goods provided by the Amazon basin are implicitly assigned a value of zero and therefore extracted and/or destroyed at an inefficient rate. By definition, markets are unable to provide meaningful prices for these services because they cannot be traded easily in markets, therefore government intervention or other protective measures are required.

Some services such as timber may carry market prices, but services such as flood control and access to genetic diversity remain unpriced. In order to assign prices the value of these services must be discovered. The economic value of an ecological service is related to the contribution it makes to human welfare (Bockstael et al. 2000). An individual's welfare is a highly personal conception but perceptions of wellbeing or changes in it can be used to assess the value (benefit or costs) of any change. Economists use compensation tests to place a dollar value on changes by asking participants for the compensation that would be needed (positive or negative) to make them as 'well off' with or without the change. The change in this case is the loss of certain ecosystem services. The ability to price ecosystem services using compensation tests is limited because individuals do not understand large-scale changes and/or the complexity of environmental systems. As such, available research provides a geographically limited set of valuation studies for a disconnected subset of ecosystem services (Bockstael et al. 2000).

To overcome these limitations, some analysts quantify the macroeconomic contributions of ecosystem services. Following this approach, ecosystems are valued by their marginal contribution to value added or gross domestic product (Richmond et al. 2007), in which the aggregate level of 'ecosystem services' available is measured by the total level of net primary productivity (NPP). NPP is the amount of energy available to autotrophs after the costs of maintenance respiration are subtracted from gross primary production, which is defined as the total amount of inorganic energy that is converted to organic energy. NPP represents the total amount of chemical energy available to the food chain and so represents the total amount of energy that an ecosystem can use for work (Richmond et al. 2007). We define ecosystems as a system formed through the interaction of biotic organisms and abiotic factors such as sunlight, soil, and water, and ecosystem services as "the aspects of ecosystems utilized (actively or passively) to produce human well-being" (Fisher et al. 2009).

Assigning an economic value to ecosystem services does not imply that all must be preserved. From a conventional economic perspective, ecosystem services are managed efficiently when the marginal benefits (MB) derived from the economic gains associated with land-use change equal the marginal damages (MD) due to the loss of ecosystem services that are created by land-use change (Barbier and Burgess 1997). To maximize total social welfare, conversion from natural ecosystems to agriculture should stop just before expected damages from conversion are greater than the expected benefits. Unfortunately, this balance is elusive because the marginal damages due to the loss of ecosystem services are difficult to calculate and not easily internalized by practical policy instruments. Conversely, the marginal benefits of land conversion (e.g. to soybean agriculture) are easily measured by the revenue stream of the resultant economic activity.

Here, we examine the degree to which the lack of a price for ecosystem services affects the total social welfare that can be generated by the Amazon and the ability to avert welfare losses with various policy instruments. The marginal benefits of land-use change are measured by the rent associated with soybean agriculture (Vera-Diaz et al. 2008) because it is one of the key economic drivers of land-use change in the Amazon (Mann et al. 2010). Rents from soybean agriculture are generated from spatially explicit models that simulate their respective climatic, ecological, and economic determinants (Vera-Diaz et al. 2008). The resultant loss in ecosystem services is calculated from a Cobb-Douglas production function that represents the contribution of labor, capital, and NPP to GDP (Richmond et al. 2007). National estimates for NPP value are spatially disaggregated to represent differences in the ability of a unit of landscape to provide ecosystem services. Five policy scenarios (business as usual/no policy intervention, three types of land conversion taxes or conservation subsidies, and one excise tax) are used to calculate the annual domestic welfare implications of converting natural ecosystems to soybean agriculture, each of which is measured by summing the difference between rents and marginal damages over each pixel in the Amazon basin. The results indicate that ecosystem services generate economic contributions that are comparable to those generated by soybean agriculture and that policy intervention can increase the total social welfare generated by the Amazon basin.

2 Methodology

2.1 Data

All data are registered to a 991 m^2 grid with uniform missing data across layers. Landcover is defined from the 2006 MODIS product using the IGBP land classification scheme with values between 0–16 and 254 (Friedl et al. 2002). Because they cannot be target for conversion, all water, cropland, urban/built, cropland mosaic, and barren pixels are omitted from the sample. The remaining land classifications include all forest classes, shrub-land, savanna, grass, and wetlands. Forests are defined as IGBP classes 1–5. Protected area maps are acquired from Soares Filho et al. (2008) and designate four protection statuses: strictly protected, indigenous lands, sustainable use, and military. All protected areas, regardless of type, are assumed to provide perfect protection. To evaluate the role of protected areas, all reported statistics exclude conversion of protected areas (*status quo* scenario), except for where reported otherwise (*development* scenario). Due to limitations of data, we ignore the effects of other important environmental protections including "Legal Reserve" requirements. Riparian zone protection is represented by removing all 991 m pixels that are classified as water. A 991 m^2 resolution MODIS derived product for net primary productivity (NPP) in 2006 is acquired from Zhao et al. (2005) in gC/m^2/year, and is converted to $tC/991m^2$/year. Zhao estimates NPP as a function of leaf area index, temperature, fractional photosynthetically active radiation, precipitation, and edaphic properties. Annual estimates for 2006 are derived from 8-day composites, which are averaged over the calendar year. The value of net primary productivity ($/tC/year) for 2006 is calculated using data for capital stock, NPP, and labor for 2006, and represents the economic factor returns of NPP to GDP for pixel i for the 'base case random coefficient' model (Richmond et al. 2007). The map of net primary productivity is multiplied by the average marginal value of net primary productivity to estimate the unpriced contribution of ecosystem services to economic output ($/991 m^2/year).

