The Financial Costs of REDD:

Evidence from Brazil and Indonesia

Nathalie Olsen and Joshua Bishop

Full report

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The Financial Costs of REDD: Evidence from Brazil and Indonesia

Nathalie Olsen and Joshua Bishop, IUCN[1]

Executive Summary

Opportunities to mitigate climate change by reducing emissions from deforestation and forest degradation (REDD), especially in developing countries, have risen to the top of the international climate policy agenda, attracting increasing attention and investment from environmental organizations, development assistance agencies and the business community. Deforestation is one of the largest sources of global greenhouse gas (GHG) emissions and produces about 17 percent of total emissions (IPCC, 2007). There is growing consensus that REDD may offer a large pool of relatively low-cost emission reductions, which could significantly reduce the costs of meeting GHG reduction targets (see Beinhocker *et al.*, 2008; Stern, 2006; Grieg-Gran, 2008). However, proponents of REDD are still striving to win political endorsement for the approach as a compliance mechanism. The potential of REDD and other land-based carbon storage and sequestration opportunities as part of a post-2012 climate change regime remains uncertain, in part due to lack of detailed information on the likely costs associated with forest carbon projects, and REDD programmes in particular.

This desk study reviews the financial costs of abating GHG emissions through REDD from the perspective of an institutional investor seeking cost-effective abatement options. The objective is to investigate the main factors that determine the costs of REDD and to assess the range of likely costs in countries and regions where the potential to deliver significant abatement through REDD is greatest. As such, this review seeks to contribute to the current debate on the design and costs of REDD by focusing on field-level empirical issues and data and on financial, rather than economic, costs, i.e. actual costs to individual investors.

Analytical framework to assess costs of REDD
This paper looks at the two main cost components of REDD: (i) compensating the opportunity costs of forest conservation and (ii) implementation and transaction costs. There is a large degree of variation both within and between countries with regard to the opportunity costs of forest land, depending on the direct and indirect drivers of deforestation, and the carbon content of forests. As a result, on-the-ground estimates of opportunity cost vary according to local conditions and land use, and are often significantly lower than estimates produced using global economic models. This paper attempts to provide an analytical framework to quantify the full financial costs of REDD, in order to facilitate private and public sector investment.

[1] The authors gratefully acknowledge support from Rio Tinto for the preparation of this report. The views expressed in this report to not necessarily reflect those of IUCN or Rio Tinto. The authors remain responsible for any errors.

Opportunity costs of forest land

Compensating governments and/or land owners for the opportunity costs of conserving forests is likely to be the largest single cost component of any REDD scheme, assuming it is paid. The opportunity cost of forest conservation may be defined as the net income per hectare per year or net present value (NPV) that is sacrificed as a result of *not* logging (or logging more sustainably) or *not* converting land to agriculture. Opportunity cost is thus the profit gained from continuing 'business as usual'. Opportunity costs vary according to the drivers of deforestation in a specific region or country (for in-depth discussion of drivers of deforestation see UNFCCC, 2007; Geist and Lambin, 2002).

Grieg-Gran 2006 summarises the factors that affect the opportunity costs of REDD, including methodological issues such as:

- How timber harvesting and land clearing costs are treated;
- What type of forest land is considered;
- How alternative land uses are modelled;
- Which carbon density estimates are used; and
- Whether cost curves or point estimates for carbon abatement are calculated.

And economic, social, and geographical/physical factors, such as:

- Primary commodity prices;
- The suitability of particular forest lands for different uses;
- Soil and climate conditions which affect yields and hence returns to agriculture;
- Scale of operation – small, medium, large;
- Inputs and technology;
- Distance from market and the quality of transport infrastructure.

The case studies of Indonesia and Brazil presented below examine some of these factors for which data is available.

This paper reviews empirical evidence of the per hectare financial returns (US$/ha) to alternative land uses in Indonesia and Brazil. Per hectare returns are converted to returns per ton carbon (US$/ton C) based on local or regional estimates of carbon content of forests. It is assumed that GHG emissions from deforestation equal the total carbon content of above-ground vegetation, expressed as tons CO_2 equivalent (CO_2e), i.e. returns per ton carbon (US$/ton C) are converted to returns per ton CO_2e (US$/ton CO_2e) using the standard conversion factor of 3.67. Expressing opportunity costs per ton of CO_2e enables comparison with other climate mitigation options and with prevailing carbon prices. All net present value estimates of opportunity costs have been converted to 2005 US dollars, for ease of comparison.

Opportunity costs are generally treated as the most important cost component of REDD, and are often the only costs estimated given current uncertainty regarding the future design of REDD and hence the costs associated with implementation and transactions. This paper reviews available estimates of implementation and transaction costs from a range of studies in order to identify a rough global estimate.

Implementation and transactions costs

The second major component of the costs of REDD is implementation and transaction costs. This paper considers the costs associated with search, negotiation, verification, certification, implementation, monitoring, enforcement and insurance. Implementation costs are affected by economies of scale and vary depending on whether REDD policies and measures are national or project-based. Implementation and transaction costs are expressed in terms of cost per ton CO_2e and added to opportunity cost estimates (despite the fact that this may not be practical for project implementation which would operate on a per hectare basis).

Empirical estimates of the implementation and transaction costs of REDD presented here are based on experience with Payments for Ecosystem Services (PES) projects, other climate change mitigation projects, simulations and the observed costs of implementing Sustainable Forest Management (SFM)[2]. Recent studies suggest that there are significant economies of scale and that large projects and programmes have lower implementation costs per unit of emissions avoided (measured in tons CO_2e). Transaction costs, on the other hand, are likely to be more fixed than variable. Experience suggests that transaction costs will be greater for smaller projects than for larger projects and also greater for a large number of small transactions versus a smaller number of larger transactions (Börner and Wunder, 2008a).

Distributional issues and implications for costs

The distribution of the cost and benefits of REDD amongst different stakeholders will affect the net cost (and ultimate success) of investments in REDD. To date, the distribution of benefits of most PES schemes has been characterised as neutral, at best, with respect to poverty. For example, small landowners and the poor may be marginalised from PES due to high implementation and transaction costs, poorly defined land tenure, and lengthy, complicated administrative procedures. Moreover, there may be an equity-efficiency trade-off; for example, investment in the Clean Development Mechanism (CDM), has tended to focus on low cost emissions reductions, through the adoption of cleaner technologies in China and India, with relatively limited benefits for local people. Although the unit costs of carbon abatement via REDD would most likely increase with efforts to integrate equity and poverty concerns, these increased costs need to be met in order to ensure the delivery of project or programme outputs – indeed this expenditure is likely to be highly cost-effective.

This study adopts US$1/ton CO_2e as a rough global estimate of implementation and transaction costs. This estimate is derived in Boucher (2008) and is based on the aggregation of sub-sets of implementation and transaction costs from a range of studies: Antinori and Sathaye's (2007) estimate of transaction costs (US$0.38/ton CO_2e), Nepstad *et al.*'s (2007) implementation cost estimate (US$0.51/ton CO_2e) and Grieg-Gran's (2006) highest administrative cost estimate (US$0.04/ton CO_2e) to derive a total of US$1/ton CO_2e. While there is some overlap in the components of this sum, a small degree of double-counting ensures that the estimate is "conservative". In parallel, in a 'back of the envelope' calculation, Sohngen (2008) calculates the potential transaction

[2] See Wunder and Alban, 2008; May *et al.*, 2004; Cacho *et al.*, 2005; van Kooten, 2008; Antinori and Sathaye, 2007; Grieg-Gran, 2006; Boucher, 2008.

costs of REDD based on the budget of the USDA Conservation Reserve Program (CRP) and, coincidentally, also estimates these costs at about US$1/ton CO_2e. While these costs are not negligible, they are likely to be significantly smaller than the opportunity cost component of most REDD programmes.

The Brazilian Amazon

Brazil is responsible for approximately half of annual global deforestation (Hansen, 2008) and is the second largest emitter of GHG from deforestation, accounting for roughly 2.5 percent of global GHG emissions. Roughly 75 percent of Brazil's GHG emissions are from deforestation in the Amazon, and represent 8-14 percent of global emissions from land-use change.

The main direct drivers of deforestation in the Amazon are cattle ranching, soybean monoculture and logging. Empirical estimates of opportunity costs per hectare are low and the studies reviewed here suggest that, at current carbon prices in both voluntary and compliance markets, carbon sequestration can compete with most prevalent land uses in the Amazon. Expressed in terms of cost per ton CO_2e, the opportunity cost of cattle-ranching ranges from zero for traditional pasture and small scale ranching to US$2/ton CO_2e for ranching on improved pasture. As roughly 80 percent of recently deforested land is used for ranching, the scope for achieving cost-effective reductions in CO2emissions through avoided deforestation seems promising.

REDD is somewhat less competitive with soybean production, which has opportunity costs ranging from US$2.5 to US$3.4/ton CO_2e. Nepstad *et al.* (2007) calculate that to eliminate deforestation completely in the Brazilian Amazon would cost US$1.49/ton CO_2e, but that to reduce deforestation to 94 percent of projected levels would cost only half that amount at US$0.76/ton CO_2e. The difference is largely attributable to the high opportunity costs of forgoing soybean production. Similarly, Vera Diaz and Schwartzman (2005) estimate the cost of eliminating deforestation at US$5.44/ton CO_2e including soybean production and US$2.34/ton CO_2e excluding soybean production. Adding an estimated US$1/ton CO_2e in implementation and transaction costs to the opportunity cost estimates reviewed above increases the total costs of avoided deforestation significantly. For the highest opportunity cost estimates reviewed, i.e. high-productivity timber harvest followed by ranching and soybean production, the addition of implementation and transaction costs increases the costs to US$7.1/ton CO_2e.

Indonesia

In 2007, Indonesia became the third largest emitter of GHGs globally. Roughly 85 percent of Indonesia's emissions are due to deforestation, forest degradation and forest fires. Indonesia is the world's top emitter of GHGs associated with the draining of peatlands which is responsible for over five percent of annual global emissions of GHGs from human activities (Joosten 2009). The rate of deforestation has increased in recent years from 1.61 percent per year (1990–2000) to 1.91 percent per year (2000–2005) while the annual loss of primary forest has increased by 25 percent over the same period. Clearly, Indonesia's extensive tropical forest cover is threatened with rapid degradation and conversion.

The direct causes of deforestation and forest degradation in Indonesia include logging for timber, the establishment of large-scale tree crop estates and industrial timber plantations, smallholder farming, internal migration and government-sponsored resettlement. Industrial timber plantations mainly supply the pulp and paper industry. Rising commodity prices have accelerated the conversion of forest for the production of cash crops, notably palm oil. Illegal logging is a significant problem while forest fires destroyed over five million ha of forest in 1994 and another 4.6 million ha in 1997–98.

There are also important indirect drivers of deforestation in Indonesia. Over the last few decades, rapid economic growth has seen the emergence of a powerful class of landowners whose interests are often in conflict with small-scale land users (Swallow *et al.*, 2007). The fall in value of the Indonesian currency during the Asian financial crisis in 1997 provided additional incentives to convert forest to export tree crops, such as oil palm, rubber, cocoa and coffee. Competition between migrants, indigenous people and large investors accelerates deforestation on islands with greater population density, e.g. Sumatra. Logging is a powerful driver of forest degradation on other islands, e.g. Kalimantan.

Including incentives to reduce forest degradation in REDD is particularly important for Indonesia, where forest degradation may be a larger source of GHG emissions than forest conversion. Indonesia contains one-half of the world's tropical peatlands, which are extremely rich in carbon. In recent decades, these ecosystems have been widely cleared and converted to oil palm, fast-growing tree plantations for the pulp and paper industry, large scale irrigated rice and small scale agriculture. Large carbon emissions occur when peatlands are burned or drained. In a study of three provinces (East Kalimantan, Jambi, Lampung), Swallow *et al.* (2007) find that the economic returns from conversion of peatlands are very low, while carbon emissions are very high. The conservation of peatlands is thus a very low opportunity cost carbon abatement option, which has attracted wide attention as a priority for REDD investment.

As opportunity cost estimates per ton of CO_2e are highly sensitive to estimates of the carbon content of forest, this paper uses data from both the Alternatives to Slash and Burn (ASB)study (high carbon scenario) and the FAO Forest Resource Assessment (FRA) (low carbon scenario). Based on data from the ASB study, returns to land are adjusted by the net change in carbon storage per hectare that results from land use change. The net change in carbon is defined as the difference in the carbon content of undisturbed forest and the carbon content of the alternative land use (i.e. slash and burn farming) for the for high carbon content scenario. However, for the low carbon content scenario, carbon content data on alternative land uses is not available. In this case, it is assumed that the carbon content of alternative land uses is zero.

The highest opportunity cost of REDD in Indonesia occurs where forest conservation competes with palm oil production. Opportunity costs range from US$0.49/ton CO_2e for small holder farming in Sumatra up to US$19.6/ton CO_2e for conversion of degraded forest land to palm oil. Most palm oil production generates returns equivalent to US$3–7/ton CO_2e. Logging (unsustainable) is the next most profitable land use. Assuming a carbon content of undisturbed forest of 300 ton/ha, opportunity costs range from US$1.65/$CO_2e$ for commercial logging in Sumatra to US$3.44/ton CO_2e for unsustainable commercial logging in Southeast Asia and the Pacific. Both subsistence agriculture and cattle ranching have low rates of return in Indonesia; expressed as costs per ton CO_2e, most estimates are close to zero (and negative in some cases) due to low per hectare returns and the low carbon content of these land uses.

