Direct conservation payments in the Brazilian Amazon: Scope and equity implications

Jan Börner [a,*], Sven Wunder [b], Sheila Wertz-Kanounnikoff [c], Marcos Rügnitz Tito [d], Ligia Pereira [e], Nathalia Nascimento [f]

[a] Amazon Initiative & CIAT, c/o Embrapa, Tr. E.Pinheiro s/n, 66095-780, Belém-PA, Brazil
[b] Center for International Forestry Research (CIFOR), Rua do Russel 450/ s.601, 22.210-010, Rio de Janeiro-RJ, Brazil
[c] Center for International Forestry Research (CIFOR), P.O. Box 0113 BOCB, Bogor 16000, Indonesia
[d] Amazon Initiative & ICRAF, Embrapa, Tr. E.Pinheiro s/n, Belém-PA, Brazil
[e] Conservation International, c/o Av. Gov. José Malcher, 652, 2°, 66.035-100, Belém-PA, Brazil
[f] Federal University of Pará, Rua Augusto Corrêa 1, Belém-PA, Brazil

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ABSTRACT

This article looks into the scope and equity implications of applying payments for environmental services (PES) as a REDD implementation mechanism in the Brazilian Amazon. We establish a set of economic and institutional preconditions for PES to become a feasible and cost-effective conservation mechanism. We proceed with a macro-scale spatial analysis and overlay of opportunity costs, deforestation patterns, carbon services, and land tenure, in order to assess where these conditions hold. We then screen how the benefits of potential PES schemes might be distributed across different socioeconomic groups of service providers in different land tenure categories. Our economic–quantitative analysis, though sensitive to documented assumptions, suggests that under current carbon prices the economic preconditions are in place to pay for avoided deforestation in over half of threatened forests over the next decade. Unfortunately, the same optimism does not apply to institutional preconditions. Land grabbing, insecure tenure, overlapping claims, and lacking information on private tenure constitute real medium-term impediments to PES. If payments were to accrue to current landholders regardless of current tenure insecurities, large landowners who account for about 80% of all deforestation would reap the highest benefits, though per-capita benefits other tenure categories are also high. Schemes that closely align payments with opportunity costs are preferable for cost-effectiveness, and not necessarily more inequitable in outcomes. Essentially, PES systems cannot substitute command-and-control measures: the former depend on the latter for basic governance systems to secure effective rights of exclusion, which land stewards essentially need in order to become reliable service providers.

1. Introduction

Few issues have dominated the recent environmental debate as much as the proposal of including Reduced Emissions from Deforestation and Degradation (REDD) in a post-2012 international climate-policy agreement. Since the 2007 Conference of the Parties (COP13) in Bali, several international donors and industrialized countries have earmarked substantial funding for conditional cash transfers to tropical countries that commit to reducing deforestation rates (Dutschke and Wertz-Kanounnikoff, 2008).

Two characteristics make the Amazon region a chief target for REDD initiatives. First, it holds the world's largest tropical rainforest, with biomass corresponding to about 1.5 decades of anthropogenic carbon emissions (Soares-Filho et al., 2006). Second, Brazil has the world's highest absolute deforestation amounts—almost one fifth of global CO_2 emissions from land-use change.[1] According to the Brazilian National Institute for Space Research (INPE), between 2000 and 2005 roughly 130,000 km^2 of Brazilian Amazon forest were lost (see also Fig. 1, Section 3). In addition, reducing forest emissions may secure important co-benefits, conserving biodiversity, cultural values, and regional climate regulation (Malhi et al., 2008; Turner et al., 2007).

Many REDD proponents embrace the concept of payments for environmental services (PES), because it can be seen as the most direct form of providing conservation incentives to local land users (Ferraro and Kiss, 2002). PES can be defined as a voluntary transaction, where a well-defined ES (or a land use likely to secure

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* Corresponding author. Fax: +55 91 3204 1108.
E-mail addresses: j.boerner@cgiar.org (J. Börner), s.wunder@cgiar.org (S. Wunder), s.wertz-kanounnikoff@cgiar.org (S. Wertz-Kanounnikoff), m.tito@cgiar.org (M.R. Tito), ligiaspereira@ig.com.br (L. Pereira).

[1] Climate Analysis Indicators Tool, World Resources Institute (data from 2000): http://www.wri.org/.
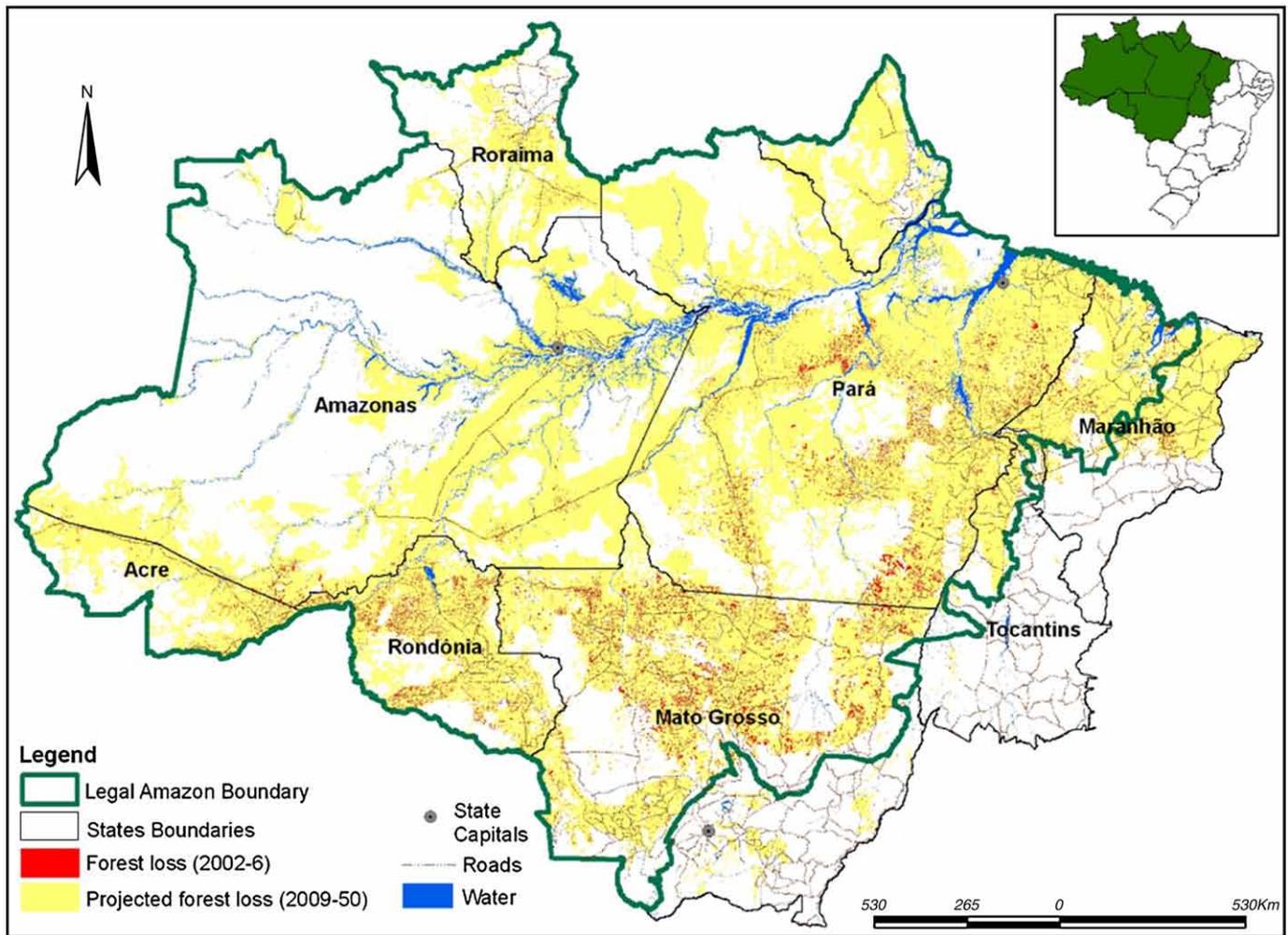
Fig. 1. Historical (2002–06) and predicted (2009–50) deforestation in the Brazilian Amazon.
Sources: INPE-PRODES, Soares-Filho et al. (2006).

that service) is paid for under the condition that the seller continues to provide the service (Wunder, 2005). Compared to command-and-control and other disincentive-based environmental policies, the PES concept has two innovative features: voluntariness and conditionality. Voluntariness prevents service providers from bearing the often substantial opportunity costs of conservation vis-à-vis locally more profitable land-use options. Thus, PES is at least welfare-neutral for those who participate in a voluntary scheme. Conditionality implies that non-compliance can be sanctioned by reducing or discontinuing payments, thus at least partially foregoing the enforcement costs of disincentive-based policies.