Soybean rents ($/991 m^2/year), the marginal benefit of converting a parcel to soybean agriculture, are the expected annual returns for pixel i from soybean cropping, and are derived and updated to 2006 expected values (Vera-Diaz et al. 2008). Rent is calculated as the parcel specific difference between revenues and costs (Vera-Diaz et al. 2008). Expected revenues are the product of potential soybean yield, as calculated by a hybrid yield model that simulates climatic, edaphic and socioeconomic determinants, and soybean price. Soybean prices are updated using the average price of soybeans between 1984 and 2006. Revenues are reduced by the cost of transporting soybeans to the nearest export center (Vera-Diaz et al. 2008). This paper corrects original published findings in which some areas with zero potential soybean yields display positive rents. The efficacy of using this estimate for agricultural rent as a determinant of conversion benefits is demonstrated by logit models that indicate soybean rents are a better predictor of conversion to soybean agriculture than existing proxy variables (e.g. distance to roads), and that the effect of a one-dollar increase in costs and a one-dollar reduction in revenues are not statistically distinguishable (Mann et al. 2010). All data inputs and results are reported in US$2006.

The value of ecosystem services lost due to conversion to soybean agriculture is based on an empirical measure for the contribution of net primary production (NPP) to gross domestic product (GDP) (Richmond et al. 2007). NPP is used to proxy ecosystem services because it represents the amount of energy available to autotrophs for growth, storage, and reproduction, flows that support the entire food chain. As such, net primary production can be seen as a flow that maintains the stock of natural capital, which generates ecosystem services.

The contribution of net primary production to real GDP is estimated using an expanded Cobb–Douglas production function:

$$\mathrm{Y_{it}} = \mathrm{AL_{it}^{\alpha}K_{it}^{\beta}N_{it}^{\lambda}} \tag{1}$$

in which Y is real GDP (thousands 1996$) for nation i at time t, L is the number of workers, K is the capital stock, N is total net primary production within the borders of nation i (million kg carbon/year), A is a technology scalar, and α, β and λ are output elasticities of labor, capital, and net primary production respectively.

Equation (1) is estimated from a panel of 72 nations, including Brazil, each with nineteen observations between 1982 and 2000 (Richmond et al. 2007). The output elasticity for net primary production, which measures the effect of NPP on GDP, is not sensitive to the estimation technique or model specification. Nor does the value of λ (Eq. 1) change if the value added from agriculture, forestry, and fiber is removed from GDP. This result indicates that the contribution of NPP to GDP extends beyond any possible correlation between GDP and the value added by sectors that depend directly on terrestrial net primary production.

The marginal contribution of NPP to gross domestic product (i.e. the shadow price for ecosystem services) is calculated as follows:

$$\frac{\partial \mathrm{Y}}{\partial \mathrm{N}} = \lambda \mathrm{A_i L_{it}^{\alpha} K_{it}^{\beta} N_{it}^{\lambda-1}} \tag{2}$$

The shadow price of ecosystem services is updated to 2006 by calculating Eq. (2) using 2006 values for labor, and physical and natural capital. The shadow price of ecosystem services for Brazil in 2006 is calculated to be \$32.66 per metric ton of carbon (i.e. NPP) per year. This final calculation corrects for an important unit problem from the original published findings. Specifically, Fig. 2 in the original publication should be labeled as \$/10^3 Kg C. This error is caused by an oversight, GDP was in thousands of dollars. Using \$32.66 as the average marginal value for Brazil, ecosystems services varies across the basin based on the local rate of net primary production. Converting natural ecosystems to soybean agriculture does not eliminate all NPP, therefore NPP for soybean production is assigned a value of 140 gC m^2 each growing season (Suyker et al. 2005).

The method used to spatially disaggregate the shadow price of NPP from Eq. (1) requires some explanation. The national values for NPP that are used to estimate Eq. (1) are generated by summing spatially explicit measures of NPP within a nation's border for any given year. As such, the spatial disaggregation simply reverses the aggregation performed to calculate the NPP data used to estimate the Cobb-Douglas production function. The value of each pixel therefore represents the location and average value of the supply of ecosystem services either captured in that pixel or transported elsewhere for human use. That is, we attribute the value of the ecosystem services to the location of their presumed source, rather than the final location of their ultimate beneficiary. This method is consistent with methods used to aggregate/disaggregate other factor inputs, such as capital stock. The value of capital stock reported in the system of national accounts is based on the purchase price of the value of capital—it does not embody its 'installed value' that may be partially based on its location. Furthermore, the empirical results reported in Table 1 of Richmond et al. (2007) suggest the output elasticity for net primary production generated by the fixed effects estimator (0.16) is not statistically different from the output elasticity for net primary production estimated by random coefficients model (0.13). Based on the construction of the data and the nature of the fixed effects estimator, the fixed effects estimator represents the year-to-year relationship between GDP and NPP (and capital and labor) within each individual nation. If spatial variations in the value of a unit of net primary production dominate the non-spatial marginal value, there would be no year-to-year relation between GDP and NPP within group. We do not deny the existence of a spatial component but the within group relation estimated by the fixed effects estimator indicates that the spatially invariant value of a unit of fixed carbon must be significant.