The sensitivity of the results with regard to carbon content estimates of both undisturbed forest and the land use following deforestation underscores the need to assess carbon stocks at a local level as there is significant variation within forests. Adding US$1/ton CO_2e in implementation and transaction costs to the opportunity cost estimates reviewed here increases the costs of avoided deforestation significantly. However, the cost of abating carbon emissions based on REDD in Indonesia remains below US$10/ton CO_2e for most land uses and below US$5/ton CO_2e for many land uses.

The Costs of REDD versus other carbon abatement opportunities

Empirical evidence on the financial returns to alternative land uses on recently deforested land in Brazil and Indonesia suggests that avoiding emissions from deforestation may provide a cost-effective climate mitigation option. The financial returns to a number of land uses, expressed in terms of net profits per ton of CO_2e, are below current market prices for carbon. In other words, forest carbon can provide attractive investment opportunities simply from a financial perspective. Moreover, due to the large variation in opportunity costs within forest-rich countries, there appears to be significant scope to achieve efficient outcomes by allowing trade in REDD obligations across land users, while focusing REDD interventions on avoiding the conversion of forest to low-return agricultural uses.

Table 11 summarises the range of estimates of the opportunity costs of REDD, based on different sources and methodologies. The estimates reviewed in this paper are compared to those provided in the Stern Review and those of global partial equilibrium models of the forest sector. Implementation and transaction costs are not included.

Table 11: Opportunity costs for REDD from different sources

Approach	**Land use**	**Opportunity cost estimate $/ton CO_2e**		
		Average	High	Low
Global models (various)		11.26	17.86	6.77
Stern Review (2006)		5.52	8.28	2.76
Regional, empirical (various)		2.51	4.18	0.84
This review:				
Brazil	Ranching		2.6	0
	Soybean		3.4	2.5
	Subsistence agriculture		1.1	0
	Timber+ranching+soy bean		6.1	3.9
Indonesia – high carbon scenario	Palm oil		4.29	0.18
	Subsistence agriculture		0.47	0
	Logging		3.44	1.65
Indonesia – low carbon scenario	Palm oil		19.6	0.5
	Subsistence agriculture		1.53	0
	Logging		7.96	3.82

Source: Adapted from Boucher (2008), including estimates from this review.

McKinsey & Company (2009) use global estimates to compare the cost-effectiveness of a range of carbon abatement opportunities across all sectors (see Figure 18). The estimates reviewed in this study (US$ estimates are converted into euro at the 2005 exchange rate of US$1=1.25 EUR) are consistent with McKinsey's estimates for reduced slash and burn agriculture and reduced pastureland conversion at less than EUR 5/ton CO_2e. These abatement options appear to be more cost effective than many non-forestry sector abatement opportunities such as solar energy, wind energy, carbon capture and storage, etc. Moreover, abatement based on reduced slash-and-burn agriculture and reduced pastureland conversion is more cost effective than all other forest sector abatement options, e.g. the restoration of degraded land, afforestation of pastureland, and reforestation of degraded forest. In line with this review, McKinsey find that the costs of abatement based on the reduced conversion of forest to intensive agriculture are higher and cannot compete with solar and wind power, for example. However, it must also be acknowledged that the data, and indeed many of the published estimates of abatement costs, do not measure risk consistently, i.e. the reliability of different abatement strategies.

There is a wide range of estimates of the costs of carbon abatement strategies in the forest sector and of REDD in particular. Much of the difference is due to the fact that micro-level estimates, based on particular local conditions, more accurately capture variation in local opportunity costs. This type of information is critical to guide public and private investors seeking to develop forest carbon projects and REDD activities in particular areas. For many stakeholders, global estimates and regional averages do not provide sufficiently accurate estimates of the relevant costs and risks.

The key findings of this review may be summarised as follows:

- There is significant variation in per hectare opportunity costs in Brazil and Indonesia, reflecting differences in local conditions, land use and proximity to transport infrastructure and markets. National, regional and global averages are of limited usefulness for determining where REDD is most cost-effective.
- There is significant variation in the carbon content of forest land at national, provincial and local level. Moreover, there is some inconsistency between published estimates of carbon content, based on the application of different methodologies. It is therefore essential to not only estimate local opportunity costs, but also to measure carbon content on a local basis.
- A review of empirical opportunity cost estimates suggests that REDD is competitive with most land uses in the Brazilian Amazon and many land uses in Indonesia at a carbon price of less than US$5/ton CO_2e. REDD is competitive with most land uses in Indonesia at US$10/ton CO_2e. Subsistence agriculture and most livestock production systems are characterized by very low returns in both Brazil and Indonesia. Logging and cash crops generally exhibit higher opportunity costs.
- While implementation and transaction costs add roughly US$1/ton CO_2e to opportunity costs, these additional costs are not so large as to make REDD (or other forest carbon activities) financially unattractive relative to non-forest sector carbon abatement options.

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1 Context, rationale and analytical framework

1.1 Introduction

Opportunities to mitigate climate change by reducing emissions from deforestation and forest degradation (REDD), especially in developing countries, have risen to the top of the international climate policy agenda, attracting increasing attention and investment from environmental organizations, development assistance agencies and the business community. Deforestation is one of the largest sources of global greenhouse gas (GHG) emissions and produces about 17 percent of total emissions (IPCC, 2007). There is growing consensus that REDD may offer a large pool of relatively low-cost emission reductions, which could significantly reduce the costs of meeting GHG reduction targets (see Beinhocker *et al.*, 2008; Stern, 2006; Grieg-Gran, 2008). However, proponents of REDD are still striving to win political endorsement for the approach as a compliance mechanism. The potential of REDD and other land-based carbon storage and sequestration opportunities as part of a post-2012 climate change regime remains uncertain, in part due to lack of detailed information on the likely costs associated with forest carbon projects, and REDD programmes in particular.

This desk study reviews the financial costs of abating GHG emissions through REDD from the perspective of an institutional investor seeking cost-effective abatement options. The objective is to investigate the main factors that determine the costs of REDD and to assess the range of likely costs in countries and regions where the potential to deliver significant abatement through REDD is greatest. As such, this review seeks to contribute to the current debate on the design and costs of REDD by focusing on field-level empirical issues and data and on financial, rather than economic, costs, i.e. actual costs to individual investors.

A number of studies on the costs of REDD attempt to estimate the forest area which could be conserved or the volume of CO_2 emissions which could be avoided given a fixed global budget, i.e. how much carbon can remain fixed in existing forests at a carbon price of US$X /ton or how many tons of CO_2 equivalent (CO_2e) emissions can be avoided for a global budget of US$X million? The debate on REDD within the UNFCCC is working to determine the costs of nationally appropriate mitigation actions (NAMAs) for REDD that are measurable, reportable, and verifiable (MRV). However, micro-level analytical studies focusing specifically on national, sub-national and project level costs of REDD are not common; this paper attempts to begin to fill this gap by proposing a simple framework and reviewing data available for Brazil and Indonesia.

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The paper reviews empirical work which suggests that the costs of REDD lie in a range from US$2–10 per ton CO_2e, including implementation and transaction costs. As a portion of the market for voluntary carbon offsets, REDD and related projects are becoming more significant. In 2007, more than two million tons of CO_2e were generated from avoided deforestation projects at an average price of US$4.80 ton/ CO_2e (Johns and Johnson, 2008). For comparison, the price of emission allowances in the EU Emission Trading Scheme, in October 2008, ranged between EUR 18–25 (US$ 23–33) per ton CO_2e (Ecosystem Marketplace, 2008). Finally, compared to the cost of cutting industrial emissions, which can exceed US$50 per ton CO_2e, the costs of REDD seem quite competitive. In short, avoiding deforestation appears to provide cost-effective opportunities to reduce GHG emissions, particularly when forest land with the lowest opportunity cost is conserved.

Section 2 of this paper sketches out an analytical framework to assess the costs of REDD in different countries, provinces or project areas. Sections 3 and 4 apply this analytical framework to areas of Brazil and Indonesia. Section 5 compares estimates of the costs of carbon abatement based on REDD with other forest sector and non-forest sector carbon abatement opportunities.

1.2 Scope of the paper

Before turning to the costs of REDD, it is useful to define more clearly what REDD is and for which elements of REDD this study attempts to estimate costs. This section looks more closely at REDD at various levels (international, national, sub-national) and some important design elements (scope, leakage and additionality) which will determine whether forest carbon projects qualify as REDD projects, i.e. whether they meet criteria that are likely to be included in a future REDD regime.

Project-level REDD activities (and the associated costs) will be influenced by and must be consistent with national, sectoral or policy based REDD activities. Policy based REDD programmes focus on how land use policy affects deforestation. For example, agricultural subsidies and the expansion of transportation infrastructure encourage deforestation through the increased profitability of agriculture and logging. Reforming land use policies could potentially free up financial resources previously used for subsidies; if reform of land use policies were specifically linked to REDD and if by reducing or eliminating the subsidy the pressure on forests is reduced, the potential net cost of REDD implementation could be negative. Sector-based REDD programmes are also being discussed and could produce carbon credits based on reducing net deforestation at the provincial or national level by committing to an emissions cap in the forestry sector.

REDD involves the creation of a multi-level Payment for Ecosystem Services scheme (see Figure 1, Angelsen and Wertz-Kanounnikoff, 2008). While this review focuses on the sub-national, an integrated project level and national level REDD approach is needed. At the international level, buyers purchase (in compliance or voluntary markets) from providers (countries or sub-national bodies or projects) an ecosystem service, in this case reduced emissions from deforestation and forest degradation. Measures likely to deliver this service, e.g. support for tenure reform or law enforcement, may also be paid for/supported (Angelsen and Wertz-Kanounnikoff, 2008). Discussions on the scale of

REDD focus on the accounting level of an international financing mechanism for REDD, but the level of implementation is less important as a REDD regime could include both nationally implemented projects and a national REDD strategy that credits projects implemented by others. Similarly, a sub/national project approach to implementation would benefit enormously from sound national REDD policies (Angelsen and Wertz-Kanounnikoff, 2008).

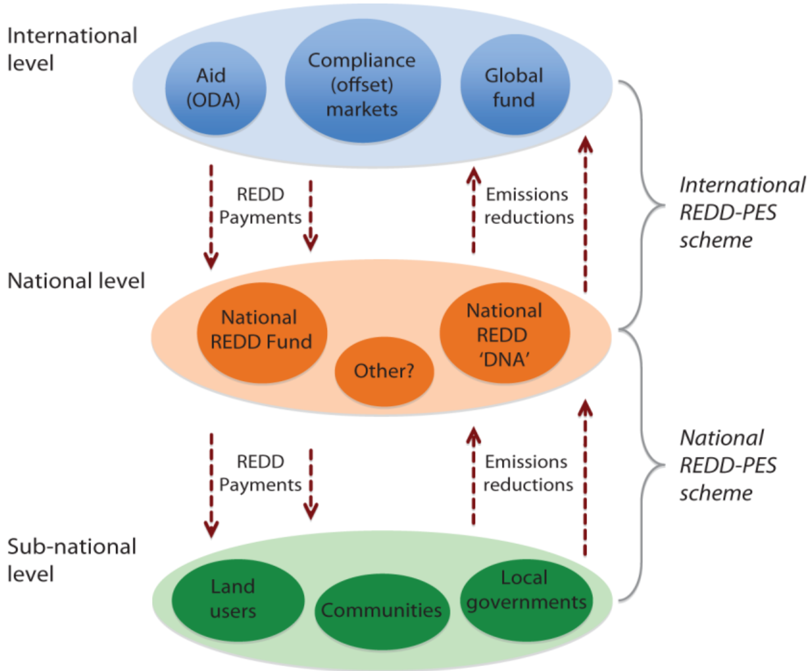


Figure 1: REDD as a multi-level Payment for Ecosystem Services scheme
Source: Reproduced from Angelsen and Wertz-Kanounikoff, 2008.

Within the UNFCCC, discussions surrounding REDD began in earnest in 2005 with RED, i.e. limited to deforestation only. Discussions expanded to REDD to include forest degradation and have since begun to consider forest conservation, sustainable forest management and enhancement of carbon stocks (REDD+). There is currently a general consensus that a REDD mechanism should cover all forests and only forests (Angelsen, Brown, *et al.*, 2009) although disagreement over how to define forests renders this consensus somewhat tenuous. There is still disagreement over whether there should be a primary set of measures addressing deforestation and forest degradation, and a supplementary set of measures for other forest-based mitigation options. It is also not clear whether Parties to the UNFCCC mean "enhancement of forest carbon stocks" to include only forest restoration on land already classified as forest or whether forestation on non-forest land will be included in this category. (Angelsen, Brown, *et al.*, 2009). Afforestation and reforestation can be interpreted as the positive complement to avoided deforestation just as forest restoration is the positive complement to avoided forest degradation (Angelsen and Wertz-Kanounikoff, 2008).