Much of the PES debate focuses on the potential to increase the effectiveness of environmental policy. Taking stock of a broad range of case studies in both developed and developing countries, a recent special issue in this journal[2] found that PES schemes likely differ remarkably in terms of conservation effectiveness. For example, a variety of PES-labeled conservation schemes fall short of making payments fully conditional on service provision (see also Muradian et al., 2010-this issue and Vatn, 2010-this issue). The review, nevertheless, stresses the demonstrated potential for cost-effective PES schemes to explicitly address the tradeoffs among private landowners' and society's wellbeing.

However, for various reasons PES will not work everywhere. One of the objectives of this paper is therefore to assess the potential *scope* of PES as a REDD vehicle in the Brazilian Amazon. While we recognize that several preconditions exist for PES to be a cost-effective and equitable environmental policy instrument, we will assess the major economic and institutional preconditions for PES. By definition, PES only makes sense if an externality exists, i.e. an outside benefit (e.g. watershed or carbon-stock protection) for which external beneficiaries are willing to remunerate local land users. A major economic precondition for PES thus is that beneficiaries' willingness to pay exceeds providers' willingness to accept. Put differently, the value of the service(s) at hand must at least exceed provider's opportunity costs—determined by profits foregone from abandoning their first-best land-use plan, plus transaction costs.

Although natural-resource externalities are widespread globally, in few places have PES been developed locally without external support. The lack of adequate institutional preconditions is often the main bottleneck (Vatn, 2010-this issue). The institutional preconditions for PES are manifold, and range from trust between service users and providers (in user-financed schemes) to the capacity of governments to effectively secure PES contract monitoring and enforcement (in government-financed schemes). Yet, one fundamental precondition is the exclusiveness of rights to the land providing the service in question. If local land users, i.e. service providers, cannot guarantee service provision on the basis of the right to exclude others from modifying service quantity and quality, PES is not applicable. Given the frequency

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[2] Special Issue of Ecological Economics 65(4) on PES.

of insecure land tenure arrangements in the Southern Hemisphere, this institutional PES constraint is often binding.

Regardless of whether REDD implementers contemplate PES or other conservation tools, effectiveness will not be the only bottom line. Welfare impacts will also be closely screened, including both poverty alleviation and equity (cf. Wunder, 2008 and Pascual et al., 2010-this issue). Forest conservation policies that hurt the poor or privilege the wealthy disproportionately may not be acceptable to those who are to pay for REDD and/or the governments in recipient countries. The second objective of this paper is thus to scrutinize the expected impact of PES-led REDD on the distribution of benefits among potential Amazon service providers. In summary, we provide an overlay of spatially explicit criteria for biophysical, economic, and distributional scope, which is the main potential contribution of our paper to the REDD debate in Brazil, and beyond. Policy makers need to be better informed about where and under which circumstances PES is likely to contribute to effective and equitable REDD outcomes.

The paper is organized as follows. Section 2 documents the data sources and approaches used to assess potential scope, costs and equity implications of reducing deforestation in the Brazilian Amazon region. In Section 3 we present the results regarding scope, opportunity costs, and equity implications of selected PES modalities. Section 4 concludes with a discussion of the results in the wider context of REDD in the Amazon and outlines emerging implications for policy design.

2. Data and Methods

2.1. Estimating REDD Opportunity Costs

Land stewards' opportunity costs of avoiding deforestation consist of the foregone net benefits of converting forests to other land uses. We previously used a municipal-level methodology based on official Brazilian deforestation and land-use statistics in a pilot analysis for Mato Grosso and Amazonas federal states (Börner and Wunder, 2008). In the following, we will extend this approach to the entire Brazilian Amazon, including additional data sources (Table 1).

Our analysis requires a series of assumptions. Land-use expansion data from The Brazilian Institute of Geography and Statistics (IBGE) (2) notoriously underestimate conversion of forests to agriculture, as better evidenced by INPE data. We thus assume agricultural expansion (2) to mirror forest loss (1), and the land-use mix after clearing to follow the patterns suggested by IBGE data. Lacking municipal-level profit rates (4), we use alternative secondary sources from the literature (Table 1) to provide cost–benefit estimates for

selected Amazon sites. For instance, considering cattle ranching returns from Margulis (2004), we applied his high-range estimates to Mato Grosso, low-range estimates to Amapá and Amazonas, and medium-range values to all other states, due to well-documented inter-state differences in sectoral development (Chomitz and Thomas, 2001). Average timber yields from Barreto et al. (1998) were applied to all states but Mato Grosso and Maranhão, where forest-savanna transition zones have about 30% lower timber yields (Mato Grosso State Environmental Secretariat, personal communication, 2007). To simplify, we also neglected benefits from standing forests (e.g. non-timber forest products) and assumed real product and input prices to remain constant over time (see Börner and Wunder, 2008 for more discussion of the underlying assumptions and their implications).

Estimating opportunity costs of forest conservation requires taking into account multiple sequential land uses over time. According to Vosti et al. (2002), these land-use trajectories after deforestation often follow similar patterns. Timber extraction is usually followed by a short cycle of annual crops, before land is turned into pasture, permanent annual crops (e.g. soy), itinerant annual cropping, or perennial cash-crops. Economic returns to these land-use trajectories, as measured by net present values (NPV), differ widely (Table 2).

NPV values in Table 2 represent the opportunity costs of leaving the forest untouched, and are calculated at the municipal-level, according to the following three steps:

$$\Pi_{ik} = GR_{ik} * \left(1 - \frac{c_k}{b_k}\right) \tag{1}$$

where Π_{ik} is net profit per ha of crop k in municipality i, GR are annual gross returns per ha in i calculated from PAM/PPM/PEV data. b and c are per ha gross returns and total costs, respectively, derived from the sources documented in Table 1.

$$NPV_j = \sum_t \frac{\Pi_{k=1,t=1}}{(1+r)^{t=1}} + \frac{\Pi_{k=2,t=2}}{(1+r)^{t=2}} + \dots + \frac{\Pi_{k=K,t=T}}{(1+r)^{t=T}} \tag{2}$$

where NPV_j is the net present value per ha of land-use trajectory j in a given municipality and k depicts the different crops/land uses that follow each other during a planning horizon (with $T = 10$) in j.

$$NPV_i = \sum_t \frac{\sum_j s_j NPV_{ij}}{(1+r)^t} \tag{3}$$

where NPV_i is the net present value per hectare in municipality i, s is the share of land-use trajectory j in the total municipality's annual land-use expansion, and NPV_{ij} is the net present value of the ten-year land use trajectory j in i, while r is the discount rate.

The NPV values presented in the third column of Table 2 are the result of step 2 above. Step 3 yields an aggregate measure of opportunity costs at the municipality level, considering that each year a new land-use trajectory is initiated on newly deforested land during an established period of time (here: ten years). This aggregate

Table 1
Data sources used in this study.