Nonetheless, the calculation embodied by Eq. (2) probably understates the efficient price for ecosystem services because contributions of NPP to GDP are not valued completely by the market. Nor does this price include intangibles such as aesthetics, bequest, existence and cultural benefits outside of those captured by contributions to GDP through ecotourism or other services. Furthermore, the shadow price calculated here reflects contributions to domestic value added (GDP) only. Therefore, this value does not include global positive externalities, such as climate regulation due to carbon storage.

Output elasticities for priced inputs in Eq. (1) show constant returns to scale, but the positive output elasticity for the non-priced net primary production causes Eq. (1) to show increasing returns to scale. Under these conditions, the sum of value-added by all factors of production (K,L,N) is greater than the total value of GDP; therefore summing the marginal value of ecosystem services across Brazil overstates their total value. Nonetheless, Eq. (2) can be used to evaluate ecosystem services at the margin (Richmond et al. 2007).

2.2 Optimal Land Allocation

The factors that determine the optimal allocation of land among competing land-uses is an essential research focus in the land-use literature. This study builds on a model of the optimal economic distribution of land-use in the Brazilian Amazon (Barbier and Burgess 1997) by using spatially explicit data on agricultural rents and ecological service values.

For any given area of the Amazon basin, there are competing land-use options. If the parcel is held as a natural forest ecosystem, the benefits (B^F) derived are the discounted value of environmental benefits (B^E) plus any profits from sustainable yields of timber (B^T). This study assumes one non-forest land-use alternative (i.e. soybean agriculture) and that conversion costs equal the one-off timber clearing rents.

Landowners are expected to use market signals to choose the most profitable land-use. The social planner's optimality rule for the allocation of tropical forest is as follows (Barbier and Burgess 1997):

$$MB_i^{NPP} = MB_i^E + MB_i^T = MB_i^A \tag{3}$$

Land should be converted to agriculture up to the point where marginal benefits of conversion to agriculture MB_i^A for the ith pixel, are equal to benefits of ecosystem services MB_i^{NPP} for the ith pixel, where MB_i^{NPP} equals the sum of environmental flows MB_i^E plus the values from sustainable timber harvests MB_i^T. In this study, the contribution of net primary production (NPP) to gross domestic product (GDP), as described above, is used as a proxy for both components of MB_i^{NPP}. Note that MB_i^{NPP} can also be written as MD_i^{NPP}, because marginal damages are equal to the forgone ecosystem service benefits if the parcel is converted to agriculture. From the perspective of a social planner, converting natural ecosystems to soybean agriculture should proceed to the point at which the rent obtained from the last parcel converted equals the loss of ecosystem services generated by that parcel. This optimal solution will be replicated by individual landowner decisions if the costs and benefits of converting natural ecosystems to soybean agriculture are fully internalized. The failure to meet this condition will lead to land-use allocations that generate less than maximum total social welfare. This solution can be expanded from parcel-level to forest-level decision-making (Barbier and Burgess 1997). For more details on methods see Appendix.

3 Results and Discussion

We evaluate the effect of five policy options on the social welfare generated by two land-uses, soybean agriculture and natural tropical forest ecosystems. We recognize that more that two land-use options exist. Nontheless, this simple dichotomy allows us to illustrate many empirical issues associated with converting natural ecosystems to economic uses in a market that does not price ecosystem services. The effectiveness of policy options are evaluated for their ability to rectify this source of economic inefficiency. Each policy alternative is evaluated under two property regimes: a '*status quo'* that maintains the current status of federal and state protected lands and a '*development*' regime that opens all protected lands to development. Although the true status of protected areas is uncertain, protected areas are 7–11 times less likely to be deforested within the basin (Ricketts et al. 2010).

Because the market currently assigns most positive ecosystem services a price of zero, the use of agricultural rents alone to guide individual conversion decisions is not economically efficient. The area of positive soybean rents (based on conditions in 2006) can be considered an upper bound for total potential deforestation[1]. This conversion criterion, positive soybean rents, corresponds to a 'business as usual' policy scenario in which no additional effort is made to internalize the value of ecosystem services. As of 2006, soybean rents are positive ($MB_A > 0$) for about 63.5 % (Fig. 1b) of the study area (*development* regime), or for 45.6 % of land outside of protected areas (*status quo* regime). The development regime generates 68.0 billion dollars in annual soybean rents and creates soybean fields on 53.9 % of standing forests, much of which lies within protected areas, while the *status quo* regime eliminates 33.9 % of existing forest or 46.6 % of existing forests and cerrado.

To evaluate the economic efficiency of the business as usual and intervention scenarios, we compare spatially explicit estimates for the value of ecosystem services and soybean rents on a parcel-by-parcel basis. Although our estimates for soybean rents and the value of ecosystem services are generated from different data sets, models, and estimation techniques, the values are comparable. For the development regime, the value of ecosystem services is greater than soybean rents ($MD_{NPP} > MB_A$) for about 60.6 % of the total area (Fig. 1c) or 24.1 % of the area where soybean rents are positive. For the *status quo* regime, outside of protected areas, the value of ecosystem services is greater than soybean rents for 30.2 % of the total area or 13.0 % of the area that would otherwise be converted to soy. In other words, 13 % of land would be converted at a net loss to society. This implies that without government intervention that accounts for the value of ecosystem services, individuals will convert too much land to soybean agriculture. Converting all available land with positive rents reduces domestic total social welfare by 5.9 and 10.7 % relative to the maximum that is possible under the status quo and development regimes, respectively (Table 1). Maxima here are calculated as total benefits from optimal conversion behavior, which accounts for the cost and benefits of conversion.