Table 1: Possible scope of mitigation activities to be included in REDD mechanism

Changes in:	Reduced negative change	Enhanced positive change
Forest area (hectare)	Avoided deforestation	Afforestation and reforestation (A/R)
Carbon density (carbon per hectare)	Avoided degradation	Forest restoration and rehabilitation (carbon stock enhancement)

Source: Reproduced from Angelsen and Wertz-Kanounikoff, 2008

With regard to the scope of REDD, this paper focuses primarily on the cost of avoided deforestation and avoided degradation as this is the subject of most of the reviewed micro-level studies. Unfortunately, it is not always clear in empirical studies whether it is the costs of avoided deforestation or degradation that are being estimated. In this review, opportunity costs are generally estimated based on the conversion of primary forest to agricultural uses. Forest degradation is covered in a limited manner for Indonesia in the discussion on the conversion of peatlands to agriculture.

While forest carbon projects may fall within the scope of REDD, there are criteria which will determine the eligibility of forest carbon projects for REDD financing. *Leakage* is an important issue in current discussions and refers to the emissions displacement that occurs when interventions to reduce emissions in one geographical area cause an increase in emissions in another area through the relocation of activities. Table 2 outlines potential leakage channels for different project types (AR, REDD set aside conservation, REDD-SFM). Leakage is a significant problem for project-based activities as they are local in area. In general, as the scope and scale of REDD interventions increases, opportunities for leakage decrease (Angelsen and Wertz-Kanounikoff, 2008). To qualify for REDD, it is likely that projects must demonstrate measures to successfully manage leakage. This may be done through internal measures, e.g. by increasing the project area to create buffer areas and, depending on the form of leakage, provide alternative livelihood earning opportunities or other forms of compensation. Alternatively and perhaps more likely, project based activities will operate in the context of national or international REDD programmes which provide the institutional infrastructure for monitoring, reporting and verification at a scale the addresses problems of leakage. It is unlikely that project based REDD-like activities can succeed in the absence of a higher level REDD programme.

Table 2: Potential leakage channels for REDD

Project types / *Leakage channels*	Afforestation and Reforestation	REDD – Set-aside Conservation	REDD – Sustainable Forest Management
A. Land markets	Substituting crops/ livestock through plantations	Curbing planned agricultural land conversion	Curbing planned agricultural land conversion
B. Labour markets	Labour-using initially; later variable	Less employment may cause out-migration	Labour-saving, or Labour-using shift?
C. Capital markets	Returns may attract capital	Crowding out effects from lower returns	Impact on returns disputed
D. Technological innovation	Variable	None (unless combined with ecotourism, non-timber forest products)	Reduced impact logging, etc.
E. Output markets	Planted forest products (medium run) reduce extraction pressures	No agricultural or timber supply from set-asides	Less timber (short to medium run)
F. Income generation	Variable	Variable	Variable
G. Ecological conditions	Plantations increase or decrease ecological integrity (pests, wind, biodiversity, etc.)	Increase in landscape integrity and adaptation, avoided 'edge effects'	Increase in landscape integrity and adaptation, avoided 'edge effects'

Source: Reproduced from Angelsen and Wertz-Kanounikoff, 2008

In addition to leakage management, REDD programmes will almost certainly include a criterion for *additionality*. Additionality requires that evidence be provided that long term, measurable GHG emission reductions would not have occurred in the absence of the project, policy or activity. This implies that those who have already invested in conservation of forest land or restoration of degraded forest land will not be eligible for inclusion in a REDD programme. This may have implications for private investors, who will have to demonstrate a clear and verifiable threat to forest land selected for REDD-relevant project activities.

In summary, due to issues of scope, leakage and additionality, many investors in forest carbon may find that their activities do not qualify for REDD financing. The degree of restrictiveness of a future REDD regime is currently taking shape, but many of the details remain to be fleshed out. However, it is likely that there are opportunities for receiving carbon credits for forest carbon activities outside of UNFCCC REDD discussions as witnessed by the expanding volume and value of forest carbon in the voluntary carbon market. Because it is currently difficult to know the precise future requirements for inclusion of forest carbon project in a REDD regime, this paper refers to projects in the REDD pipeline as forest carbon projects or REDD-like projects rather than REDD projects. Forest carbon projects include a broad range of projects which generate carbon credits based on the maintenance of local carbon stocks.

1.3 Estimating the costs of REDD

The costs of abating a ton of carbon by reducing deforestation and forest degradation are not well-established because few REDD-relevant initiatives have been implemented. Most published estimates of the costs of REDD are based on simple models using data on the two main cost components of forest carbon schemes: the opportunity costs of forest conservation and implementation and transaction costs.

Pagiola and Bosquet (2009) define categories for the 'costs of REDD' and highlight the need to address the question of 'cost to whom?' They distinguish between (i) cost to the country, (ii) costs to individual actors, and (iii) budgetary costs to government agencies. This review focuses on costs to individual actors and therefore uses financial prices rather than economic (or 'social' prices). Pagiola and Bosquet (2009) emphasize the need to carefully estimate costs to different parties as, for example, costs and benefits to individual groups may not be costs to the country as one group's costs are another group's benefits. Moreover, identifying the distribution of costs assists in understanding incentives to deforest or degrade forest and thus may provide critical guidance in developing policies to reduce deforestation and forest degradation. Finally, consideration of the distribution of costs is important to understand potential impacts on stakeholders, particularly vulnerable groups.

1.3.1 Opportunity costs

Compensating governments and/or land owners for the opportunity costs of conserving forests is likely to be the largest single cost component of any REDD scheme, assuming it is paid. The opportunity cost of forest conservation may be defined as the net income per hectare per year or the net present value (NPV) that is sacrificed as a result of not logging (or logging more sustainably) or not converting land to agriculture. Opportunity cost is thus the profit gained from continuing 'business as usual'. Opportunity costs vary according to the drivers of deforestation in a specific region or country (for in-depth discussion of drivers of deforestation see UNFCCC, 2006; Geist and Lambin, 2002). While these vary, there are regional trends based on shared social, economic and environmental conditions (Grieg-Gran, 2006).

- In Africa, population pressure and agricultural expansion combined with insecure property rights are the predominant causes of deforestation.
- In Latin America, agricultural expansion by medium and large scale operations is encouraged by current land use policies.
- In Asia, current policies favour large scale logging, the expansion of cash crop estates and the expansion of smallholder agriculture.

Recent studies have documented a shift from small-holder driven deforestation to more large-scale, corporate based deforestation across regions.

In published reviews of the literature on opportunity costs of forest land in the tropics different approaches are taken to estimate returns per hectare under different types of land use and conditions (see Grieg-Gran, 2006; Boucher, 2008). These methods are discussed briefly below.

A first approach derives local/micro-level estimates based on empirical data on per hectare opportunity cost estimates (US$/ha). These opportunity cost estimates are specific to a region and reflect local conditions and costs; as a result, these estimates vary significantly across localities and regions. A second approach is to derive estimates based on generic or average production costs per hectare or per ton agricultural output. Average costs may be calculated on the basis of data from other countries, although these estimates do not take account of differences in local conditions or scale. This approach is used when local level data is not available. Third, land price estimates should in theory reflect the discounted stream of returns from the most productive use of land. However, due to the lack of clear property rights in many forest areas (e.g. Brazilian Amazon) and the absence of markets, land prices may not accurately reflect the present value of future profits (as economic theory suggests they should). Fourth, global partial equilibrium models simulate the relevant parts of the world economy, including the forest, agriculture, and energy sectors to estimate supply curves for emissions reductions. These global models generally produce higher estimates for the costs of reducing CO_2e emissions from deforestation (discussed in more detail in the results section).

1.3.1.1 Factors which affect opportunity costs

Compensating governments and/or land owners for the opportunity costs of conserving forests is likely to be the largest single cost component of any REDD scheme, assuming it is paid. The opportunity cost of forest conservation may be defined as the net income per hectare per year or the net present value (NPV) that is sacrificed as a result of not logging (or logging more sustainably) or not converting land to agriculture. Opportunity cost is thus the profit gained from continuing 'business as usual'. Opportunity costs vary according to the drivers of deforestation in a specific region or country.

Grieg-Gran (2006) summarises the factors that affect the opportunity costs of REDD, including methodological issues such as:

- How timber harvesting and land clearing costs are treated;
- What type of forest land is considered;
- How alternative land uses are modeled;
- Which carbon density estimates are used; and
- Whether cost curves or points for carbon abatement are estimated.

And also various economic, social and geographical/physical factors, such as:

- Primary commodity prices;
- The suitability of particular forest lands for different uses;
- Soil and climate conditions which affect yields and hence returns to agriculture;
- Scale of operation – small, medium, large;
- Inputs and technology;
- Distance from market and the quality of transport infrastructure.

The case studies of Indonesia and Brazil presented below examine some of these factors for which data is available.

This paper reviews empirical evidence of the per hectare financial returns (US$/ha) to alternative land uses in Indonesia and Brazil. Per hectare returns are converted to returns per ton of carbon (US$/ton C), based on local or regional estimates of the carbon content of forests. It is assumed that GHG emissions from deforestation equal the total carbon content of above-ground vegetation, expressed as tons of CO_2e, i.e. returns per ton carbon (US$/ton C) are converted to returns per ton CO_2e (US$/ton CO_2e) using a standard conversion factor of 3.67. Expressing opportunity costs per ton of CO_2e enables comparison with other climate mitigation options and with prevailing carbon prices. All net present value estimates of opportunity costs have been converted to 2005 US dollars, for ease of comparison.

1.3.1.2 Primary commodity prices

The prices of primary commodities are important determinants of opportunity costs as they are a key component of returns to alternative land uses. In general, the long-term trend for real non-fuel primary commodity prices has been downwards (Figure 2). However, in 2006 and 2007, primary commodity prices increased dramatically as the global economy experienced rapid and consistent growth rates and the demand for biofuels increased.

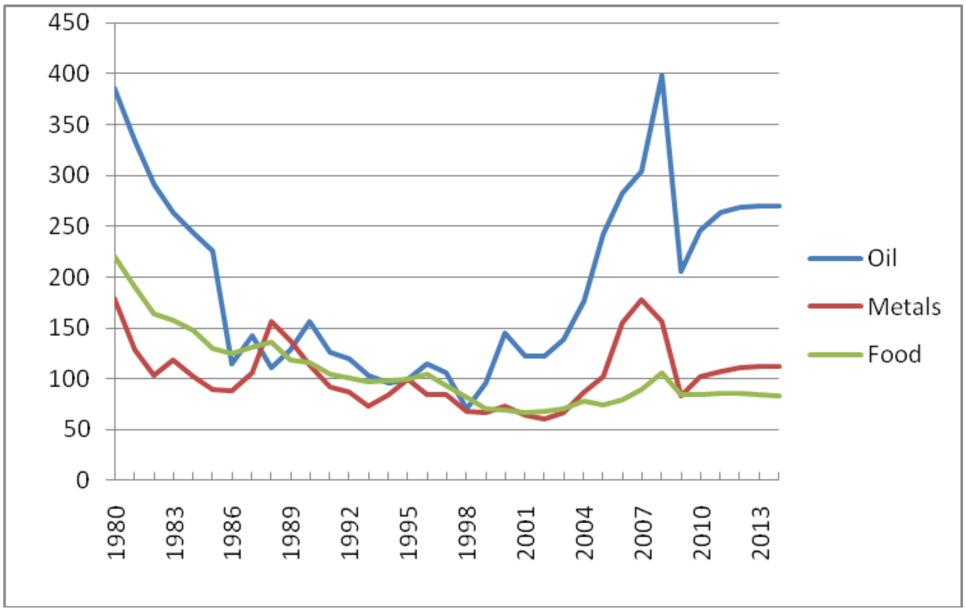


Figure 2: Real commodity prices (1995=100, 1980 to 2014)
http://www.imf.org/external/pubs/ft/weo/2009/update/01/index.htm accessed 02.11.09

In 2008 and 2009, the financial crisis followed by global recession drastically cut demand for raw materials and primary commodities. As a result non-fuel primary commodity prices in early 2009 have fallen dramatically and, as a whole, are only slightly higher than 2005/2006 levels.

However, the prices of some individual commodities have remained well above their historical average. For example, prices for palm oil and hard logs in early 2009 are still 40–60 percent higher than the average for 2005. Given the current highly uncertain economic environment, it is difficult to predict what these commodity prices will do in 2009 and beyond. The IMF (Figure 2) and the FAO forecast that real primary commodity prices will not increase significantly from the low levels of early 2009. However, price trends (Figure 3) and forecasts for individual commodities vary.