Input data	Source
1. Annual deforestation at municipal-level (2000–6)	INPE-PRODES[a]
2. Dominant expanding land-use categories at municipal-level (2000–6)	IBGE-PAM/PPM/PEV[b] and Agricultural censuses 1996 and 2006 (preliminary results)
3. Deflated gross per-hectare return returns to 2.	IBGE-PAM/PPM/PEV, IPCA
4. Profit rates for land-use categories under 2.	FNP (2007), Margulis (2004), Holmes et al. (2004), Barreto et al. (1998)
5. Municipal-level per-hectare biomass/carbon estimates	Saatchi et al. (2007)
6. Simulated future deforestation scenario for 2003–2050	Soares-Filho et al. 2006
7. Location and size of land-reform settlements	INCRA (2007)
8. Location and size of protected areas	IBAMA (2007)

[a] The Brazilian Space Research Centre's (INPE) Program for the Calculation of Deforestation in the Amazon.
[b] The Brazilian Institute for Geography and Statistics (IBGE). Municipal Agricultural, Animal, and Extractive Production data bases (PAM/PPM/PEV).

Table 2
Average share and net present value (NPV) for major land-use trajectories in the Brazilian Amazon.

Land-use trajectory	Average share in land-use expansion [% of expanding land uses]	Average NPV[a] [R$[b] per hectare]
Pasture	79%	$1661.77
Fallow-based annual cropping	6%	$1615.08
Permanent annual cropping (soybeans)	12%	$2358.56
Perennial cash-crops	2%	$6603.04

[a] Time horizon of $T = 10$ years and 10% discount rate.
[b] US$1 = R$2.2; 10.2008.

municipal-level measure of opportunity costs represents the main input to our analysis in Section 3.1.

2.2. Estimating Equity Implications among REDD Sellers

If policymakers choose PES to land stewards as a key component of a REDD strategy, two distributional questions become key (Wunder, 2008)[3]: First, who will be selected as participating sellers of environmental services? And second, how much will different seller groups derive in net benefits? Starting with the question of participation, potential recipients of direct REDD payments could in principle come from the following land tenure categories (in brackets, approximate land coverage in the Brazilian Amazon[4]):

1. Indigenous lands (~22%)
2. Protected areas allowing for sustainable use (below labeled sustainable use areas) (~10%)
3. Formal rural settlements (~13%)
4. Private land (including large landholders, smallholders, and community-owned lands—each in different stages of tenure regulation) (~24%)

Inhabitants of the other two major tenure categories would generally not qualify for direct REDD payments:

5. Unclassified public land (terra devoluta) (~24%)
6. Strictly protected areas—(e.g. national parks) (~7%)

Dwellers on public land, with few exceptions, could normally not guarantee effective rights of third-party exclusion, and are thus inefficient service providers. Strictly protected areas, by their definition and by law, rule out any legal economic activity 'avoided', the opportunity costs of which one could reasonably compensate for through REDD interventions. REDD thus generally has to be targeted to the four first-mentioned tenure categories, even if several restrictions may apply, as pointed out in Sections 3 and 4.

To answer the second question, regarding the size distribution of net benefits, we propose to classify total net PES benefits of eligible service sellers as:

$$NB_T = NB_{IL} + NB_{SU} + NB_{RS} + NB_{SL} + NB_{LL} \qquad (4)$$

where NB = net benefit (in absolute terms or per-capita terms) and subscripts refer to seller categories: IL = sellers on indigenous land, SU = sustainable use areas, RS = land-reform settlements, SL = small landholders and community lands, LL = large landholders.

Net benefits for each seller category are defined as:

$$NB_i = AD_i * (P - OC_i) \qquad (5)$$

where i is the seller category, AD is the amount of additional REDD each seller category can provide, P is the price paid (per hectare of deforestation or ton of emissions avoided) and OC is the related opportunity cost per seller category, i.e. the equivalent to NPV_j in Eq. (2).

Net benefits of REDD sellers are the difference between payments received and the individual opportunity cost (neglecting for a moment seller transaction costs). Net seller benefits may vary for several reasons, including the selected pricing mechanism. We are looking at three options. First, REDD buyers (i.e. governments or external funders) could buy emission offsets at market prices according to each plot's specific carbon content. Second, buyers might not bother with carbon density, and instead pay a fixed per-

hectare price for conserved forest. Third, buyers might want to customize payments further through negotiated or quasi-auction processes. In Section 3.2, we discuss hypothetical payment modalities that reflect these three alternatives.

Disaggregating total and per-capita net benefits by tenure category (Eq. (4)) will allow us to examine the distributional effects of each payment scenario across REDD seller categories. A few important caveats, however, apply. Some public land tenure categories (indigenous territories, protected/sustainable use areas, and land-reform settlements) are generally well-delimited in space, whereas Amazon private land registers are often faulty and overlapping. While well-delimited private land would be perfectly eligible for PES, ill-delimited plots often cannot be separated from unclassified public land. Similarly, while the land-cover percentages above represent general approximations, we cannot distinguish smallholder areas from large landholders in a geo-referenced manner. Land tenure chaos thus represents the single largest impediment to our analysis, and to REDD implementation. Nevertheless, the following sources provide at least some guidance:

1. The IBGE Agricultural Census 1996/7 published district-level land distribution among agricultural establishments, but does not accurately determine tenure forms. A new agricultural census (2006) is expected to be published during 2009.
2. The National Agrarian Reform Institute's (INCRA) 2005 rural land register informs about private land titles and declared size of registered establishments, but is known to hold significant inconsistencies, i.e. overlaps of registered titles, especially in the states of Mato Grosso, Pará and Rondônia.[5]
3. As suggested by Chomitz (2006), the size of annually deforested areas (e.g. polygon areas of classified satellite images provided by INPE) could be used as an indicator of deforestation agency. For example, deforestation polygons exceeding 20 ha are very unlikely to be caused by smallholders.

To avoid double counting (e.g., of farms inside public land tenure categories considered private establishments in the IBGE data), we decided to use the more recent INCRA data to determine the number of small (≤100 ha) and large landholders (>100 ha) in each district. Following Chomitz (2006), we further assumed that:

a) clearings smaller than 20 ha (years 2002–2006[6]), and situated outside known public land categories, were caused by expanding smallholder agriculture;
b) deforested areas larger than 20 ha were correspondingly attributed to large landholders;
c) the observed municipal ratio between large- and smallholder deforestation remains constant.

The final lacuna is how deforestation outside known public land categories divides up between private and unclassified public land—the former usually being adequate, the latter being unsuitable for PES. We will discuss some 'best guesses' in Section 3.2.

3. Analysis and Results

3.1. Opportunity Costs and REDD Additionality

PES only makes environmental sense when it can increase service provision relative to a business-as-usual scenario, i.e. a dynamic baseline of what would happen over time without PES. PES-induced

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[3] Other equity questions are discussed in Section 4).

[4] Information from the Socio-Environmental Institute (ISA), Amazon Institute of People and the Environment (IMAZON), National Agrarian Reform Institute (INCRA), Brazilian Institute for the Environment (IBAMA), and IBGE.

[5] In some municipalities, the sum of all privately claimed areas in INCRA's registers actually exceeds total municipal area, due to large overlapping and unsettled claims. INCRA began in 2001 a country-wide effort to reregister rural properties, but had in 2008 processed merely 4% of agricultural establishments (Barreto et al., 2008).

[6] We excluded 2000/01 data from the analysis, because the total size of polygons attributed to these years in the spatial data provided by INPE was largely inconsistent with official annual deforestation rates.

increases in service provision vis-à-vis the baseline are labeled "additional". Based on empirically observed spatial dynamics of Amazon deforestation, Soares-Filho et al. (2006) simulated future deforestation until year 2050. Fig. 1 shows observed deforestation for 2002–06 (11.5 million ha) and the by Soares-Filho et al. predicted deforestation for 2009–50 (140.5 million ha). Forests in yellow areas can thus be considered potentially threatened. Here, conservation interventions—whether PES, command-and-control investments, or protected-area creation—are "additional" in reducing forest loss relative to the business-as-usual scenario. Conversely, any action in non-threatened areas is "non-additional": it has no potential to make a difference for the targeted environmental outcome.