Because ecosystems services are a positive externality, domestic welfare losses associated with excessive conversion to soybean agriculture can be avoided only if the government implements policies that internalize the value of ecosystem services. The specific policy regime used to internalize the value of ecosystem services determines the degree to which the welfare losses can be avoided.

To date, much of the intervention regarding land-use change relies on command and control policies, which forbid or limit economic development in protected areas. If protected

[1] Real estate speculation could expand deforestation beyond these bounds, but by definition, this effect is difficult to simulate and is ignored here.

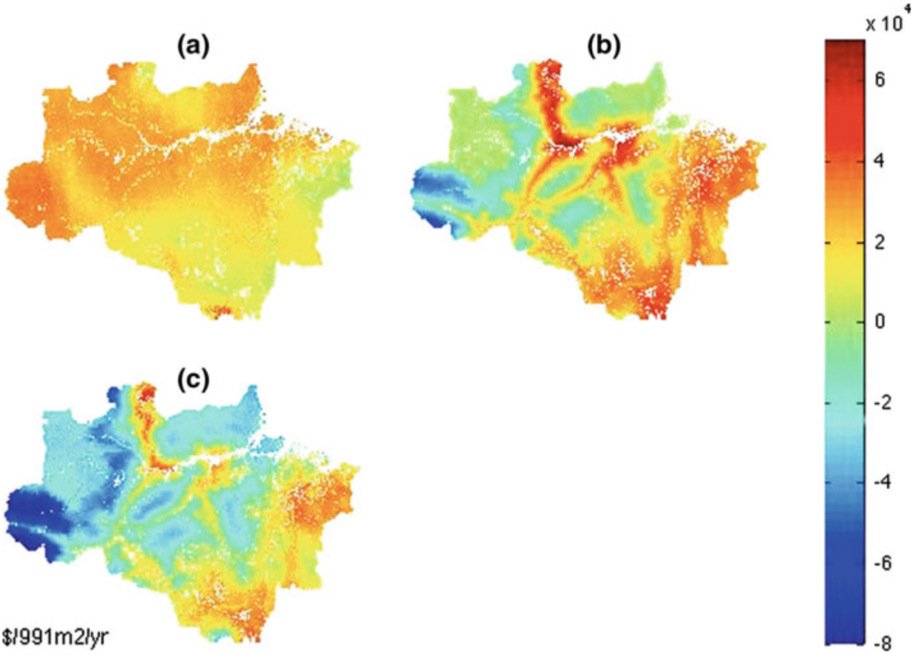

Fig. 1 Components of net social benefit of conversion to soybean for the Brazilian Amazon. Annual ecosystem system services value (**a**), annual soybean rents (**b**), and annual net social benefits of land conversion (MB_A–MD_{NPP}) from natural ecosystems to soybean agriculture (**c**), in $ per pixel (991 m^2)

Table 1 Social welfare, tax burden, & marginal rates— by policy type & protection status

Property Regime	*Policy Type*	*Policy*	*Max TSW (bill. yr $^{-1}$)*	*% Area Converted*	*Tax Burden (bill. yr $^{-1}$)*	*Marginal Rate*
Normal Protection *(Status Quo)*	Annual Land Conversion Taxes	Optimal	$120.45	32.60	$20.44	NPP_j
		Municipal	$120.07	32.88	$21.45	$211±79/ha/yr
		Uniform	$119.27	32.40	$29.08	$202.53/ha/yr
	Excise Tax	Soybean	$118.16	35.42	$26.81	$91/ton
	Other	None	$113.32	45.60	-	-
Ignoring Protection *(Development)*	Annual Land Conversion Taxes	Optimal	$124.10	39.39	$25.72	NPP_j
		Municipal	$123.73	40.05	$27.16	$213±78/ha/yr
		Uniform	$122.60	39.38	$35.35	$202.49/ha/yr
	Excise Tax	Soybean	$120.95	43.28	$33.67	$94/ton
	Other	None	$110.79	63.50	-	-

areas are chosen solely for the purpose of managing land-use conversion to agriculture, they would include parcels where conversion to soybean agriculture generates positive private benefits ($MB_A > 0$), but results in a net social loss ($MD_{NPP} > MB_A$). Consistent with this criterion, protected areas in Brazil are more likely to include parcels where positive soybean rents are less than the value of ecosystem services (Fig. 2). Indeed, the value of ecosystem services generated by protected lands in 2006 ($39.2 billion annually) is 2.7 times greater than the rents that would be generated by converting all profitable protected lands to soy. Nonetheless, not all parcels that have positive soybean rents but have a negative social conversion value ($MD > MB$) lie within protected areas. This is not surprising because areas

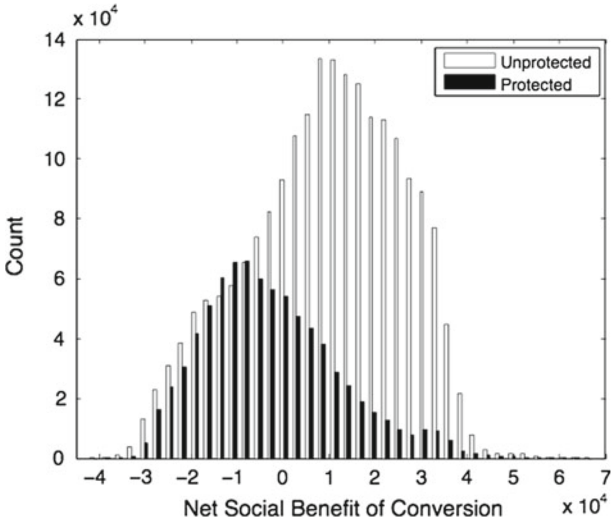

Fig. 2 Pixel count of net social benefits of conversion by protection status, protected lands (*black*), unprotected (*white*)

are protected to achieve several objectives, such as preserving indigenous peoples. Protected lands, as indicated above, are an integral piece of environmental protection throughout the Amazon basin.