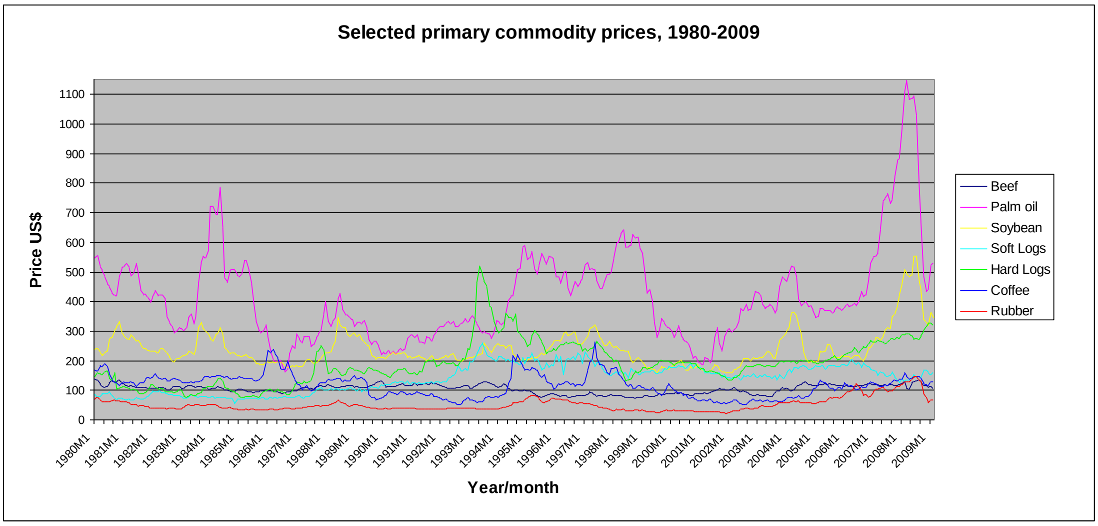


Figure 3 Selected primary commodity prices, 1980-2009 (nominal prices)
Source: Data from IMF primary commodity price database accessed March 2009

The opportunity cost estimates in this review are based on data collected between 1997 and 2008, expressed in 2005 US$. To take account of changes in commodity prices, opportunity cost estimates are adjusted. Certain commodities exhibit a high degree of price volatility (soybean, palm oil). In these cases, commodity prices in the year opportunity costs were estimated are compared to long term average prices. If there is a significant difference, i.e. 10 percent, opportunity cost estimates are adjusted by the percentage price difference (palm oil, beef, rubber). While beef prices are stable, the average 1997 price is roughly 20 percent lower than the long-term (2000–2009) historical average price. If prices at the time opportunity costs were estimated were higher than the long term average commodity price, no adjustment is made (soybean, logging). Estimates for non-traded sectors, i.e. subsistence agriculture were also not adjusted.

Future primary commodity prices will undoubtedly affect the costs of REDD. Investors face this uncertainty along with the risks associated with other uncertain factors which generally affect investment decisions, which involve decisions today based on current conditions and uncertain returns in the future. However, investments in REDD and other carbon abatement opportunities may actually be subject to less price volatility as governments may be more likely to take action to stabilise carbon markets or take indirect actions to support the carbon market than more traditional commodity markets.

1.3.1.3 One-off timber harvesting

When forests are converted, the harvest of timber has costs and benefits. If timber resources are commercially valuable (this depends on the species, quality and proximity to markets) there are significant net profits in the first year of forest conversion. However, timber resources may not be commercially valuable if the forest is already degraded, far from roads and buyers, or property rights are poorly defined and, as a result, timber cannot be harvested and sold without undue risk. In some cases the costs of converting forest to pasture may exceed the benefits of timber harvest. Whether one-off timber harvesting costs are included in estimates of the per hectare costs of REDD significantly influences the results.

Grieg-Gran (2006) addresses this issue in detail. Her estimation of the opportunity costs of land is based on three different scenarios: 0%, 100% and a country specific share of returns to one-off timber harvesting which have to be compensated as part of land conversion. The issue of whether to include timber harvesting and pasture establishment costs depends very much on country specific drivers of deforestation and the land use trajectory following deforestation. For example, studies in tropical Africa (Kotto-Same *et al.*, 2000; Osafo, 2005, cited in Grieg-Gran, 2006) do not include timber revenues because deforestation is primarily driven by smallholder agriculture. Because timber rights belong to the State and smallholders are only allowed to harvest for own use, timber is frequently burned rather than sold. A similar situation is found in parts of Indonesia (Tomich *et al.*, 1998, in Grieg-Gran, 2006) although, with the recent significant expansion of large-scale palm oil monoculture, smallholder agriculture may no longer be the main source of deforestation. On the other hand, in Brazil, timber harvesting is sometimes included (Börner and Wunder, 2008a; Nepstad *et al.*, 2007) and sometimes excluded (Börner and Wunder, 2008b; Vera Diaz and Schwartzman, 2005; Margulis, 2003 cited in Grieg-Gran, 2006) depending on land uses and land use trajectories analysed for particular regions. The ASB study for Indonesia excludes returns to one-off timber harvest from its calculations simply because the returns to timber harvest dwarf the returns to land uses that follow timber harvest which makes it difficult to compare the returns of alternative land uses (Swallow *et al.*, 2007).

1.3.1.4 Type of forest land considered

Opportunity cost estimates for a particular forest area should in theory be adjusted by the probability of being cleared over a given time period. In addition to local deforestation rates, the level of threat faced by a particular area of forest is determined by the type of property rights in place, proximity to transport and markets and the suitability of the land (climate and soil conditions) for alternative non-forest based economic activities,. The implementation of REDD projects in highly threatened areas is desirable, but the costs of these projects will be higher; forest at high risk of conversion is threatened precisely because of poorly defined property rights, proximity to roads and suitability for high-value cash cropping.

There is significant variation in the types of land included in the studies estimating opportunity costs. Börner and Wunder (2008a) estimate per hectare opportunity costs for private land in the Amazon in Mato Grosso and Amazonas. Only privately owned land is considered as it is assumed that public land is either protected (national parks and indigenous areas) or should be excluded from REDD programmes in order to avoid providing incentives for illegal land grabs. Nepstad *et al.* (2007) estimate foregone profits from forest-replacing agricultural and livestock production systems on *all* currently and potentially forested lands in the Brazilian Amazon regardless of the type of ownership. Nepstad *et al.* 2007 argue that the existence of protected areas and forest concessions is not guaranteed in perpetuity and can be undone to permit forest-replacing agriculture if continuous positive economic incentives to maintain forests are not provided. Other studies look only at forests that are directly threatened by deforestation in the near future. For example, Strassburg *et al.* (2008) overlay a GIS-referenced global map of potential economic returns from agriculture and pasture on to two GIS-referenced global databases of the spatial distribution of deforestation. They then derive final cost curves for avoided deforestation for each of the twenty countries included in the study.

For investors who hold property rights over forest land, the level of threat facing a forest may not be directly relevant. However, for investors seeking to obtain access to forest land the level of threat will be important and will move in line with opportunity costs. Forest land with climatic and soil conditions suitable for agriculture will be more threatened with conversion to agriculture. This land will be generally more costly to obtain. Moreover, if the land is suitable not only for pasture, but also for soybean cultivation, for example, that land will have a high opportunity cost and there will be intense pressure to develop that land.

The local rate of deforestation often reflects the level of threat to forests at a national level and is important for national and sectoral level REDD programmes. It may also be relevant for forest carbon projects if investors are keen to make interventions in countries and areas within countries which suffer from high deforestation rates. The most common source of global deforestation data is the UN Food and Agriculture Organisation (FAO) Forest Resource Assessment (FRA); this data is largely based on reporting from national forestry departments. A new study (Hansen, 2008), however, provides estimates based on an analysis of a combination of satellite images from NASA's Moderate Resolution Imaging Spectroradiometer (MODIS) and Landsat programmes. The findings of this new method suggests that Brazil accounts for nearly half of global deforestation with deforestation rates nearly four times the rate of Indonesia, the next highest country. FAO data suggest a broader distribution of deforestation and higher rates of deforestation in tropical Africa and Indonesia. Hansen (2008) also shows a shift from subsistence driven deforestation to industry and enterprise driven deforestation which has important implications for the design and cost of future REDD programmes.

1.3.1.5 Land use modelling method

Most empirical estimates of opportunity cost are based on the per hectare returns of individual crops which replace forests. A small number of studies, however, model land use trajectories based on observed land use patterns. For example, in the Amazon, forest land may be harvested for timber, after which it is cleared for pasture establishment for ranching. After a few years of ranching, soybean is planted (if climate and soil conditions are suitable and the land is near transport infrastructure and markets). The returns based on this type of land use trajectory more accurately reflect farming practices.

1.3.1.6 Carbon density estimates

Estimates of the opportunity cost of forest conservation are adjusted by the carbon-richness (ton carbon/ha) of the landscape to derive an estimated cost/ton carbon (C). For example, peat swamp forests are extremely carbon-rich and this reduces the cost/ton of carbon sequestered. Carbon stocks vary greatly within countries and forests. Carbon content estimates are presented in the case study section. Carbon content data also vary greatly depending on the source. For Indonesia, the FAO Forest Resource Assessment provides regional carbon content data which are a quarter of the values used by the Alternatives to Slash and Burn (ASB) Study. This divergence in discussed in more detail in the case study section, but the significant variation in estimates undermines the accuracy of the cost estimates produced. Where possible, net changes in carbon are calculated based on the difference in the carbon content of forests and the carbon content of the alternative land use.

1.3.1.7 Cost curves versus point estimates

An increasing number of studies estimate supply curves (prices) and cost curves which reflect changes in prices and costs as the amount of emission reduction changes rather than calculating point estimates of cost/ton CO2e avoided. Unit costs are not single constant values but differ for each quantity of reduction. It is therefore useful, when possible, to estimate a supply curve which graphs the quantity of CO2e supplied against the price paid (Boucher, 2008). In general, supply curves (with tons of CO2e or carbon along the x-axis and price along the y-axis) are curved lines with a flat slope which becomes very steep as the "choke price" is reached (Figure 4). The choke price is the price that would eliminate deforestation completely. At low levels of emissions abatement, unit costs are low; as the scale of abatement increases and low cost opportunities are exhausted, unit costs rise sharply. The shape of the curve illustrates diminishing returns as the scale of emissions reductions increases.

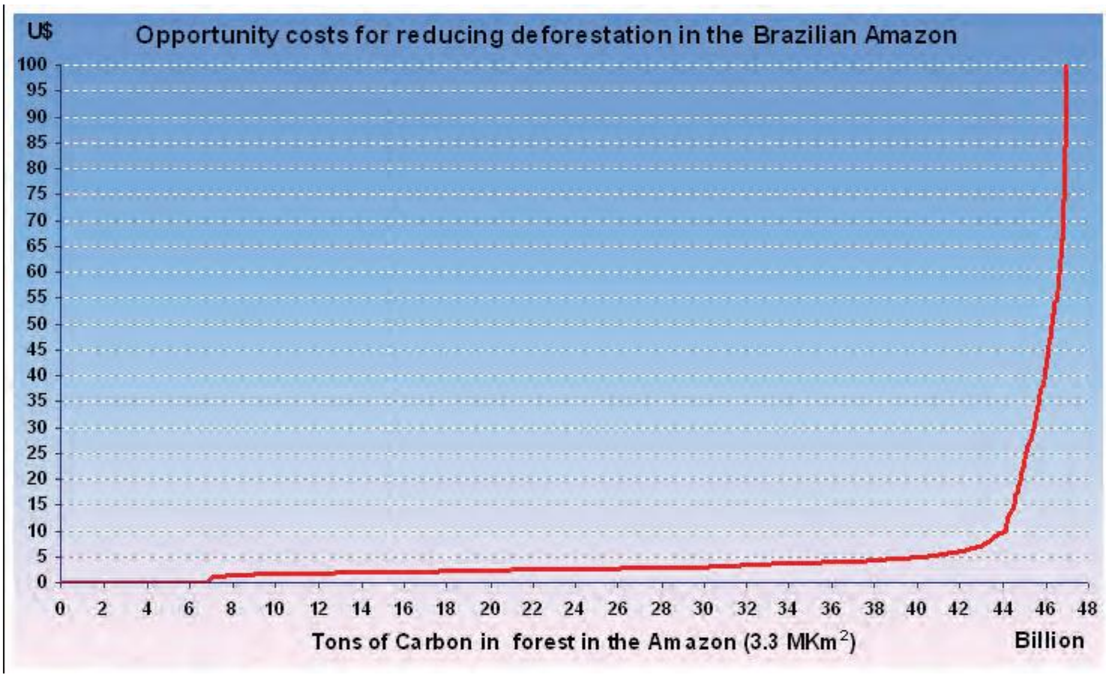


Figure 4: Marginal opportunity costs of reductions in carbon emissions in the Brazilian Amazon
Source: Reproduced from Nepstad *et al.*, 2007

Supply curves of avoided CO_2 e emissions or cost curves for avoided deforestation show that there significant opportunities for low-cost climate change mitigation based on reducing deforestation for early actors (Boucher, 2008; Strassburg *et al.*, 2008; Nepstad *et al.*, 2007). Cost curves have been estimated for regional and global carbon abatement based on REDD, but there is less data for costs at the micro-level. Moreover, small scale emissions abatement activities would not affect emission abatement levels sufficiently to cause significant movements along the cost curve. In short, it is difficult to estimate carbon supply curves or cost curves at the project level. For this reason, this review focuses on point estimates. Moreover, as REDD is currently implemented on a very limited scale, it is likely that point estimates of the cost are at the beginning of the cost curve where the unit costs of emissions reduction are very low. However, the expansion of forest protection through REDD will increase agricultural intensification and will thereby increase opportunity costs as the scale of forest protection increases over time. This will have implications for local livelihoods, poverty, food and energy security, etc.