According to Fig. 1, past deforestation occurred mainly in what is known as the "arc of deforestation" at the southern to northeastern borders of the Brazilian Amazon and alongside important terrestrial transport ways, such as the Transamazonian Highway. Forest loss was highest in Mato Grosso and Pará states, where cattle and crops have been most expansive. Due to infrastructure expansion, future clearing is gradually expected to reach remoter forests in traditional low-deforestation states, such as Amazonas and Amapá.

Fig. 2 characterizes threatened forest areas in terms of land tenure. Two thirds of future deforestation (67%) will take place in areas under hitherto ill-defined tenure, i.e. on unclassified public lands or private land predominantly without consolidated boundaries. Another 8% is projected to occur in strictly protected areas, whereas indigenous territories and sustainable use areas each account for 9%, and land-reform settlements for 7% of projected deforestation.

Following the calculus outlined in Section 2.1, Fig. 3 characterizes threatened forest areas in terms of the opportunity costs of not converting them to agriculture. Carbon prices on voluntary markets usually apply to permanent certified emission reductions, but emission reductions from avoided deforestation are often considered temporary. We illustrate this by applying a 39% discount to convert the 2007 average price of permanent carbon offsets at the Chicago Climate Exchange market to temporary offset prices, following Dutschke and Schlamadinger (2003). Green areas (81%) thus depict opportunity costs that lie below the average price of temporary carbon offsets (Hamilton et al., 2008). Conversely, in red areas (19%) opportunity costs are currently too high for PES to 'buy out' deforestation. Red, uncompetitive areas are often close to cities and roads, tend to have low biomass, or high-value remaining timber resources. Large continuous low-cost REDD opportunities are concentrated primarily in the western Amazon (Amazonas and Acre states), but also northern Amapá. These areas combine limited, high-cost access with high biomass density.

Fig. 4 depicts a potential Amazon REDD supply or abatement-cost curve. Different from the 2050 horizon in Fig. 3, this curve is based on an initial ten-year REDD program (2009–18). Grey areas show a sensitivity analysis with regard to key input parameters, such as profit rates, carbon content, and NPV discount rate. Using standard Monte Carlo simulation, these parameters were simultaneously varied by ±30%, resulting in independent oscillations around mean values. In the ten-year scenario, voluntary-market carbon prices for temporary offsets would cover the opportunity costs of approximately 12.5 million ha of projected forest
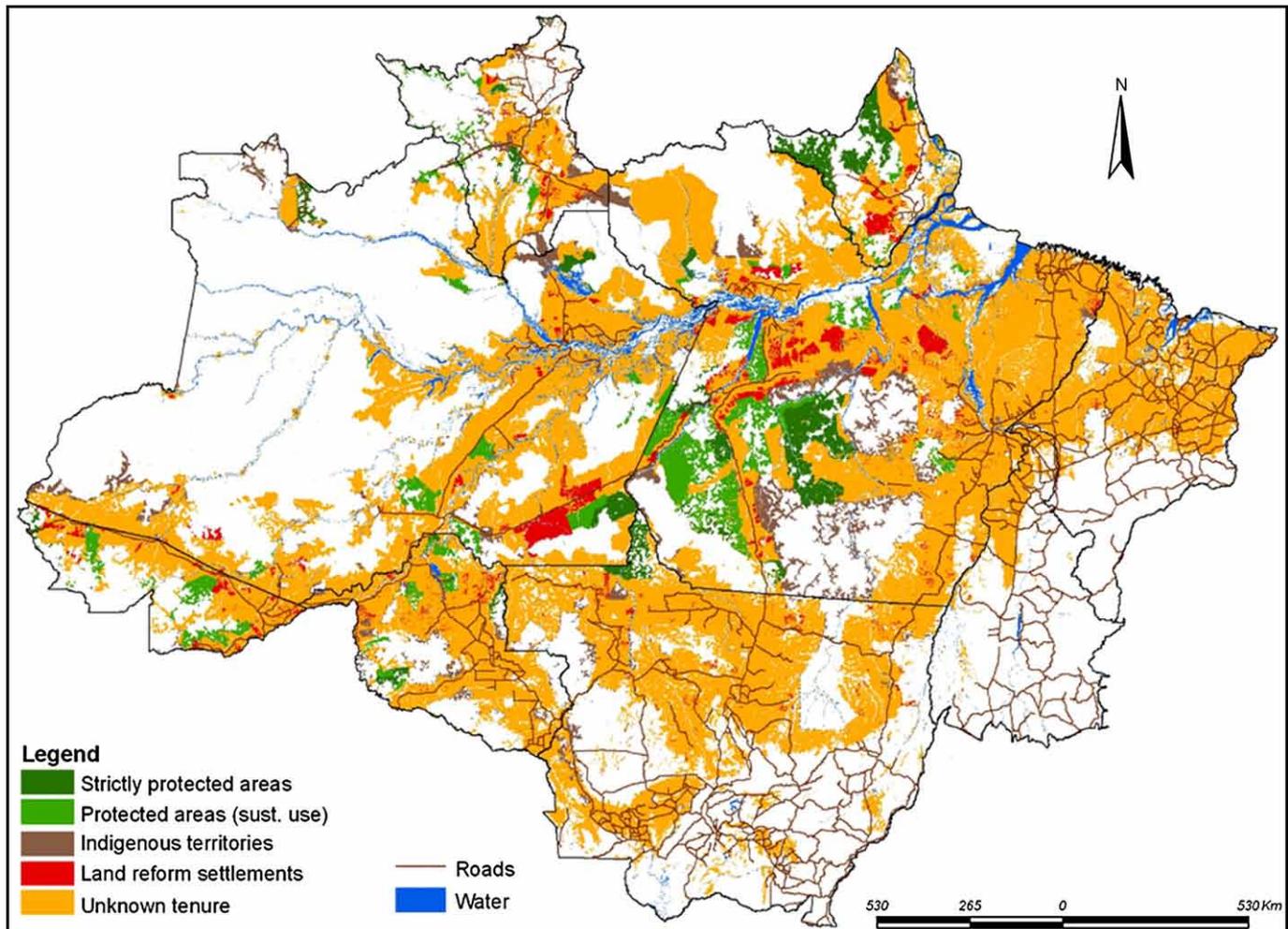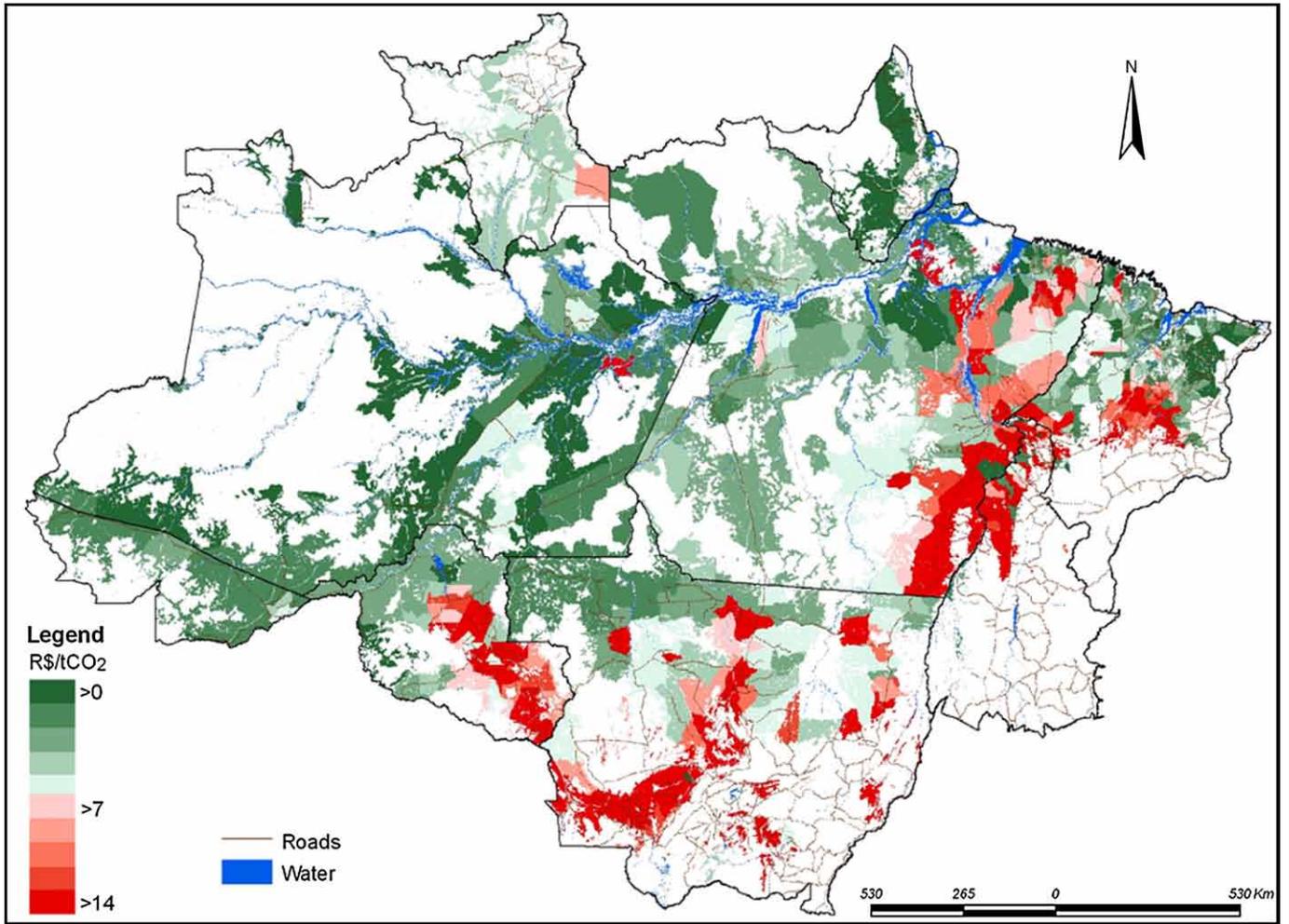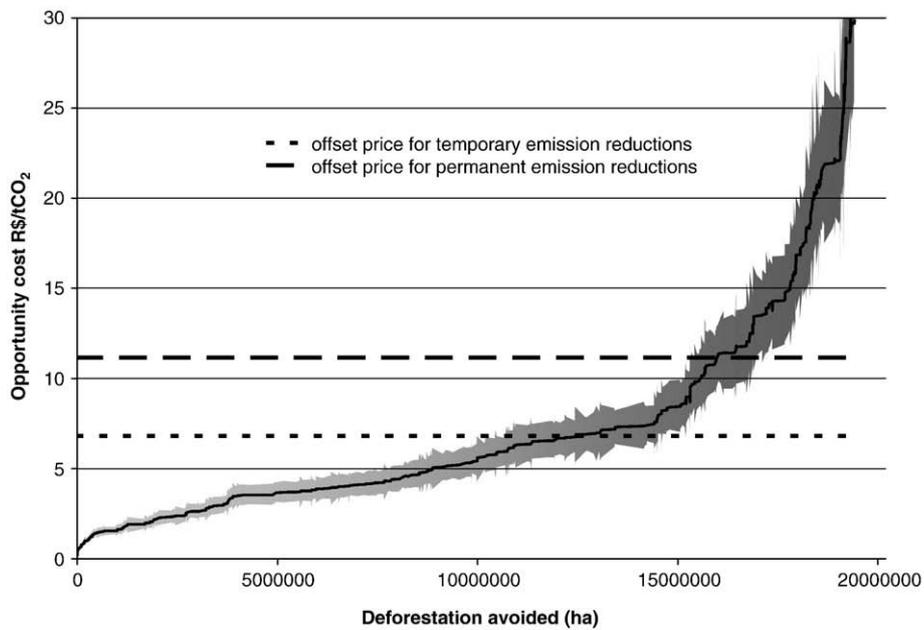