Short of protecting all parcels where conversion to soybean agriculture fails to generate net social benefits, the value of ecosystem services could be internalized using a market-based mechanism, such as a Pigovian tax or a conservation subsidy. Taxes seek to increase the cost of conversion or reduce the value of output, while conservation subsidies to land owners seek to align the reward for preservation with the social benefits of ecosystem services. Results for annual land conversion taxes (Table 1) can be applied identically to conservation subsidies. The introduction of subsidies however increases land rents and would likely induce additional conversion behavior not adequately addressed in this model.

The economically efficient level of land conversion could be generated by a spatially explicit annual tax that is equal to the value of ecosystem services lost. This tax would ensure that new soybean fields are created only on parcels where annual soybean rents exceed the annual value of ecosystem services. This first-best tax in the status quo regime increases total social welfare by \$7.13 billion annually [compared to not intervening ($MB_A > 0$)]. This represents a net present value of 116 billion dollars for 30 years at a 5 % discount rate. However, implementing and administrating this spatially explicit tax across millions of parcels would likely be cost prohibitive.

In theory, an equally efficient outcome could be generated by a tax that is set to the price at which the diminishing returns to soybean agriculture (MB_A) intersects the rising costs of foregone environmental benefits (MD_{NPP}). However, the efficacy of this tax depends on the spatial relationship between the marginal benefits and marginal damages of conversion. Textbook representations of environmental taxes assume that the marginal damages and benefits of land conversion are perfectly (inversely) correlated. Accordingly, a spatial ranking of parcels from the highest marginal benefits of conversion to the lowest generates monotonic functions for both the rising marginal damage (MD_{NPP}) and the declining marginal benefit

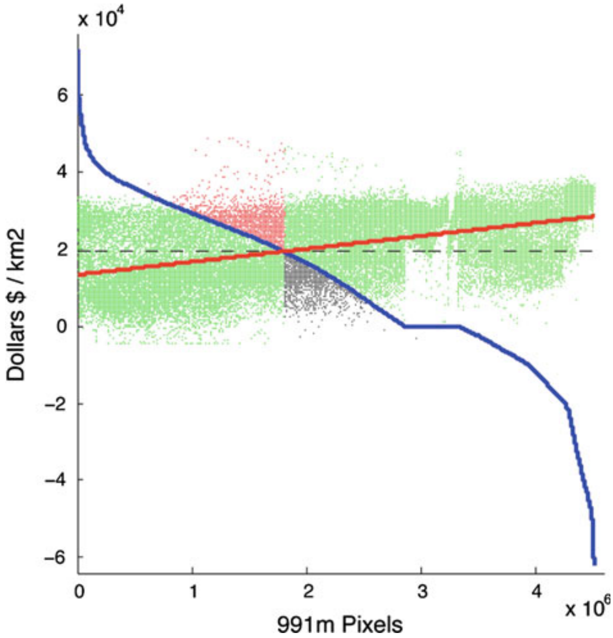

Fig. 3 Diminishing returns to soybean agriculture (*blue line*), as generated by ranking annual soybean rents. Corresponding values of ecosystem services, in the same rank order, are given by individual points (*red, green, black*). The scatter is caused by the weak correlation ($p = 0.51$) between a pixel's rent and the value of ecosystem services. The *red line* represents a best-fit line to ecosystem service value. The intersection occurs at $203 per ha. *Red points* represent pixels that are converted to soybean agriculture at a loss to total social welfare. *Black points* are not converted yet could be converted at a net social benefit. To highlight these issues, individual points for the value of ecosystem services reflect a random sample of 1/100th of all points. (Color figure online)

(MB_A) curves. According to this stylized model the single intersection between MB_A and MD_{NPP} identifies the economically efficient tax rate that maximizes total social welfare.

In the real world, deviations from perfect (inverse) spatial correlation between MB_A and MD_{NPP} means the marginal damage function may not be monotonic. If the marginal damage function does not rise monotonically, the ability of a single tax to generate the optimal level of conversion to soybean agriculture is lost. A marginal damage curve (MD_{NPP}) that rises and falls can intersect the marginal benefit curve (MB_A) at thousands of points (Fig. 3). Each intersection represents a possible tax, none of which may be optimal.

A single uniform tax rate can be identified from the intersection between MB_A and a line fit to the MD_{NPP} function ($R^2 = 0.51$). The *status quo* intersection implies a tax rate of $203 per hectare per year (Fig. 3, dashed black line). However, this tax does not generate the 'optimal conversion decision' for every parcel. Red parcels in Fig. 3, which lie to the left of the intersection of soybean rents (blue line) and the horizontal tax line (dashed black line), and above the marginal benefits curve (blue line) are converted to soybean agriculture because the tax rate is less than the value of soybean rents. These parcels' conversion reduces total social welfare because the value of ecosystem services lost are greater than the soybean rents gained ($MB_A < MD_{NPP}$). Conversely, black parcels, which lie to the right of the same intersection and below the marginal benefit curve (blue line), are not converted to soybean agriculture because the value of rent generated is less than the tax. But the failure to convert these parcels reduces total social welfare because the value of soybean rents is greater than the value of ecosystems services. These spatially uniform tax 'conversion errors' reduce *status quo* total social welfare by about 1 % relative to the optimal conversion criterion ($MB_A > MD_{NPP}$). This small percentage nonetheless, represents a substantial loss of $1.2 billion per year.