1.3.2 Implementation and transaction costs

1.3.2.1 Identifying the components of implementation and transaction costs

The implementation and transaction costs of potential REDD programmes and forest carbon projects will vary according to:

- local drivers of deforestation,
- the current state of institutions and structures to implement and enforce forest rules,
- the type of REDD programme adopted (project, sectoral, national),
- the type of financing mechanism used for REDD, whether it involves selling REDD credits in carbon markets, a voluntary fund, etc.

There is little empirical quantitative work on these costs and some argue against estimating these costs prior to more resolution on UNFCCC decisions regarding REDD in a post-2012 regime (Blaser and Robledo, 2008). A further challenge in collecting, comparing and aggregating estimates of implementation and transaction costs is that many studies define loosely or differently what components are included in implementation and transaction costs. The elements of implementation and transaction costs are listed in detail below.

Implementation costs are the costs associated with implementing a REDD programme, i.e. costs directly associated with actions to reduce deforestation. Examples are costs of guarding a forest to prevent illegal logging, relocating timber harvesting activities away from natural forests to degraded land due to be reforested, agricultural intensification (Pagiola and Bosquet, 2009).

Transactions costs are additional costs to be paid by parties (buyers, sellers, donors, recipients) to a transaction involving a REDD payment, as well as external parties such as market regulators or payment system administrators to ensure a certain amount of emissions reduction has been achieved (Pagiola and Bosquet, 2009). Activities which are components of transaction costs include identifying REDD programmes, negotiating transactions, and monitoring, reporting and verifying emissions reductions. Transaction costs differ from implementation costs in that they do not themselves reduce deforestation or forest degradation, but are needed to establish the transparency and credibility of REDD programmes.

It is important to distinguish between implementation and transaction costs faced by carbon sellers (land holders), carbon investors or purchasers (private or public sector) and by the regulating body (national or international). Table 3 below presents a typology of costs (Cacho *et al.*, 2005) and begins to sketch out a distribution of costs incurred at various stages of REDD. It would be useful to complete the table with quantitative estimates (US$/ha), but there is currently little empirical data available to do so.

Table 3: Typology and distribution of implementation and transaction costs

Implementation and transaction cost	Description	Fixed/ variable cost*	Cost incurred by carbon seller	Costs incurred by carbon investor
Search	Costs incurred in seeking project partners	Fixed	-cost to find buyers	-broker's fees -charges for information services -cost of advertising willingness to invest - baseline
Negotiation	Costs of partners reaching an agreement	Fixed	-legal costs -costs of project design -distribution of responsibilities -delays -assignment of benefits (payment in cash, carbon credits, etc)	
Verification	Validity of claims of project must be verified by accredited agency (ex ante)	Fixed		-advance approval of project involves out of pocket costs, and costs associated with uncertainty and delay
Certification	Carbon sequestration must be checked and certified by accredited agency (ex post)	Fixed		
Implementation	Costs of putting into practice project design	Variable		-set up project office -keep records of project participants -administration of payments -managing disagreements
Monitoring	Costs of observing transaction and to verify compliance with terms of agreement	Variable		-measure actual GHG abatement
Enforcement	Costs of enforcing compliance if monitoring detects breach of agreed terms of contract	fixed		-costs of enforcing contracts and litigation
Insurance	Costs arising from project failure (due to fire, seller fails to provide emissions abatement, investor fails to pay)	variable	-purchase of financial insurance policy -development of contingency plans	-deduction of risk premium from price paid -purchase on spot markets -diversification of portfolio

*Variable costs based on existence of economies of scale for cost category.

Source: Adapted from Cacho *et al.*, 2005

1.3.2.2 National versus project-based REDD implementation and transaction costs

Implementation and transaction costs for project level REDD activities borne by individual groups are a sub-set of the costs for national and international REDD programmes. However, project level REDD will operate in the context of international and national REDD programmes. It is therefore important to consider what the costs of REDD at different levels will be and who bears these costs. REDD readiness will most likely be financed with public money as it requires significant capacity building and policy formulation/reform.

There is scope for carbon revenues to be a predominant source of financing for REDD implementation costs. However, REDD financing is likely to be ex-post–deforestation should be reduced and then verified prior to carbon credits being issued. As such, implementation funds can either come from governments committed to a national REDD strategy or private sector investors for individual REDD projects. This suggests that REDD will be more "do-able" in more developed, well-governed countries where either (a) governments will have revenue available for up-front investments or (b) private sector investors willing to make up front investments will be attracted to REDD activities in low risk countries (Hoare *et al.* 2008).

1.3.2.3 Economies of scale

The scale of REDD initiatives significantly influences implementation costs. On the one hand, if there are significant economies of scale in combating deforestation, it may be most cost-effective to undertake REDD at a national level. On the other hand, if the costs of reducing deforestation are highly variable within a country, it may be possible for early entrants to lock in relatively low cost carbon credits by focusing on 'low hanging fruit' in a more project-based REDD approach.

Antinori and Sathaye (2007) show empirically that there are significant economies of scale in implementation costs. As a result, it has been argued that it is not appropriate to simply add on a fixed multiplier or percentage for these additional costs as unit costs fall with the size of the project or programme (Boucher, 2008). However, in the absence of practical alternatives, this study applies a fixed cost per ton CO_2 e. An alternative is to include implementation costs as a fixed proportion of opportunity costs. For example, ONF International (2008) sets implementation costs at one third of opportunity costs, Grieg-Gran (2006) at 5–20 percent and Boucher (2008) at 20 percent Neither approach is optimal, and further research is needed.

While there are economies of scale in implementation costs, transaction costs are likely to be fixed at the project or national level (Pagiola and Bosquet, 2009). This implies that the share of transaction costs in the costs of REDD depends on the degree of success in emissions reduction; if emissions reduction is large, transaction cost expressed in terms of cost per ton CO_2 e will be low.

1.4 *Empirical estimates of implementation and transaction costs of REDD*

1.4.1 Project-based costs

Little data is available on implementation and transaction costs of Payments for Ecosystem Services (PES) schemes, and ex ante estimates have frequently been inaccurate. What is known, however, is that transaction costs for a large number of small volume transactions are likely to be high, as are transactions in immature markets (Börner and Wunder, 2008a). Early carbon-based PES schemes in Brazil and Bolivia experienced very high transaction costs in large part due to the uncertain market environment and restrictive carbon market rules (May *et al.*, 2004). Other studies (Cacho *et al.*, 2005 in a study of carbon projects in Indonesia) have found large start up costs but relatively low recurrent costs.

Some research suggests that transaction costs account for a quarter of the costs of providing ecosystem services (Slangen *et al.*, 2008 discussed in van Kooten, 2008). Other studies (Börner and Wunder, 2008b) find that the opportunity costs of forest conservation are large relative to potential implementation and transaction costs and exclude these costs from their calculations due to the paucity of data and the speculative nature of including quantitative estimates of transaction costs. The section below looks at empirical estimates at the project, sector and national-level costs. Table 4 summarises available empirical estimates of implementation and transaction costs from projects from Payments for Ecosystem Services (PES) projects.

Table 4: Summary of transaction and implementation costs for PES projects

Project	Country	Transaction or implementation cost \$/ton CO_2e	Start up costs \$/ha	Recurrent cost \$/ha	Implementation costs	Administration costs \$/ton CO_2 e	Source
11 forestry offset projects	Global	0.38 (0.03-1.23)					Antinori and Sathaye (2007). Transaction costs.
Pimampiro watershed protection	Peru		76	7			Wunder and Alban (2008)
PROFAFOR carbon sequestration	Ecuador		184	3			Wunder and Alban (2008)
Full implementation REDD in Amazon	Brazil				0.58		Nepstad *et al.* (2007). Implementation costs including project and national level costs.
PES projects in Central and Latin America	Americas					0.01-0.04	Grieg-Gran (2006). Includes Costa Rica, Mexico, Ecuador.
U.S. Conservation Reserve Programme (CRP)	United States	1					Sohngen (2008).
Sustainable Forest Management	Global		6.25				UNFCCC (2007) from ITTO expert panel, cost of SFM for tropical production forests

In a study of 28 mitigation projects (avoided deforestation, afforestation and other offsets) worldwide, Antinori and Sathaye (2007) estimate that average transaction costs for eleven forestry offset projects is US$0.38/ton CO_2e; transaction costs range from US$0.03/ton CO_2e for large projects to US$1.23/ton CO_2e for small projects. Transaction costs were estimated to be lower for large projects than for small projects and forestry projects have lower transaction costs than non-forest GHG projects. Transaction costs were higher for projects in South America than elsewhere and lower for energy efficiency projects and projects in Asia.

Focusing on transaction costs in PES schemes in Latin America, Wunder and Alban (2008) assess the costs of a watershed protection programme in Pimampiro in Ecuador, with start up costs of US$76/ha and annual recurrent transaction costs of US$7/ha. Also in Ecuador, PROFAFOR's carbon sequestration programme start up costs were estimated at US$184/ha with annual recurrent transaction costs of US$3/ha.

Nepstad *et al.* (2007) estimate implementation costs including project level and national level costs. At full implementation of a REDD programme in the Brazilian Amazon, implementation costs are US$0.58/ton CO_2e.

Administration costs estimated by Grieg-Gran (2006) range from US$4–15 per hectare ($0.01–0.04/ton CO_2e). This estimate is based on national level payments for environmental services schemes in Costa Rica, Mexico and Ecuador. Costs include administration costs of those administering the schemes (e.g. FONAFIFO in Costa Rica) and costs incurred by PES recipients in the application process. This study also finds that small schemes face high transaction costs due to the large fixed cost element, and that there are significant economies of scale.

In order to get an aggregate measure of implementation and transaction costs, Boucher (2008) sums Antinori and Sathaye's (2007) estimate of transaction costs (US$0.38/ton CO_2e), Nepstad *et al.*'s (2007) implementation cost estimate (US$0.51/ton CO_2e) and Grieg-Gran's (2006) highest administrative cost estimate (US$0.04/ton CO_2e) to derive a total of US$1/ton CO_2e. While there is some overlap in the components of this sum, this double-counting ensures that the estimate is "conservative". While these costs are not negligible, they are likely to be significantly smaller than the opportunity cost component of most prospective REDD programmes.

The Noel Kempff Mercado National Park project in Bolivia, a large emissions avoidance (avoided deforestation) project, estimated at start up costs of US$0.50/ ton C sequestered (US$0.14/ton CO_2e). This estimate was later revised upwards to US$1/ton C (US$0.28/ton CO_2e) based on improved baseline data (on carbon content in particular) and the implications of a new forest law passed in 1996 (May *et al.*, 2004).

The experiences of the Peugeot afforestation project and the Ilha do Bananal carbon project in Brazil suggest that it may be difficult to undertake forest carbon projects on a commercial basis (May *et al.*, 2004) if carbon prices are low. Low carbon prices, high initial costs (imported technologies and technical delays), and the absence of harvesting of forest and agricultural by-products to offset costs undermined the cost-effectiveness of the Peugeot project. However, it is a pilot project and lack of effectiveness and high transaction costs have largely been attributed to mistakes made in the conception and management of the project. The Ilha do Bananal project sought to show that there are ways to generate foreign exchange in Brazil other than ranching and soybean production even though other land use might not be as profitable on a per hectare basis.

An aspect of REDD which has been predicted to cause transaction costs to increase is the need for clear identification and definition of property rights over forest land. As forest carbon credits go to land owners via contracts between carbon investors and land owners, the lack of widespread land titling in many countries with large tracts of forest will likely pose significant barriers to forest carbon projects (May *et al.*, 2004). However, recent work on the costs of recognising local and indigenous rights (Hatcher, 2007) suggests this type of expenditure is highly cost effective (although the political costs could be high). It is commonly accepted that REDD cannot proceed successfully without ensuring the rights of local and indigenous people are recognised. The costs of doing so have been estimated using empirical data from Brazil's demarcation of indigenous territory (US$0.05/ha), Mozambique's demarcation programme (US$0.18/ha), the creation of social reserves and protected areas in Brazil (US$0.50/ha) and World Bank calculated costs for land titling in Laos, the Philippines, Indonesia and Cambodia (Hatcher, 2007). Costs of recognising community tenure and land rights range from $0.05-9/ha (US$0-0.2/ton CO_2e), with an average costs of $3.35/ha (US$0.08/ton CO_2e). Costs increase with the remoteness of the area, political opposition, and the need for international expertise. These costs include the costs of awareness raising, dispute resolution, equipment and material, staff costs (government and NGO), training, administrative costs and recurrent costs of training, staff, etc. These costs expressed per ton CO_2e are very small. However, in terms of the success of REDD projects and programmes, investment in rights and tenure recognition is critical.

1.4.2 Sector-based costs

Sohngen (2008) calculates potential transaction costs of REDD by looking at the budget of the U.S. Conservation Reserve Program (CRP) and, coincidentally, also estimates these costs to be US$1/ton CO_2e (in a rough back of the envelope calculation). Over 25 years, the CRP changed the use and management of more than 36 million acres of land in the U.S. CRP rental payments to farmers required roughly seven percent of the Farm Service Agency's US$25.5 billion budget in 2004. Adding seven percent of FSA's expenditure on salaries and expenses on all its programmes, Sohngen estimates CRP's implementation cost at $92.8 million annually. For 36 million acres, this amounts to US$2.50 per acre/year. An average afforested acre in the U.S. is able to sequester 2.4 tons CO_2e per year. Administrative costs are thereby estimated at roughly US$1/ton CO_2e. Sohngen considers this estimate comparable to that of Antinori and Sathaye (2007) as the latter consider projects which are much smaller than what would be required under a policy to stabilise national level emissions. In a national programme costs would be higher as all carbon at the national level would have to be measured, not just the carbon in areas where forest conservation was being considered (Kindermann *et al.*, 2008). Sohngen concludes that transaction costs are not large relative to the carbon prices that are likely to emerge under future global climate policies.