Fig. 2. Land tenure in areas threatened by deforestation until 2050.
Sources: Soares-Filho et al. (2006), INCRA 2007, IBAMA 2007.

Fig. 3. REDD opportunity cost in areas threatened by deforestation until 2050. Green areas = competitive REDD options; red areas = excessive opportunity costs for REDD. Sources: Soares-Filho et al. 2006, IBGE-PAM/PPM/PEV 2000–6.

Fig. 4. Avoided deforestation cost curve (municipal averages) for 2009–18. Grey band = values within 5–95% sensitivity range.
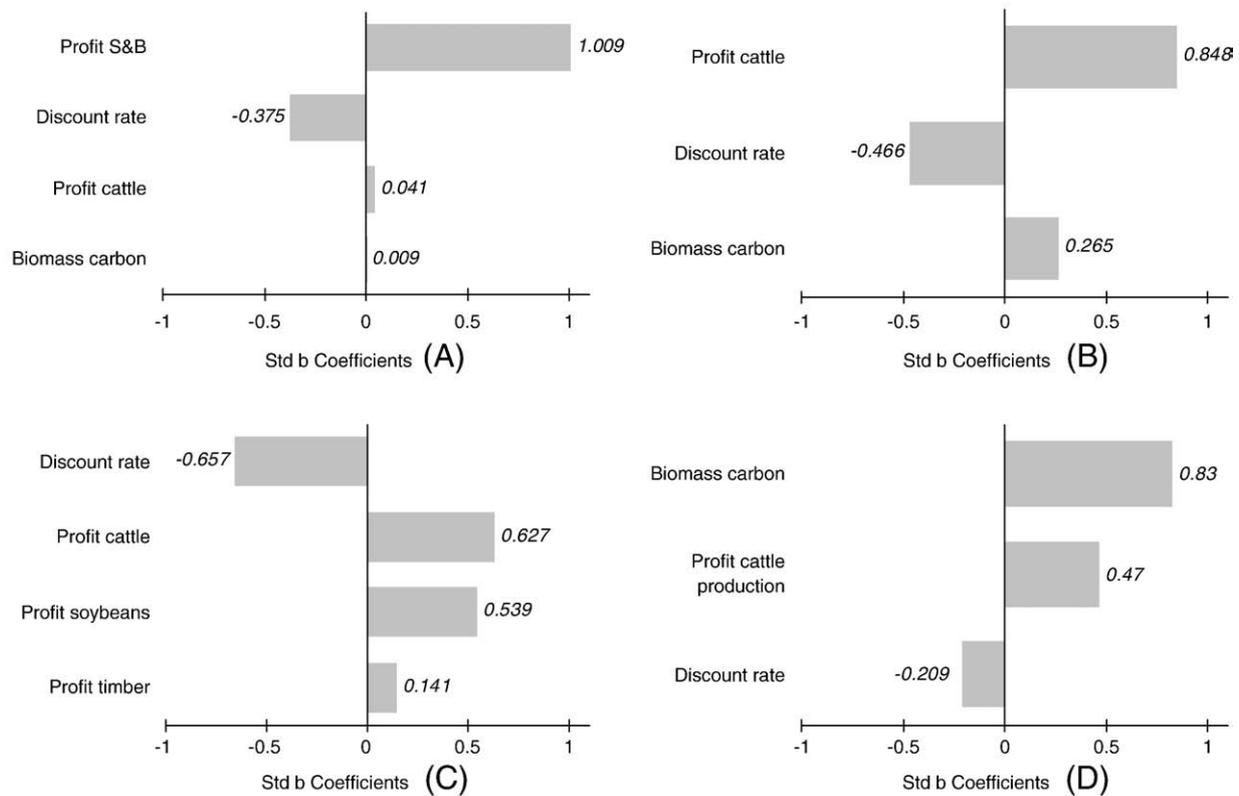
Fig. 5. Sensitivity of avoided deforestation cost curve at selected sections. Panel (A) depicts sensitivity at R$1.0/tCO$_2$, panel (B) at R$6.8/tCO$_2$, panel (C) at R$11.16/tCO$_2$, and panel (D) at R$20/tCO$_2$.

loss, with a confidence range of ± 2 million ha for variations in input parameters.