Excise taxes on agricultural outputs are another policy option. Despite the low correlation between soybean rents and ecosystem services (−0.51), it is possible to create an efficient outcome. Assuming that soybean agriculture is the only non-natural land-use, *status quo* social welfare is maximized at $118.16 billion per year (or 98.1 % of optimal) at a tax rate of $91 per ton of soy (Table 1). Although collecting this tax probably is less costly than land conversion taxes, the efficiency of an excise tax could be reduced in several ways. First, welfare measures would drop if landowners shift production to avoid taxation after the forest has already been cut. Thus, damages are incurred yet no additional tax revenues, which would recoup the value of these losses, are collected. Second, output taxes also act as a tax on productivity and therefore penalize intensive land-use, because they do not discriminate between higher yields from additional deforestation or greater productivity. Finally, total social welfare is sensitive to the chosen excise tax rate. While optimal soybean excise and land conversion taxes generate similar levels of total social welfare, errors in setting the tax rate have different effects on annual total social welfare. When converted to dollars per hectare through application of average yield (147.95 tons/Km^2), the excise and land conversion taxes generate similar levels of total social welfare (TSW) up to the point that the total social welfare associated with the excise tax reaches its maxima. As tax rates increase beyond $140 per ha per year ($94 per ton of soy), total social welfare declines rapidly (Fig. 4). Small mistakes in setting the tax rate 'too high' can have large negative consequences.

The spatial heterogeneity of soybean rents and ecosystem services creates difficulties for existing policy instruments intended to internalize the value of ecosystem services. The relatively weak correlation between a parcel's soybean rent and the value of ecosystem services implies that an excise tax on agricultural products does not maximize total social welfare. In addition, taxing individual outputs encourages production-switching away from the taxed commodity, and therefore is distortionary.

The degree to which second-best solutions can ameliorate losses can be demonstrated by modifying existing land conversion taxes to be more consistent with the spatial heterogeneity in the value of ecosystem services. The Brazilian tax code recognizes 805 municipalities within the study region. The average standard deviation of ecosystem service values within these municipality boundaries is roughly half the standard deviation of the region as a whole. Thus, the mean value of ecosystem services within each municipality could be used to set that administrative unit's tax rate for agricultural land conversion, thereby reducing efficiencies lost due to spatial heterogeneity. This regional approach has a precedent in Brazilian tax law. The Rural Land Tax (ITR), which is currently administered under federal jurisdiction by the Receita Federal of Brazil (MFFR 2008), is designed to increase the productivity of rural land use. The tax is assessed based on the size and composition of each 'module' (size defined regionally), with rates that vary between 0.03–20 % of assessed land value (MFFR 2008).

Defining municipality-level tax rates based on the mean value of ecosystem services increases the correlation between the annual land conversion tax and the value of ecosystem services from 0.51 to 0.90. The mean *status quo* municipal specific tax would be $211 per ha per year, with a standard deviation of $79 per ha per year. Compared to a basin-wide uniform tax, the municipality-level tax raises total social welfare by 0.8 billion dollars per year (Table 1). As such, the total social welfare generated by the municipality-level tax is only 0.3 % less than the maximum total social welfare generated by the optimal spatially-explicit tax. Despite these gains, the existing ITR is notoriously easy to evade because of information

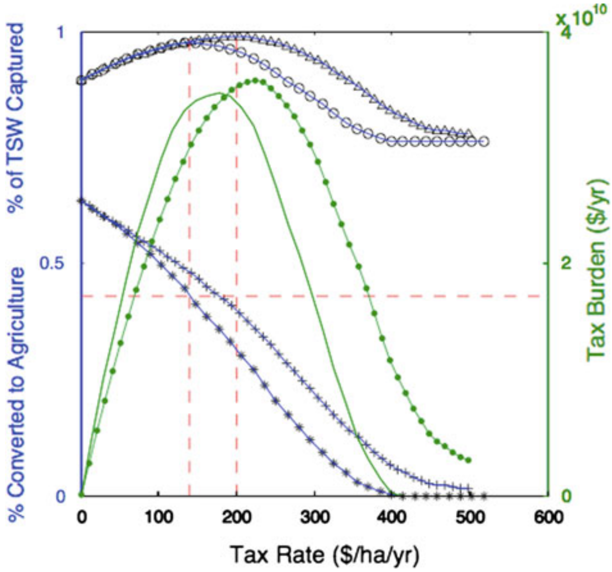

Fig. 4 Percent of total social welfare captured for the development scenario given tax levels for a soybean excise tax (*open circle*) converted to $/ha/year using average yields, and a uniform annual converted land tax (*open triangle*) $/ha/year. Percentage of study area converted to agriculture given tax levels for soybean excise tax (*asterisk*), and an annual converted land tax (*plus*). Tax burdens given tax levels for soybean excise tax (*green line*), and annual conversion tax (*green filled circle*). *Vertical red lines* pass through maxima of both total social welfare measures, *horizontal red line* indicates percent converted to agriculture at maximum social welfare for the excise tax. Colors correspond to axis label. (Color figure online)

asymmetries between the government and landholders, and often a general lack of enforcement (Moreira and Assuncao 2001). Nonetheless, the introduction of revenue sharing (50 %) with local municipalities makes it more likely to succeed (MFFR 2008).