The ITTO Expert panel (UNFCCC, 2007) estimates the costs of achieving sustainable forest management for tropical production forests to be US$12/ha which is equal to US$0.14/ton CO_2e assuming a low carbon content estimate of 90 ton carbon/ha (the lowest carbon content estimate used in both the Brazil and Indonesia case studies). While there has been little experience of REDD projects to date, there has been significant implementation of SFM projects which could potentially provide empirical and country-specific estimates of transaction costs for forestry type projects.

1.4.3 National capacity building and implementation costs

Hoare *et al.* (2008) look at the costs of building institutional capacity in rainforest nations and define three types of governance interventions (with associated costs) needed for these tropical countries to participate in REDD.

- Mechanism costs include costs associated with calculating baselines, development of monitoring and measuring capacity, and issuing credits. The Hoare study cites a number of empirical estimates for mechanism costs:
    - Based on experience in India and Brazil, the cost of setting up national monitoring systems is estimated at US$500,000 to US$2 million per country.
    - IPCC estimates the cost of establishing national vegetation carbon inventories at between US$0.05–0.6 per hectare (2000 prices).
    - In Cameroon, a recent national forest survey undertaken in collaboration with the FAO cost EUR 622,692.
- Developing a national strategy for REDD includes costs associated with identifying the local drivers of deforestation and degradation, identifying mechanisms to address the main drivers and prioritising the drivers to be addressed and identify the best mechanisms to do so.
- Implementing a REDD strategy involves costs of policy and institutional reform (to address lack of governance, tenure, land-use planning, tax and other policy drivers) and costs of specific activities (e.g. tackling illegal logging, sustainable forestry management, alternative livelihoods, protected areas, etc)

The Hoare study's quantitative estimates focus on "readiness" costs for REDD. These costs must be met for REDD to operate on a national basis, but also to a certain extent for private sector investment in REDD projects to be recognised, monitored and rewarded within a REDD programme. The study identifies types of intervention required for REDD and estimates the costs associated with these interventions based on the costs of similar activities implemented in the past by a number of donors (DFID, World Bank, ITTO, AusAID and various experts). These estimates are based on national REDD programmes; the costs of project-based REDD initiatives would be a sub-set of these. These costs are very rough estimates. In fact, if the authors were to do this exercise again, the authors would increase a number of the estimates (Hoare, personal comm. 03.11.2009). Accuracy is affected by the fact that donor expenditure is not based on estimates of what components cost but on the size of funds available and on donor priorities. The study notes that even the upper estimates of readiness costs are low relative to what could be achieved and relative to donor commitments already made. A summary of these costs is reproduced in Table 5 below.

Table 5: Estimates of costs of "readiness" for REDD

Activity	Lower estimate	Upper estimate	Upper estimate excluding "size dependent"	Comment
Development of REDD strategy	*$200,000*	*$1,000,000*	*$1,000,000*	
Establishment of REDD infrastructure	*$700,000*	*$1,500,000*	*$1,500,000*	
Stakeholder consultations	*$150,000*	*$2,000,000*	*$150,000*	*size-dependent*
Pilot testing	*$250,000*	*$500,000*	*$500,000*	
Establishment of baseline, monitoring system and inventory	*$1,000,000*	*$7,000,000*	*$7,000,000*	
Land tenure reform	*$4,000,000*	*$20,000,000*	*$4,000,000*	*size-dependent*
Land-use planning & zoning	*$1,750,000*	*$10,000,000*	*$10,000,000*	
Development of capacity to provide support services for implementation activities, e.g. RIL, agricultural intensification	*$1,750,000*	*$10,000,000*	*$10,000,000*	
Forest policy and legislation reform	*$300,000*	*$1,000,000*	*$1,000,000*	
Tax reform (e.g. removal of subsidies/ tax incentives	*$300,000*	*$1,000,000*	*$1,000,000*	
Standards and guidelines	*$50,000*	*$1,000,000*	*$1,000,000*	
Enforcement of planning & environmental requirements & forest laws	*$500,000*	*$2,000,000*	*$2,000,000*	
Independent monitoring	*$1,000,000*	*$5,000,000*	*$1,000,000*	*size-dependent*
NGO capacity building	*$100,000*	*$1,000,000*	*$1,000,000*	
Effective judicial system	*$500,000*	*$5,000,000*	*$5,000,000*	
Institutional reform, clarification of roles & responsibilities, capacity building	*$600,000*	*$14,000,000*	*$14,000,000*	
Treasury reform	*$500,000*	*$5,000,000*	*$5,000,000*	
Establishment of ability to process and manage payments to project beneficiaries	*$100,000*	*$5,000,000*	*$5,000,000*	
5-year costs for one country	**$13,750,000**	**$92,000,000**	**$70,150,000**	
Total 5-year costs for 25 countries	**$343,750,000**	**$2,300,000,000**	**$1,753,750,000**	

Source: Reproduced from Hoare *et al.*, 2008

1.5 REDD as part of a bundle of measures to reduce deforestation

In practice, REDD programmes are likely to include a mix of actions, such as support for agricultural intensification, agricultural and forest subsidy reform, land tax reform, changes in infrastructure planning and investment, forest law enforcement, land tenure reform, decentralization, forest certification, fire prevention, establishment of protected areas, payment for ecosystem service schemes, etc. While at the project level, investors will likely have little influence over national policies, the nature and changes to national policies will significantly affect the costs of investing in REDD. For example, changes in commodity prices due to changes in agricultural subsidies and fluctuations in world market prices will affect the opportunity costs of forest land. Changes in forest law enforcement, land tenure reform, decentralisation will affect the transaction costs of REDD at all levels.

1.6 REDD and co-benefits

Forests provide a range of valuable ecosystem services in addition to carbon storage, and there is currently discussion on the extent to which carbon credits can potentially incorporate non-carbon benefits. What is the scope for establishing synergies between carbon storage (the ecosystem service for which markets are most developed and sophisticated) and biodiversity (which is trickier to measure and buy) or water-related benefits? Currently, the standard unit traded in the compliance carbon market is ton CO_2e – there is little scope for differentiating credits based on benefits additional to carbon storage.

On the other hand, the voluntary market serves investors who are interested and willing to pay for carbon credits differentiated on the basis of quality (sustainability) or bundled characteristics (carbon and biodiversity or pro-poor REDD). For example, the Climate, Community and Biodiversity Alliance (CCBA), a partnership between the private sector, NGOs and research institutes, had developed voluntary standards to assist the identification and design of land management projects that reduce emissions, protect biodiversity and promote sustainable development. While voluntary schemes may be successful, their scale will likely remain limited. What is the scope for integrating other benefits into carbon credits traded in the compliance market? Is it wise to do so, or would climate change mitigation and the protection of biodiversity be better served by different and separate instruments?

In terms of the costs of REDD, it is reasonable to expect that carbon credits delivering biodiversity benefits will be more costly to produce. This would be the case because project design, monitoring and verification would be more complex. On the other hand, current trade in the voluntary market suggests that there is a willingness to pay for these additional costs via a price premium on higher quality credits.

An important issue is whether the forest land that provides the most cost-effective carbon abatement opportunities (low opportunity cost) is the forest land that contains high biodiversity values. Because the cost of conservation per ton CO_2e is driven not only by the opportunity cost but also by the carbon content of land, high carbon content is an important component of cost-effective mitigation. The Carbon and Biodiversity Demonstration Atlas developed by UNEP's World Conservation Monitoring Centre (WCMC) indicates where areas of high carbon storage intersect with areas of high biodiversity on a regional basis and for six countries. In the Neo-tropics high carbon-high biodiversity forest covers more than three percent of total land area and contains more than 4 percent of the total regional carbon stock. The Oceania and Asia region shows a particularly high coincidence of carbon and biodiversity with more than ten percent of total carbon stock located in high carbon high biodiversity forests. This coincidence is especially marked in the island archipelagos and the Western Ghats (Kapos *et al.*, 2008).

1.7 Distributional issues and REDD design

The distribution of the costs and benefits of project-based REDD will depend on local conditions, institutions, community equity, the status of land tenure, and other socio-economic characteristics of the project area. While there is little practical experience of REDD and hence little is known of the distributional implications, the experience of PES schemes implemented in the last decade provides some insights into potential distributional issues (Huberman, 2009; Peskett *et al.*, 2008). There is concern that REDD would exacerbate unequal power relations in countries and regions with weak governance institutions. In particular, there is a concern that indigenous peoples who are often the de facto guardians of tropical forest, would lose out as forest resources become relatively more valuable under REDD. The rights of local and indigenous people need to be strengthened in face of government measures to restrict access and harvesting of forest resources so that local livelihoods are not negatively affected by REDD.

While the objective of PES schemes is generally not poverty alleviation, poverty alleviation is regarded as a desirable potential co-benefit. However, the distribution of benefits from PES schemes has in some cases been considered inequitable, and there appears to be a strong efficiency-equity trade-off. For the efficient delivery of ecosystem services, it is not necessarily the poor who are the optimal participants in PES schemes. National PES schemes have in some cases marginalised small landowners and the poor (Adamson-Badilla, 2003; Hope *et al.*, 2005; Zbinden and Lee, 2005; Pfaff *et al.* 2007a cited in Huberman, 2009). In Costs Rica, where PES schemes have been implemented for the last 12 years, small landowners have been marginalised due to high transaction costs, poorly defined land tenure and lengthy and complicated administrative processes (Pagiola, 2008). In general, high levels of risk associated with investments, lack of capital for upfront payments and poorly established property rights regimes have prevented the poor from benefiting. In Mexico's PES scheme initiated in 2003, incentives targeted community-owned projects (Kaimowitz, 2008 cited in Huberman, 2009). It was found that those communities with higher levels of social organisation benefited disproportionately. In general, where communities are characterised by low levels of trust and collaboration, PES benefits are captured by elite groups. In a recent case study of carbon sequestration in Panama, uneven natural resource endowment was the major impediment to participation in PES (Tschakert *et al.*, 2007 cited in Huberman, 2009). Moreover, social cohesion at the community level may be eroded with the introduction of external sources of revenue or investment as individuals struggle over local resources. Conflict between communities may also increase; conflict over unclear boundaries between villages in Borneo became a source of conflict as the opportunities for gaining revenue from conservation were introduced (Huberman, 2009).

The current status of international negotiations with regard to the design elements of REDD are discussed in detail in Peskett *et al.* (2008) on which this section draws heavily. The distributional implications of REDD depend critically on the architecture and operation of a REDD programme and will vary across countries, regions and communities. Current discussions focus on a number of critical design elements some of which are particularly important for private sector investment in forest conservation:

- How a future REDD framework fits into the existing UNFCCC framework has important implications for whether REDD-type projects and programmes can be included in the compliance carbon market or whether they will remain in the voluntary carbon offset market (where traded volumes and prices are lower). Inclusion in the Kyoto Protocol would allow REDD type projects to enter an already existing carbon trading system, avoiding the risks of setting up a new process and market and potentially insufficient demand. It would be in the interest of private investors to mainstream REDD carbon credits into the existing carbon market. If private investors in REDD activities are limited to the voluntary market, the lower prices they receive for REDD-based carbon credits would reduce their interest in REDD relative to other investment opportunities. The voluntary market has rules and procedures that differ from the regulated market under the Kyoto Protocol. However, it is likely to persist alongside the regulated market by implementing different types and scales of projects based on less restrictive rules. There is consensus that the inclusion of REDD as a market mechanism within a Protocol (Kyoto or separate) will result in higher volumes of finance available compared to an international fund for REDD. The rules under which REDD will operate will also depend on the regulatory framework. Both the volume of finance and the rules governing REDD (which activities may receive compensation and verification procedures) will have implications for the distribution of the net benefits and costs of REDD activities.

- The selection of the financial mechanism(s), whether an international fund or a market-based mechanism, has implications not only for the volume of resources mobilised, but also for the scope of private sector involvement in REDD. A fund-based approach would be based on an international fund to provide developing countries with incentives to reduce deforestation. Such a fund would not be linked to carbon markets and the amount of funding going to particular countries need not be based exclusively on performance in emissions reductions, but could also include non-performance based criteria such as pro-poor policy changes. Market-based mechanisms would use trade in carbon credits to bring about emissions reductions. The main difference in the results of the two approaches is the likely scale of finance with market-based systems expected to generate much larger financial flows for REDD with greater potential for (indirect) pro-poor impacts. A market based mechanism would provide greater scope for private sector involvement in the REDD activities.