Fig. 5 sheds light on the sensitivity of selected cost-curve sections. At R$1.0/tCO$_2$[7] variations in opportunity costs are mostly affected by returns to slash-and-burn (S&B) farming, indicating that smallholders can offer avoided deforestation at low cost. The same holds for extensive cattle production (both large- and smallholders), the returns to which heavily influence variations right up to the price level of the voluntary carbon market. In other words, ranching and S&B are the two dominant economic activities that can be bought out at current prices. At higher levels, opportunity costs become more sensitive to returns from timber, soybean and perennials production, and here PES will currently fail to avoid deforestation. Towards the curve's upper end, forest-carbon content causes most variability.

Apparently about one third of projected deforestation until 2050 will occur in tenure categories that *ex ante* rule out PES, i.e. strictly protected areas and unclassified public lands. The exact amount cannot be determined because, as discussed in Section 2, no reliable data exists for privately held tenure outside known public land tenure categories. About one quarter of projected deforestation is expected to occur in indigenous territories, sustainable use areas, and land-reform settlements; the rest is on private properties. On aggregate, the vast majority (81%) of potential REDD opportunities until 2050 appear to exhibit opportunity cost that are competitive for PES at current temporary carbon-offset prices on voluntary markets. Yet, in a ten-year REDD scenario on lands projected to be threatened during 2009–18, only slightly more than half of baseline deforestation could be 'bought out'. This is because the more time-remote deforestation (2018–50) also occurs on lands spatially more remote from roads and markets, where opportunity costs are currently much lower—at least until infrastructure is being expanded to further integrate them economically.

Revisiting the economic and institutional preconditions for PES feasibility set out in the introduction, the analysis provides important answers to the initial questions of where, under which circumstances, and at what cost PES could lead to cost-effective REDD. Estimated opportunity costs clearly suggest that most forest conversion in the Brazilian Amazon exhibits low per-hectare returns that could be compensated for, even at the most conservative carbon-offset prices. In that respect, our findings confirm previous studies that with different methods and data have arrived at similar conclusions (cf. Nepstad et al., 2007; Swallow et al., 2007).

Our analysis ignores both buyer and seller transaction costs of REDD actions, which will shift upwards the abatement-cost curve in Fig. 4. Tentative transaction-cost estimates for the state of Mato Grosso, for example, suggest that implementation, monitoring, and verification costs (ranging between 0.15 and 0.53R$/tCO$_2$) would not compromise the competitiveness of REDD supply (Börner and Wunder, 2008). Reviewing PES schemes around the world, Wunder et al. (2008) found similarly low levels of operational transaction costs, but noted that start-up costs of PES schemes were occasionally high. Nonetheless, with operational transaction costs accounting for less than 10% of PES transfers, economic preconditions should not represent a major bottleneck for PES as a REDD vehicle.

Institutional preconditions pose much severer limitations to the prospects of PES as a large-scale REDD implementation mechanism. Even if trust could be built between potential REDD providers and buyers, the main bottleneck is to identify eligible service providers who effectively control their land, and whose deforestation can be legitimately compensated. Existing Brazilian environmental legislation in principle requires private Amazon landowners to keep 80% of their property under natural or managed forests (*reserva legal*). Even though non-compliance is widely tolerated, compensating farmers for not deforesting their 80% mandatory reserve would, *de jure,* be non-additional, since this land from the lawmaker point of view should have remained forested in the first place. Unsurprisingly, current

[7] R$1–US$0.45 (10.2008).

reserva legal rules remain under political pressure, and different incentives for landowners with less than 80% forest to 'reestablish legality' are currently under discussion. Similarly, deforestation violating legally established land-use restrictions on public lands, such as indigenous territories, sustainable use areas and land-reform settlements, would not be *de jure* additional: it would be paying people to do what they are already legally required to. Hence, the wide gap between legality and reality has to be bridged before PES can be implemented on a larger scale in the Amazon. Still it is worth noticing that payments to subsidize farmers' costs for achieving legal compliance have pragmatically been used elsewhere, both in developed and developing countries (see Section 4).

In addition to the challenges posed by *de jure* versus *de facto* discrepancies, PES generally cannot be used to stop deforestation on illegally appropriated public land (*terra devoluta*), nor on lands where private tenure is disputed: in both cases, no legitimate land steward can be paid. Legal landownership and illegal occupation often cannot easily be separated in practice (Benatti et al., 2006). In addition, while all public tenure categories except strictly protected areas grant exclusion rights to their dwellers, *de facto* many traditional, indigenous, and land-reform settled populations lack the control and government support to effectively prevent invasions by powerful commercial interests (Ferreira et al., 2005; Pedlowski et al., 2005). This may eventually be a more serious problem for REDD than *de jure* restrictions.

Pre-existing ill-enforced environmental legislation, undefined tenure and tenure insecurity therefore currently limit PES feasibility, in principle to only about one third of deforestation, as projected until 2050. In addition, with our data we cannot assess the degree of invasions into publicly delimited land categories. But even assuming each household living in indigenous lands and sustainable use areas hypothetically started to deforest at the same pace as smallholders, i.e. on average two hectares annually (Vosti et al. 2002) during 2009–50, this internal deforestation would explain only between 20 and 30% of all projected forest loss: the bulk of the pressure on these lands necessarily comes from outsiders. Hence, paying local forest stewards will *per se* only have limited additionality, unless it is accompanied by substantial command-and-control improvements and empowerment to effectively fence off external intruders.

3.2. PES Modalities and Benefit Distribution

Leaving aside for a moment these specific land tenure limitations: if PES was implemented with REDD funds on a large scale, how would benefits come to be distributed among different groups of land stewards? Using "temporary offset" prices as our base, net benefits—defined as compensation minus opportunity (and other) costs (Section 2.2)—would crucially depend on the adopted payment modality: (a) per-ton of carbon payments, (b) per ha payments, or (c) quasi-auction.

Under (a), all land users receive the upper-limit price for each ton of carbon-stock, regardless of their factual opportunity costs; hence the more to the left lower-cost end of the REDD supply curve, the larger will net benefits ("rents") be. Under (b), REDD buyers determine a fixed per-hectare payment that covers sellers' opportunity costs until they break-even with the market price. Finally, under (c) farmers are subjected to some procurement auction (Ferraro, 2008) that will reveal their approximate opportunity costs, but are assumed to only accept payments that exceed costs by 10%.

Fig. 6 summarizes potential net gains for each payment modality and tenure category, in a 2009–18 PES scenario (Section 3.1). In all cases, large landholders reap the highest absolute benefits (>55%). This has to do with the simple fact that they undertake about four fifths of all deforestation; our spatial analysis showed that 80% of 2002–06 forest loss in non-publicly demarcated tenure categories occurred in large polygons above 20 ha. Land-reform settlements, indigenous territories and sustainable use areas receive equally moderate shares, while the remainder (15–20%) accrues to small-holdings. This net-benefit distribution directly reflects the patterns of projected forest loss across the five tenure categories (Fig. 2).