To avoid some of these problems, incentives could be implemented as an annual conservation subsidy using the same municipality-level rates. The case for a conservation subsidy however is more tenuous. Subsidies increase the number of rent-seeking landholders, increase land values, and thereby undermine the government's goal of providing for the landless poor. Furthermore, subsidies are difficult if not impossible to retract. Finally, a source of revenue for this subsidy is unknown. Although REDD programs may provide an ample source of revenue this would not satisfy the criterion of 'additionality' because the incentives discussed in this paper are wholly domestic.

Despite these benefits, it is unclear how modifying ITR tax rates to reflect the local value of ecosystem services would interact with existing tax policy and regulation. Although *reserva legal* requires landowners to hold significant portions of their property in forest (up to 80 % in the legal Amazon), this requirement probably is not binding and/or is currently unenforceable in many areas. However, a modified ITR tax (or subsidy) would compliment *reserva legal* regulations by providing clear financial disincentives to unproductive conversion.

Regardless of policy regime, this analysis indicates that the benefits of converting natural ecosystems to soybean agriculture show diminishing returns. For example, under the optimal tax regime (*development* regime) capturing the final 10 % of total social welfare requires the conversion of 16.6 % of natural ecosystems. These final conversions generate, on net, less than \$52 per ha per year. Actual net returns are likely to be smaller given the probable downward bias in the estimate of ecosystem service values. As such, a cautionary approach to land conversion probably would generate the greatest benefits to the Brazilian economy.

Beyond the implications for social welfare, the policy instrument used to internalize the value of ecosystem services has a significant effect on the percentage of area converted to soybeans. The optimal intervention reduces the area converted from 45.6 % of unprotected lands to 32.6 % assuming all profitable lands are converted (Table 1). The area converted changes substantially among policy interventions (land conversion tax, product excise tax, or other), while the level of spatial disaggregation (among land conversion tax types) predominately affects total social welfare and tax burden. While applying a uniform tax converts 0.2 % less natural ecosystems than the optimal tax in the status quo regime, these improvements are the result of efficiency losses as indicated by the lower levels of social welfare and higher tax burdens.

Comparing the results for the status quo and development regimes, protected lands provide substantial conservation benefits by reducing the percentage of area converted at the cost of reduced economic efficiency. Despite results that indicate protected lands are relatively well targeted (towards areas with $MD_{NPP} > MB_A$), the total social welfare of outcomes generated by the 'optimal tax' under the status quo regime are about 3.65 billion dollars less per year than the corresponding policy scenario generated by the development regime. Conversely, protected areas offer advantages by reducing the percentage of natural ecosystems converted to soybean agriculture, nearly 7 percentage points for the optimal tax scenario. However, reducing the level of deforestation comes at the cost of foregone social welfare gains. This balance implies a domestic social cost for protection of approximately \$12 thousand per km^2 or \$120 per hectare per year. This result does not argue against protected lands, but the need to be cognizant of both the ecological benefits and economic value of these lands.

4 Limitations

4.1 Down-Scaling

The national-level shadow price of carbon is applied uniformly to a raster of net primary productivity estimates. Here the implicit assumption is that each unit of net carbon uptake contributes the average marginal contribution to gross domestic product. While an imperfect assumption, data do not allow for the sub-national accumulation of capital or, for most countries, value added or labor. The applicability of this assumption will likely depend on which ecosystem services are represented. For instance, the uniform application of an ecosystem service price might be suitable to represent the contributions of evapotranspiration – where areas at great distance rely on upwind hydrological cycles. In the case of the Amazon, approximately 70 % of the rainfall in the state of Sao Paulo is generated from water vapor put into the atmosphere a minimum of 480 miles away (Fearnside 2005). Conversely, the contributions of fuel wood and agriculture would be highly dependent on access, and would therefore be at odds with the assumption of average contributions. These estimates therefore should be considered exploratory yet suggestive in nature rather than absolute.

4.2 Other Protected Areas

Another important limitation is the failure to control for the effects of "Legal Reserve" or *reserva legal* (RL) requirements. Under RL requirements, Amazon landholders must keep 80 %

of their property in native forest, 35 % for Amazonian cerrado and 20 % elsewhere (WHRC 2007). The development regime assumes that all non-federal and non-state protected lands are available for conversion, and therefore overstates the land eligible for development under current regulation. Consequently, this assumption distorts the benefits and costs of protected and unprotected land. We contend that *reserva legal* requirements are politically unsustainable in the medium to long-term. Substantial efforts are underway to decentralize control over RL and riparian zone protection laws (WWF 2011). There is also political pressure to exempt small landholders (25–400 ha) from RL laws (Santhanna 2011) and to apply reserve requirements for areas greater than four modules (25–110 ha) (Braziliense 2011). These reforms also aim to expand legal reserve requirements to include permanent preservation areas (APPs) such as riparian zones, hill tops, and steeply sloped land ($> 45°$) (Braziliense 2011). Because RL and APP requirements are not modeled here, our findings might be viewed as a baseline scenario.

4.3 Market Assumptions

This study assumes only one economic land-use, soybean production. Meanwhile Amazonian pastures support over 70 million animals, and account for over 80 % of deforested lands (Walker et al. 2009). At the time of writing, no spatially explicit measures of cattle rent are available. Concurrently, two estimates for cattle rents are being developed (Bowman et al. Under Review; Mann et al. in preparation). Once validated, these efforts can be used to extend the analysis described here.