- Liability issues – how can investors in REDD be guaranteed that emissions reductions are permanent and have not been compromised by leakage (the displacement of deforestation to non-REDD areas)? There are a range of alternative approaches for dealing with risk, each with different distributional implications. The establishment of risk buffers is based on withholding a percentage of credits (roughly 30%) from sale as insurance against project failure (forest fires, storms, institutional failure); this option lowers the profitability of investing in carbon offsets for the investor and lowers benefits to local stakeholders. Alternatively, payment after verification significantly reduces risks for carbon credit purchasers; however, lack of access to credit would prevent local communities from making necessary investments in the absence of upfront financing. In some cases, governments would take on these liabilities. Other options include holding geographically diverse portfolios (to reduce the impact of forest fires on investors), temporary credits as used in CDM afforestation and reforestation projects, repayment of revenues/fines.

- Spatial scale – national versus project based approach has important implications for how emissions reductions are accounted for and how they are credited. Important distributional issues related to spatial scale include how finances and authority are distributed between government (central and sub-national) and non-governmental actors. The greater the degree of fiscal decentralisation, the greater the likelihood that financial benefits reach the local level and the lower the chances of "elite capture" (Peskett *et al.*, 2008).

- Private investors are likely to prefer large-scale projects due to their lower transaction costs due to economies of scale. It is likely that there will be fewer opportunities for local communities to benefit from large-scale projects than from small-scale projects (Cosbey *et al.*, 2006 cited in Dutschke *et al.*, 2008).

- Efficiency-equity trade-offs are prevalent in the selection of project investments. For example, in the CDM there has been a great deal of investment in projects regarded as bearing "low hanging fruit", i.e. low cost of emissions reduction, particularly in China and India. These types of projects which include the introduction of cleaner technologies into industrial processes generally have fewer benefits for local people (Peskett *et al.*, 2008). Combined with private sector preferences for large scale projects, the equity-efficiency trade off will affect private investment in REDD; private sector investors are generally less concerned with distributional issues. That being said, some private investors are interested in projects with social or other (e.g. biodiversity) benefits as these projects attract higher prices and boost corporate reputations. There are already some standards schemes which provide "premium" credits from projects with high sustainability (Peskett *et al.*, 2008). Standards which reward investment in projects with benefits additional to carbon increase the interest of private sector investors who would benefit financially in receiving higher prices for carbon credits and in promoting their "license to operate" by being seen to operate in a responsible manner.

- Reference levels/scenarios are used to judge performance in reducing emissions related to deforestation. The baseline approach defines a scenario of projected emissions in the future (based on historical rates or projections). Actual performance is assessed relative to this baseline. Another approach uses a cap and trade system in which countries take on emissions targets or levels. The discussion of baselines is usually in the context of national REDD programmes; however, project-level REDD will also require a system by which to judge performance of emissions reductions and further consideration of project- level baselines or targets needs to be done. The issue of project level baselines is linked to issues of additionality. It is expected that some form of additionality will be part of any REDD programme. As such, projects will have to show that in their absence, a particular area would be deforested or degraded. For investors who already hold property rights over a forested area or who obtain forested land with the intention of conservation, adherence to an additionality criterion for inclusion in a REDD programme may be challenging.

At issue is if and how REDD programmes can be designed to be pro-poor or pro-local community, and if REDD is strengthened or weakened in efforts to do so. Private investors should be concerned and should undertake thorough due diligence in order to ensure projects distribute benefits equitably. Transaction costs may be higher for projects as a result of the additional measures needed to ensure distributional fairness. However, these higher costs will in most cases be necessary investments as, without them, REDD projects will not have the support of local communities needed to succeed in the long run.

2 Case study 1: Costs of REDD in the Brazilian Amazon

Brazil contains the largest portion of the largest single block of tropical forest in the world. It is also one of three countries with the highest rate of deforestation globally, despite recent improvements, and is responsible for approximately half of the annual global deforestation (Hansen, 2008). It is the second largest global emitter of GHGs from deforestation, and is responsible for roughly 2.5 percent of global greenhouse gas emissions. Roughly 75 percent of Brazil's GHG emissions are from deforestation in the Amazon, and represent 8–14 percent of global land-use change emissions (Vera Diaz and Schwartzman, 2005). Estimates of GHG emissions from deforestation do not include emissions from forest fires which in some years are very high (Nepstad *et al.*, 1999).

This case study looks at the opportunity cost of forest conservation in the Brazilian Amazon based on the main drivers of deforestation. The transaction and implementation costs associated with a potential REDD programme in the Amazon are also discussed.

2.1 Main drivers of deforestation

The main drivers of deforestation and forest degradation in the Amazon are cattle ranching, soybean monoculture and logging. Extensive, low yield cattle ranching accounts for roughly 70 percent of deforestation, despite low rates of return (3–14 percent), due to fiscal incentives and land speculation (Vera Diaz and Schwartzmann, 2005). While cattle ranching is the prevalent direct driver of deforestation, it is the expansion of soybean cultivation that is the main economic force behind deforestation. Economic returns are high although soy bean expansion is more constrained by soil and climate factors than is cattle ranching. Traditionally, soybean production takes place pasture areas and displaces cattle ranching to forest areas and is thus a powerful indirect driver of deforestation. Soybean is also beginning to be grown in forest areas near ports and highways in the Amazon.

Roughly 77 percent of cleared land in forest margins is under pasture and 8 percent is planted to annual crops (Chomitz and Thomas, 2003). Climatic and soil conditions greatly influence conversion of forest to agriculture - the probability that land is currently claimed, or used for agriculture or ranching, declines steeply with increasing precipitation levels. The Figure 5 below shows the impact of rainfall and farmgate beef prices on the rate of deforestation in the Amazon.

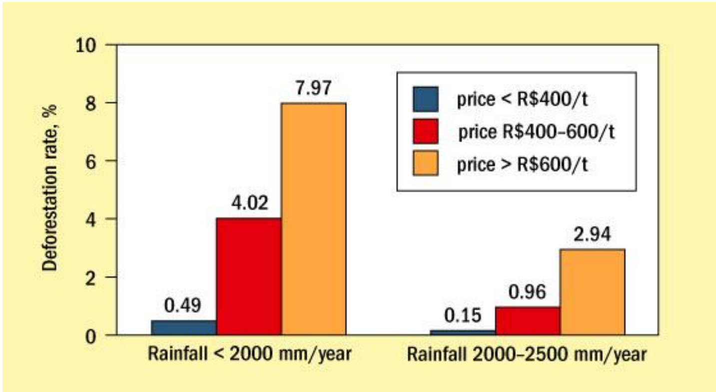


Figure 5: Deforestation in the Brazilian Amazon, Rainfall and Beef Prices, 2001-03
Source: Reproduced from Chomitz *et al.*, 2007

Chomitz and Thomas (2003) provide evidence that almost half of all agricultural land in the Amazon is operated by large scale farms and only 1.5 percent is operated by farms smaller than 20 hectares in size. The structure of property rights and climatic conditions affect not only opportunity costs of forest conservation but should also be considered in the process of identifying forest areas which are under threat (likely to be converted to non-forest uses). The 'arc of deforestation' in the Amazon is an area which is privately owned and contains conditions amenable to the cultivation of soybean, the most profitable non-forest activity.

2.2 Type of forest land for inclusion in REDD

There is significant variation in forest condition, carbon content and opportunity cost of forest land across the Amazon. Hence the selection of areas to be included in a REDD programme will significantly affect the costs of these programme. What criteria should provide the basis for selection of forest land for REDD? Most studies consider the Brazilian Amazon as a whole (Nepstad *et al.*, 2007; Vera Diaz and Schwartzman, 2005; Grieg-Gran, 2006). Others try to obtain more localised estimates of the costs of REDD due to the high level of variability across the Amazon; Börner and Wunder (2008a) focus on the state of Amazonas which is remote from markets, has low deforestation rates and conservationist policies, and the state of Mato Grosso, located in the "arc of deforestation" and home to a rapidly expanding agri-business sector based on beef and soybean with land supplied through relatively high rates of deforestation.

Börner and Wunder (2008a) identify areas directly threatened by deforestation (in projections and simulations) because REDD programmes will include an additionality criterion. Their estimation of opportunity costs is limited to private land holdings in the Amazon; if payments to farmers aggressively taking over publicly held land were included in a REDD programme, this type of undesirable behaviour would be encouraged. According to the Brazilian Forest Code, 80 percent of private forest land must remain under forest; however, this policy is not always implemented, is difficult to monitor and the Forest Code itself is often threatened with being changed or amended to allow more commercial use of private forest land.

Another approach (Nepstad *et al.*, 2007) looks at the foregone profits from forest-replacing agricultural and livestock production systems on all currently and potentially forested lands under all kinds of ownership. This approach is based on the argument that protected areas and forest concessions can be undone to permit forest-replacing agriculture. This study argues that continuous positive economic incentives are needed to maintain forests and prevent leakage within countries. Filters are used to limit the potential of soybean expansion to geographically suitable areas.

2.3 Inclusion of timber extraction rents

In general, the inclusion of net timber rents in the estimation of opportunity costs has a large influence on the estimates. Börner and Wunder (2008a) find that opportunity cost estimates are very sensitive to timber returns because (i) timber rents can be sizeable and (ii) they are received at the beginning of the NVP calculation and are therefore not discounted. The problem is aggravated by the fact that the high NPV estimates may not accurately reflect incentives faced on the ground as timber rents are not always received in full by land owners - timber harvesting may occur before (and causally disjointed from) the deforestation process. On the other hand, forest clearing for ranching is in many cases costly: Grieg-Gran (2006) discusses how the costs of cutting trees remaining after harvest of valuable species and the costs of establishing pasture may well exceed revenue from the sale of timber or timber rights. In general, however, most studies for Brazil include one-off returns from logging and provide a higher (more conservative) estimate of the full opportunity costs of forest maintenance.

2.4 Opportunity costs per hectare

A number of studies look at land use trajectories (Vera Diaz and Schwartzman, 2005; Börner and Wunder, 2008a,b) while others estimate per hectare returns to individual agricultural and livestock activities (Nepstad *et al.*, 2007; Tomich *et al.*, 2005; Grieg-Gran, 2006). Börner and Wunder (2008a) develop land use trajectories specifying that forest is first cleared for annual subsistence crops, followed by pasture and later by cycles of fallow-based slash and burn agriculture. Vera Diaz and Schwartzman (2005) model timber extraction followed by five years of pasture and then soybean.

Nepstad *et al.* (2007) calculate opportunity cost of forest maintenance using spatially explicit rent models for timber (high carbon) and agriculture and ranching (low carbon) alternative uses of forests in Brazilian Amazon. The study estimates potential rents for soy, cattle and timber production. For example, the soy model is based on a biophysical yield model, a transportation model and a production cost model to estimate returns to soy production in the Brazilian Amazon. A filter is used to constrain soy expansion to suitable areas depending on soil and climate suitability map. Because findings are not presented in numeric values, but in spatial maps, these results are not directly comparable to other estimates and are presented separately below.

Appendix 1, Table A1.1 presents detailed information on opportunity cost estimates for the Brazilian Amazon. Cattle ranching has very low levels of profitability; small scale and traditional ranching produces profits between US$2 to 332/ha. Medium and large scale ranching, extensive ranching and improved pasture ranching are more profitable at US$461 to 1033/ha. Animal grazing density is between 0.5 (Chomitz and Thomas, 2005) and 0.8 animal units per hectare and profits are generally less than $50 per hectare (Nepstad *et al.*, 2007). In future, disaggregating livestock activities to look at dairy farming versus ranching and intensive versus extensive activities would improve understanding of the persistent prevalence of cattle ranching in the face of low profitability (Börner and Wunder, 2008a).

Soybean production, on the other hand, is highly profitable at between US$1027–1924/ha. However, the expansion of soybean is limited by climatic and soil condition - soybean is only suitable for cultivation in a portion of the Amazon. Proximity to transport infrastructure and markets is extremely important for producers. For example, soybean is a major driver of deforestation in Mato Grosso which has a large road network and is close to large markets (and has a higher deforestation rate). Agricultural production in more remote Amazonas is based on more varied production of food crops, fruits, fibres and some cash crop production (coffee). The profitability of subsistence crops is low with manioc and rice (US$2/ha) and perennials and bananas (US$2/ha) the least profitable.

The impact of soybean production on the costs of REDD is significant. Nepstad *et al.* (2007) calculated that to eliminate deforestation completely would cost US$1.49/ton CO_2e, but that to reduce deforestation to 94 percent of projected levels would cost only half that at US$0.76/ton CO_2e. The six percentage point difference is in large part attributable to highly profitable soybean production. Similarly, Vera Diaz and Schwartzman (2005) estimate the cost of eliminating deforestation to be US$5.44/ton CO_2e with soybeans and US$2.34/ton CO_2e without soybeans.

The opportunity cost for tree plantations (US$2378/ha), coffee-bandarra production (US$2172/ha), and coffee-rubber (US$969/ha) is also relatively high. However, the land area used for tree plantations and tree crops is very small, accounting for only one percent of land use. Returns generated from subsistence crops are low with manioc and rice (US$2/ha) and perennials and bananas (US$2/ha) being the least profitable.

Returns to one-off timber harvesting vary significantly from US$24/ha in low productivity forest in Amazonas to US$1435/ha in high productivity forest in the Amazon. However, timber harvest followed by cattle ranching and soybeans is highly profitable in high productivity areas (US$3465/ha) and low productivity areas (US$2215/ha). The impact of including one-off timber harvest followed by ranching depends on the productivity of the forests to be cut down; Vera Diaz and Schwartzman (2005) estimate returns of almost US$1700/ha for high productivity logging followed by ranching versus less than US$450 for logging low productivity forest followed by ranching.