The "per-ton carbon payment" modality (a) produces the clearly largest net benefits for REDD providers, because it massively over-compensates cheap suppliers' provision costs, especially in North-Western market-remote areas with low opportunity costs but high carbon densities in relatively undisturbed humid forests. Large landholdings would capture over R$8 billion in net benefits, followed by smallholdings (R$2.9 billion) and public tenure categories (each ~R$1.1–1.3 billion). Leakage and other uncertainties may eventually reduce payouts to all landholders *vis-à-vis* market prices, the relative benefit distribution across stakeholders should be stable. With per-hectare payments (b), absolute net benefits for REDD providers are a staggering 55% lower. This somewhat surprising finding occurs because, other than per-ton carbon payments, per ha payments do not specifically favor the large number of carbon-rich landholdings with low opportunity costs: per-hectare prices thus come to function like a biomass-determined carbon price differentiation. Finally, in the "quasi-auction" scenario where payments are individually customized
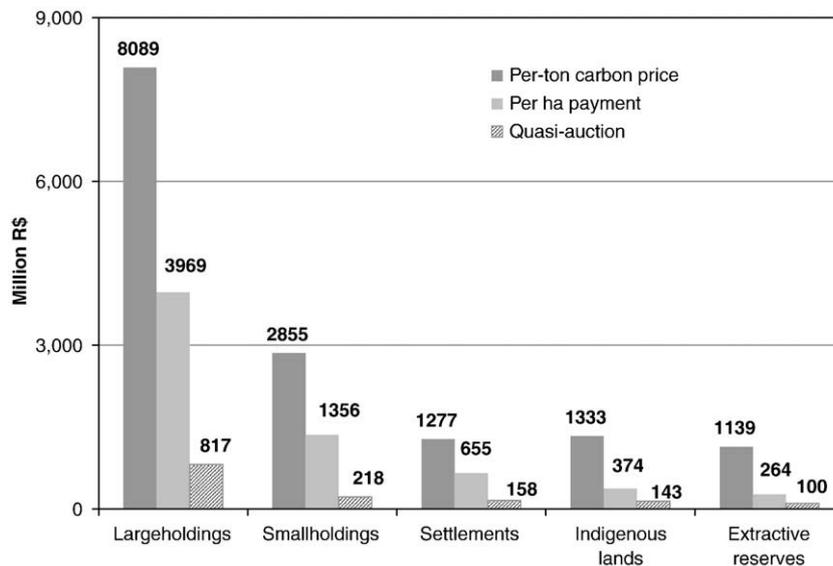
Fig. 6. Net benefits by tenure category and PES modality for 2009–18 (million R$).

to landowner opportunity costs (plus a premium) (c), REDD providers would lose up to 90% of their rents under (a).

From a REDD buyer's perspective, per-ton carbon pricing (a) costs on average R$2703 for each hectare protected, almost double providers' pure opportunity costs (R$1433/ha), because high rents are granted to low-cost providers. Switching to fixed per-hectare pricing (b) (R$1780/ha) or quasi-auctions (R$1556/ha) boosts PES cost-effectiveness by 34% and 42%, respectively. There is thus an inescapable tradeoff between PES supplier rents (i.e. recipient net benefits) and cost-effectiveness. However, "low-cost suppliers" means not only poor slash-and-burn farmers, but also large-scale cattle ranchers with extensive, low-return pastures. Paying fixed per-ton prices is thus not necessarily more pro-poor or equitable than choosing more competitive allocation mechanisms.

Which scenario is most likely to be adopted in practice? Many existing payment schemes, such as Costa Rica's, Ecuador's and Mexico's national PES programs use fixed or quasi-fixed per-hectare rates, for their administrative ease and seeming equity (Wunder et al., 2008). This is often criticized on efficiency grounds, as neglecting spatial heterogeneities in service provision, threats, and opportunity costs. One assessment of Costa Rica's national PES program found that multi-criteria spatial targeting would about double PES cost efficiency, at only 3.8%, more administrative costs (Wünscher et al., 2008). Surprisingly, our analysis suggests only modest efficiency gains by moving from per-hectare (b) to cost-targeted auction-type pricing (c). First, this could partially be an artifact from our much coarser spatial scale (municipalities instead of pixels). Second, our single-targeted Amazon service (carbon) is spatially more homogenously distributed than Costa Rica's four-service mix, reducing our service-targeting scope somewhat. But as mentioned, this paradox also occurs because opportunity costs and forest-biomass density happen to be negatively correlated in space, thus making the latter a reasonably good price-discriminating proxy for the former.

Maintaining for the moment our simplifying assumption that inhabitants of each land category are fully receiving compensations of "their" avoided deforestation area (regardless of their *de facto* ability to exclude access), we could also use average population densities for each land category to assign per-capita net benefits (Table 3). As expected, large landholders would capture high net benefits (per-capita NPV of R$76,082). However, due to the low population density in public land categories, quite high per-capita values also result for those. At the extreme, every indigenous-territory dweller would receive a staggering per-capita NPV of R$80,611, which is higher than what large landholders would receive. Again, this reflects that most deforestation in public tenure categories is caused by external actors (Ferreira et al., 2005; Nepstad et al., 2006): the indigenous dweller would thus be paid for the opportunity cost of the squatter who was projected to illegally invade indigenous land. Hypothetically, indigenous families would have to increase current clearing more than twenty-fold to 'achieve' by themselves the forest loss projected for their territories.

These proportions illustrate that deforestation is not necessarily driven by the population within each tenure category, nor is it predominantly controlled by the land stewards currently occupying

the land. To avoid deforestation, it might thus make more sense to align local PES payments to the true (and much more modest) *internal-stakeholder* opportunity costs, while spending the remainder on measures that clarify and *de facto* stabilize land tenure, i.e. controlling third-party intrusions in the future. The predominance of illegal deforestation means that the results in this section have to be treated with caution, being mechanic projections of outcomes given certain tenure categories, not a carefully designed incentive-compatible scheme.

Most REDD buyers, be they public or private, will likely concentrate PES pilots on those REDD providers that boast spatially well-defined and fairly secure property and access rights. Public PES schemes may also be restricted by what forests are *de jure* additional—which is bound to be a restrictive limitation. Agrarian reform settlements and some sustainable use areas with low external pressure may be the lowest-hanging fruits for PES-led pro-poor REDD pilots in the near future. Privately well-delimited areas, e.g. in Mato Grosso state, will certainly also qualify, although one might often have to deal creatively with the 80% *reserva legal* rule, and accept that these payments will be more anti-deforestation than pro-poor.

Abstracting from the land tenure complexities, our analysis shows that halting deforestation in the Brazilian Amazon would come at significant costs to those (legally or illegally) involved in converting forests to other uses. Reducing deforestation to zero in the coming decade would imply more than R$44 billion in lost net benefits to the region's land users. Yet, for the majority of threatened land areas, compensating for avoided deforestation is economically fully within the possibilities of even the most conservative current carbon market prices in voluntary markets with temporary credits. What remains are the institutional complexities of whom to compensate and how, to make the reduction of Amazon forest loss and the compensation of its people a reality.

4. Conclusions and Discussion

As various contributions to this Special Section indicate, the equity implications of PES are contentious. Yet in fact, any effective conservation action implies potential equity tradeoffs: in no way is this feature specific to PES. Unequal land distribution in the Brazilian Amazon, for example, means that regulatory mandated reductions in deforestation will especially hurt smallholder families struggling to maintain soil productivity in traditional slash-and-burn systems on relatively small plots: they are often unable to comply with the Brazilian Forest Code's 80% rule without compromising their welfare (Börner et al., 2007). PES, an incentive-based, voluntary, and conditional conservation approach is designed as a more equitable alternative to regulation, by explicitly compensating landholders' opportunity costs. PES are also likely more cost-effective in achieving conservation, especially perhaps in remote areas where effective command-and-control implementation may prove overly expensive.