This study implicitly assumes that land rents are the sole determinate of agent behavior. In early periods of frontier expansion, landholder behavior in the Amazon is motivated in part by rent seeking opportunities associated with subsidized credit, preferential tax treatment, risk hedging, and land rents unearthed through transportation investments. An unusually large body of literature emphasizes the importance of accessibility on land cover change (e.g. Anderson et al. 2002; Pfaff 1999; Pfaff et al. 2007; Walker and Oyama Homma 1996). While credit, tax treatment, and accessibility are important drivers of land conversion, they are all subcomponents (revenues and costs) of location rents, which can be considered the proximate cause of land cover and land-use change (Mann et al. 2010; Walker et al. 2009). With the exception of some risk hedging benefits, the value of land bought or sold depends on the capture of some part of current and future revenue streams. It should be noted however that the ‘outcomes’ presented in this paper represent ‘long run’ outcomes. The evolution of land use patterns takes place over long periods and is hampered by a variety of processes not modeled here.

The results described above depend in large part on the spatial pattern of soybean rents and the value of NPP. This spatial pattern is not static. Pixels here are assigned fixed marginal benefits and damages based on economic and ecological conditions in 2006. In a dynamic model the benefits of growing soybeans will change with total production and changes in transportation infrastructure. Furthermore, damages from conversion would be assumed to increase as natural land covers become increasingly scarce. These dynamics, amongst others, will affect the distribution and extent of net positive social benefits to conversion (Vera-Diaz et al. 2008).

5 Conclusion

This study evaluates the ability of five policy scenarios: business as usual/no policy action, three types of annual land conversion taxes, and a soybean excise tax under two property regimes to increase total social welfare by internalizing the value of ecosystem services. The efficiency of each policy instrument depends on the degree to which it is correlated with the externality (loss of ecosystem services), as well as the costs of intervention (Table 1). As Brazil seeks to expand its tax base, a land conversion tax may be more desirable than a soybean excise tax. Of the three land conversion taxes evaluated, a municipality-level tax provides a good balance between the correlation with ecosystem services and the cost of implementation. This tax generates an additional \$6.8 billion annually in social benefits over no policy action or \$1.1 billion annually over the uniform tax. This advantage carries two caveats: annual land taxes are difficult to collect in Brazil, as seen with the ITR tax scheme, and annual collections may encourage landowners to abandon converted land without full recuperation of environmental damages. These difficulties may be alleviated with a one-time upfront tax, at the time of land conversion that incorporates all discounted foregone ecosystem benefits.

This study also highlights the domestic economic benefits that Brazil receives from its ecosystem services. Quantifying these benefits is consistent with significant domestic conservation efforts yet suggests that continuing (or expanding) these efforts using command and control policies may not be economically efficient. As pressure mounts to dismember *reserva legal*, APA requirements, policymakers need to look to alternative, cost effective policy instruments. Although efforts such as REDD may offer some assistance, Brazilian lawmakers may find it necessary to take additional measures. As such, Brazil is unlikely to reach the domestically optimal level of conversion without internalizing the significant positive value of ecosystem services. Such a policy could increase the total social welfare derived from the Amazon by as much as 116 billion dollars over 30 years, or about 8 % of GDP in 2006. This same effort would limit deforestation to an additional 18.6 or 24.7 % if the status of existing protected lands is not preserved.

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Appendix

Calculations

Baseline total social welfare (TSW) is defined as follows:

$$\max \ \mathrm{TSW} = \int_{1}^{n} (\mathrm{LandValue_i})$$
$$\text{s.t.} \ \ \mathrm{LandValue_i} = \begin{cases} \mathrm{MB_i^a} & [\mathrm{MB_i^a}] > 0 \\ \mathrm{MB_i^{NPP}} & [\mathrm{MB_i^a}] \leq 0 \end{cases} \tag{4}$$

where LandValue is the value of pixel i, $\mathrm{MB_i^A}$ is the annual marginal benefits of agriculture for pixel i, $\mathrm{MB_i^{NPP}}$ is marginal benefits of natural ecosystems for pixel i, and n is the total number of pixels in the area of interest. *Status quo* scenarios exclude all protected lands from calculations.

Total social welfare for the spatially explicit, pixel-level ecosystem services tax policy is maximized as follows:

$$\mathrm{TSW} = \int_{1}^{n} \left(\max(\mathrm{MB_i^a},\ \mathrm{MB_i^{NPP}})\right) \tag{5}$$

where MB_i^A is annual soybean rent for pixel i and MB_i^{NPP} is the annual value of ecosystem services for pixel i.

Total social welfare for the soybean excise tax policy is maximized as follows:

$$\max_{t_s} \ TSW = \int_1^n (LandValue_i)$$
$$\text{s.t. } \ Land\ Value_i = \begin{cases} MB_i^a & [MB_i^a - t_s(yield_i)] > 0 \\ MB_i^{NPP} & [MB_i^a - t_s(yield_i)] \leq 0 \end{cases} \qquad (6)$$

where LandValue is the value of pixel i, $yield_i$ the soybean yield in tons for pixel i, and t_s is the soybean excise tax rate per ton.

Total social welfare for land conversion tax policies are maximized as follows:

$$\max_{t_m} \ TSW = \int_1^n (LandValue_i)$$
$$\text{s.t. } \ LandValue_i = \begin{cases} MB_i^a & [MB_i^a - t_m] > 0 \\ MB_i^{NPP} & [MB_i^a - t_m] \leq 0 \end{cases} \qquad (7)$$

where t_m is the municipality-specific tax rate and t (m is dropped) is the uniform annual land conversion tax rate.

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