As our results show, PES in the Brazilian Amazon may result in high transfers to large landowners, simply because they are the ones doing most of the deforestation. Yet, if that achieves *de jure* and *de facto* emission reductions, perhaps this is a necessary evil: after all, climate change will under 'business-as-usual' produce a series of highly inequitable outcomes for global service users. Hence, mitigating climate change by the most cost-effective means may eventually also be highly equitable, if one takes a more holistic perspective. And when four fifths of a major environmental problem are caused by large landholders, then it is not unreasonable to expect that any prospective mitigating solution will necessarily have to provide some compensations to this stakeholder group for their losses.

Norgaard (2010-this issue) rightly identifies a series of Amazon development needs, such as access to credit, markets, and secure land tenure, as preconditions for a long-term transition to more equitable development pathways. Some, but certainly not all these measures will *per se* reduce forest loss. Nonetheless, few would disagree that current

Table 3
Projected forest loss (2009–18), population density, and per-capita net benefits (net present value) by tenure category in a quasi-auction scheme.

	Projected forest loss (2009–18) [ha]	Population density [individuals per ha]	Net benefits [R$/capita]
Smallholdings	1,722,091	0.063	1999
Large holdings	5,922,673	0.002	76,082
Settlements	1,059,422	0.008	18,667
Indigenous territories	886,475	0.002	80,611
Sustainable use areas	658,933	0.007[a]	21,714

[a] Extrapolated from those extractive reserves for which IBAMA has population data.

low-cost expansion of the agricultural frontier is the most important barrier to the adoption of more land-intensive production technologies in the Amazon. Often reformed technologies (e.g. more land-intensive cattle ranching) are available at affordable costs, albeit marginally less profitable than their more land-extensive, environmentally destructive alternatives. Helping to 'close the frontier' and embark on a less land-hungry Amazon development path is a desirable goal of REDD, which the recent international willingness to pay can help achieving. Our findings suggest that PES can be one realistic on-the-ground implementation tool, be it through Costa Rican-style government payments, or the sum of decentralized NGO-led and federal state programs. However, PES is unlikely to halt deforestation without integration into national-scale REDD strategies.

With respect to the major economic precondition, we found that roughly half of projected forest loss in the 2009–18 period (55%; 12.5 million ha) exhibits net returns that could be compensated for by payments reflecting current temporary carbon credits on voluntary markets. In particular, slash-and-burn agriculture (practiced by smallholders) and land-extensive pastures (primarily in the hands of large landowners) exhibit economically inferior land-use returns, the expansion of which could easily be contained by compensating for carbon benefits.

Dynamic effects from large-scale Amazon REDD interventions, e.g. on land, output, and carbon markets, are not captured in our static analysis. Projections of future REDD carbon prices in particular would inevitably be futile, given the uncertainties regarding supply and demand in a new and potentially huge market; this is one reason why we have adopted the highly conservative Chicago Exchange prices. Our multi-faceted sensitivity analysis, however, shows that such dynamics may increase as well as decrease REDD competitiveness vis-à-vis other mitigation options, depending on the general economic environment. In addition, at least some parallel impacts create systemic stabilizers. For instance, the recent global financial crisis has reduced both commodity and carbon prices, thus lowering both mitigation-cost curve and the carbon price line.

Could staggering transaction costs reverse the above image of cost-effective Amazonian REDD options? The rough scenarios analyzed in Börner and Wunder (2008) indicate otherwise (Section 3.1). A full-blown national-scale REDD program would certainly require substantial additional investments, e.g. in national carbon monitoring, policy and governance reforms, or command-and-control improvements. That the Brazilian government is increasingly allocating national funds to combating illegal deforestation in the Amazon, however, shows that such investments must not necessarily be borne entirely by carbon markets. Neither will PES recipients receive the full market price for carbon offsets, given the need to finance the national umbrella. But, they would also be unlikely to see all transaction costs deducted from their payments.

Rather than economic obstacles, the institutional preconditions represent the major bottleneck for PES development in the Brazilian Amazon. Probably only a fraction of all low-cost REDD opportunities could effectively be addressed by PES schemes, at least in the short run. Over 67% of projected forest loss is expected to occur on land with, at least at our aggregate scale, unclear or insufficient tenure information, thus impeding REDD buyers to associate deforestation threats with land stewards that could act as PES counterparts. Another 8% of future deforestation is expected to occur in strictly protected areas, where compensation of opportunity costs cannot be justified.

In any case, careful assessments of de facto effectiveness of local land-exclusion rights have to be undertaken prior to implementing REDD pilots. Our Section 3 made it painstakingly clear that a mechanical distribution of REDD revenues according to current landholder categories makes no sense. It could, as has been propagated (e.g. Nepstad et al., 2007), result in large income transfers to forest-dwelling people, which are desirable from an equity viewpoint. However, when projected deforestation pressure comes predominantly from outside,

the illegal settlers and land grabbers would have to bear most genuine opportunity costs. 'Paying their share' to forest-dwelling people on huge, weakly protected land areas certainly does not stop deforestation. Only actions that de facto delimitate their land tenure and effectively stop invasions can achieve that.

Clearly, a more disaggregated inspection of local opportunities than our municipal-level assessment is necessary, and surely upcoming REDD pilots in the region will also increase our knowledge about where PES potentially could fly. Analyzing tenure conditions at lower aggregation levels may also disclose more PES-viable options than we could identify here. In the short term, PES promoters might prefer concentrating their efforts on land-reform settlements, sustainable use areas, and indigenous territories. Wherever conditions in those tenure categories allow effective exclusion of illegal invasions, their comparatively well-defined and spatially delimited tenure situation represents favorable context conditions for PES. Private landholdings will be a target in those regions where land tenure is already well-articulated. The Brazilian government is currently undertaking reforms that will 'change the rules of the game', including efforts to regularize land tenure, which would also favor REDD-cum-PES implementation.

How to handle the large discrepancy between de facto and de jure conservation is a key dilemma. Not rewarding avoided deforestation below the current Forest Code's 80% forest-retention rule would certainly restrict PES scope, perhaps in particular for smallholders. In fact, other countries have intelligently circumvented this problem in doing PES. Costa Rica's national PES system combined compensations with a pre-existing prohibition of deforestation: payments would certainly be suspended for breaking the law, but would also in principle be contingent on other, incremental conservation actions, such as non-extraction of timber, firebreak establishment, etc. (Pagiola, 2008). In developed-country PES, e.g. the US-based Environmental Quality Incentives Program (EQIP), subsidizing farmers' costs of regulation compliance has also not been uncommon (Claassen et al., 2008). Creative incentives to landowners to stick (or return) to legality, while avoiding general moral-hazard problems, are clearly needed. Sometimes non-governmental intermediaries have more flexibility than the State to experiment with such innovative incentives.

A second key factor affecting PES scope and equity outcomes is the choice of payment modality. A per-ton of carbon payment system would hand over immense rents to cheap offset suppliers, although those will not always be the poorest. The huge cost-effectiveness gain of more closely aligning payments with opportunity costs is hard to ignore. Doing so not only buys more carbon, but also spreads resources on a greater number of recipients. Income distribution effects would thus depend on the reduction of average rents versus the increased number of beneficiaries. Even if procurement auctions were used to optimize payments for cost-effectiveness, absolute net benefits may still be substantial: R$158 million over ten years, for instance, for about half a million people living in agrarian reform settlements.

We conclude that there is still a long way to go before achieving large-scale REDDiness in the Brazilian Amazon. PES can constitute a promising addition to the certainly wider set of policy measures that are required to make REDD in the Brazilian Amazon effective and equitable. It does not, however, substitute for the need to make complementary REDD investments, such as in creating new or better managed protected areas. Market enthusiasts hoping that economic incentives could completely replace command-and-control measures will have to realize that PES incentives crucially depend on minimum governance thresholds, including notably the regularization of land tenure and the control of access rights.

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