2.5 Carbon content

Estimates of the opportunity cost of forest conservation are adjusted by the carbon content (ton carbon/ha) of the landscape to derive an estimated cost/ton carbon or CO_2e. Assuming carbon dioxide emissions from deforestation equal the total carbon content of above-ground vegetation, the opportunity cost per ton of avoided carbon dioxide emissions is equal to per hectare opportunity cost divided by average carbon content. While there is significant variability in the carbon content between and within provinces and regions, most studies use either averages by province (Börner and Wunder, 2008a) or for the Amazon.

For carbon content per hectare of primary forest in Amazonas, Börner and Wunder (2008a) use a conservative estimate (110 tons C per ha) from Houghton *et al.* (2001) which is based on data from seven independent studies. For Mato Grosso, more detailed carbon content data were available locally. Vera Diaz and Schwartzman (2005) assume average carbon content of 155 tons C per ha (based on estimates ranging from 121 to 397 tons C per ha). Table 6 presents carbon content estimates based on work done under the Ilha do Bananal forest carbon project in the Brazilian Amazon. The carbon content data is based on micro-level sampling for different types of forest in the Amazon.

Table 6: Carbon stock by vegetation typology in Brazil

Carbon stock	Vegetation typology (ton/ha)				
		Upland forest	Flooded forest	Cerrados	Floodplain fields
Total		**111.7**	**98.6**	**31.5**	**6.8**
Dead OM		6.6	7.6	3.6	1.8
Carbon in trees	Trunk	44.2	33.9	4.0	*
	Branch	29.4	25.0	4.1	*
	Leaves	3.6	3.9	1.0	*
	Bark	5.3	4.6	0.7	*
	Roots	*	*	2.4	*
	Total	82.7	67.4	12.0	8.6
Carbon in bushes	Trunk	8.0	7.3	4.7	*
	Branch	12.0	14.7	8.2	0.5
	Leaves	2.5	1.6	3.0	0
	Grass	*	*	*	4.7
	Total	22.5	23.6	15.8	5.2

Source: Reproduced with minor corrections from May *et al.* (2004).

Nepstad *et al.* (2007) use location specific carbon stock data for each forest pixel based on a map developed by Saatchi *et al.* (2007). The forest carbon map in Figure 6 illustrates the variation in per hectare carbon stock in the Amazon.

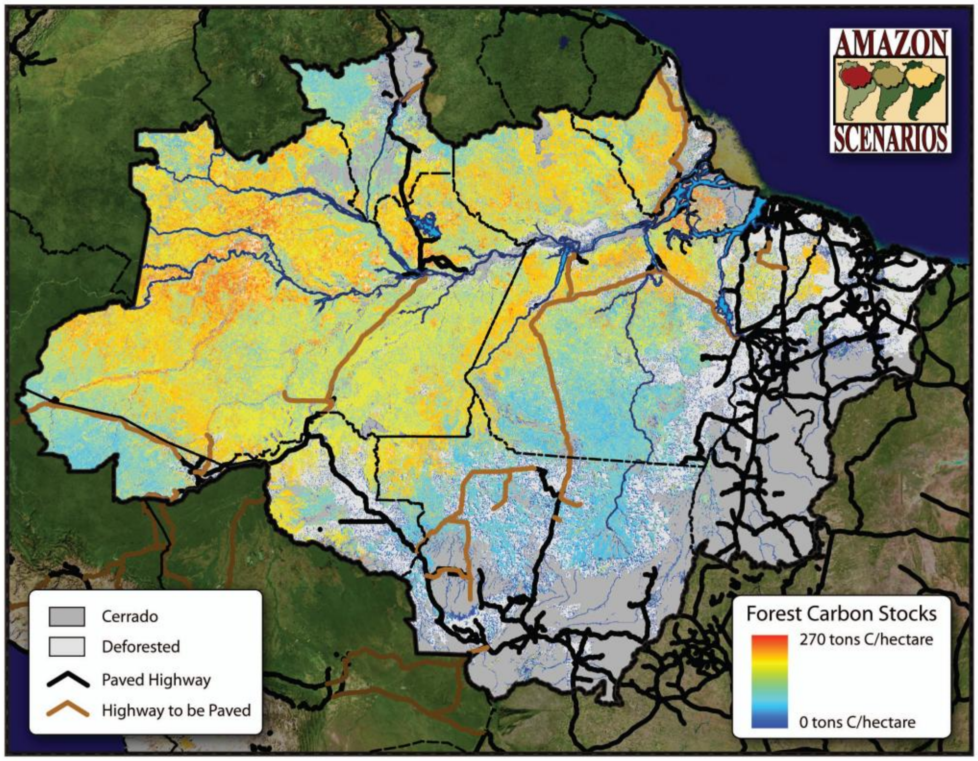


Figure 6: Forest carbon stocks of the Brazilian Amazon
Source: Reproduced from Nepstad *et al.*, 2007

2.6 Opportunity cost per ton CO_2e avoided

Combining per hectare opportunity costs and carbon content data, estimates for cost/ton C and cost/ton CO_2e are derived for a specific locality. As some studies express opportunity cost in terms of cost per hectare (Grieg-Gran, 2006; Tomich, 2005), this study has converted these estimates by using the lowest per hectare carbon content estimate for the Amazon included in this review, 110 ton C/hectare, which converts to roughly 400 ton CO_2e/ha (using the standard conversion rate of 1 ton C = 3.67 ton CO_2e).

The opportunity costs of forest conservation expressed in terms of the cost per ton CO_2e are summarised in Table 7 along with the relative importance of different land uses on recently deforested land. These costs are very low, ranging from almost nothing for small scale beef cattle production to between US$2.5 and $3.5/ton CO_2e for soybean production. Tree plantations appear to have high returns, but cover only a small area of the Amazon.

Table 7: Opportunity costs by land use in the Brazilian Amazon (US$2005)

Land use	Opportunity cost $/ton CO_2e		
	Low	High	Share in area of deforested land (%)
Subsistence agriculture	0	1.1	4
Ranching	0	2.6	77-85
Soybean	2.5	3.4	5-8
Tree plantations	1.1	5.9	1
Timber harvest	0.3	2.5	na
Timber+ranching+soybean	3.9	6.1	na

Source: See Table A.1 in Annex 1, various sources, Grieg-Gran (2006).

As roughly 80 percent of recently deforested land is used for ranching, the scope for achieving cost-effective reductions in CO_2e emissions through avoided deforestation seems promising. Figure 7 illustrates how opportunity costs vary significantly between provinces in the Brazilian Amazon. More remote Amazonas which has little transport infrastructure and hence much less land under soybean production has far lower opportunity costs than Mato Grosso. Börner and Wunder (2008a) estimate a supply curve which shows that more than a third of deforestation in Amazonas could be compensated for less than US$1/ton CO_2e, and there is no land use that could not be compensated for less than US$3/ton CO_2e. In Mato Grosso, opportunity costs are higher, but it is still possible to avoid half of deforestation for less than US$3/ton CO_2e; the supply curve is relatively flat but increases steeply to completely avoid deforestation (maximum of US$12/ton CO_2e).

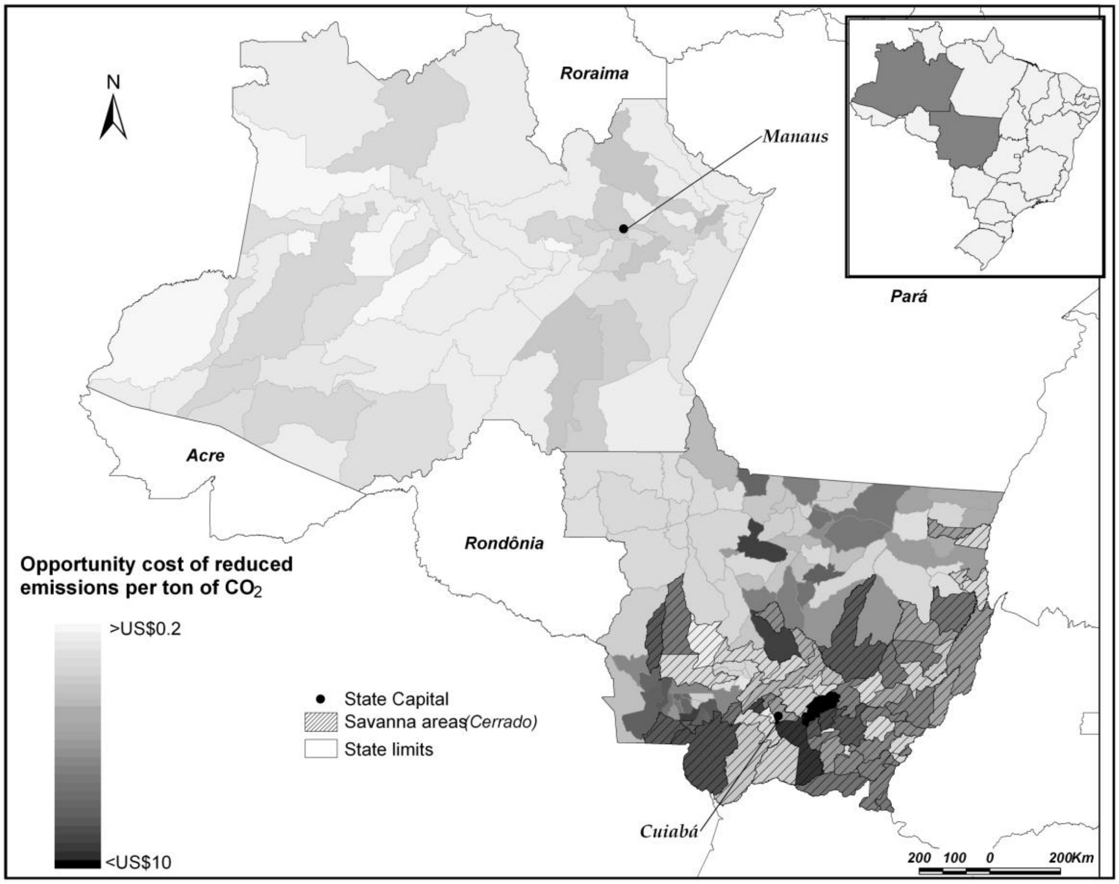


Figure 7: Opportunity cost $/ton CO_2e in Amazonas and Mato Grosso
Source: Reproduced from Börner and Wunder, 2008a

Not all the estimates discussed are directly comparable. In particular, Nepstad *et al.* (2007) estimate net NPV; the NPV of timber production is subtracted from each model since timber production maintains most of the carbon stock of a forest. The net opportunity cost is calculated by dividing the difference in NPV (soy or cattle minus timber) by the difference in the carbon stock of agriculture/livestock versus timber. The study assumes sustainable logging is implemented and decreases carbon stocks by 15 percent while soybean and pasture production reduce carbon stock by 85 percent. The results for soybean, cattle and logging are presented in Figures 8–10.

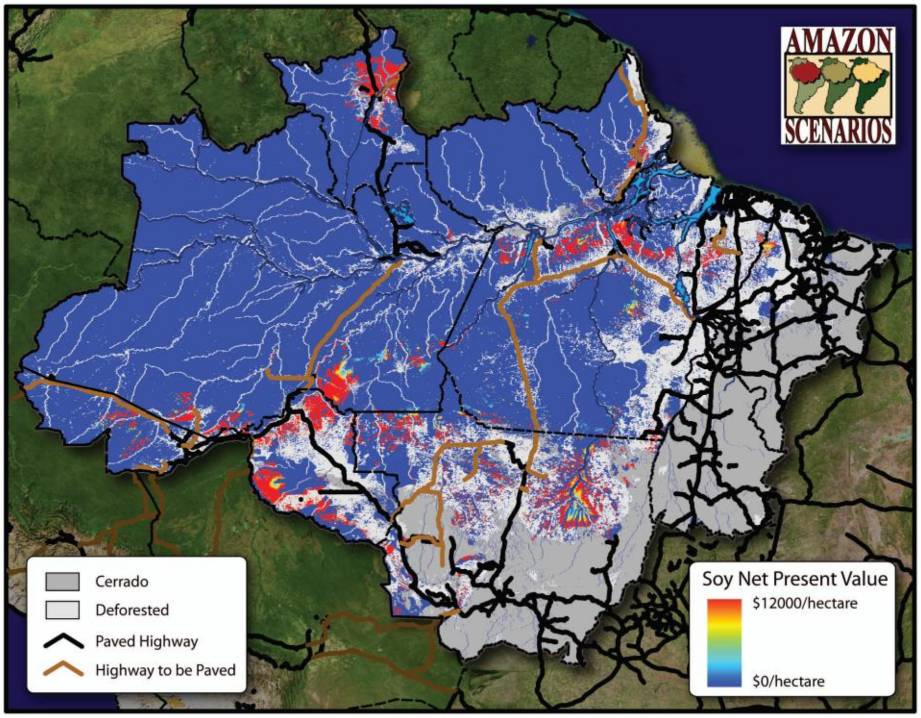


Figure 8: Potential net present value (2007–37) of soy production on the forested lands of the Brazilian Amazon
Source: Reproduced from Nepstad *et al.*, 2007 *(http://whrc.org/Brazilcarbonsupplement)*

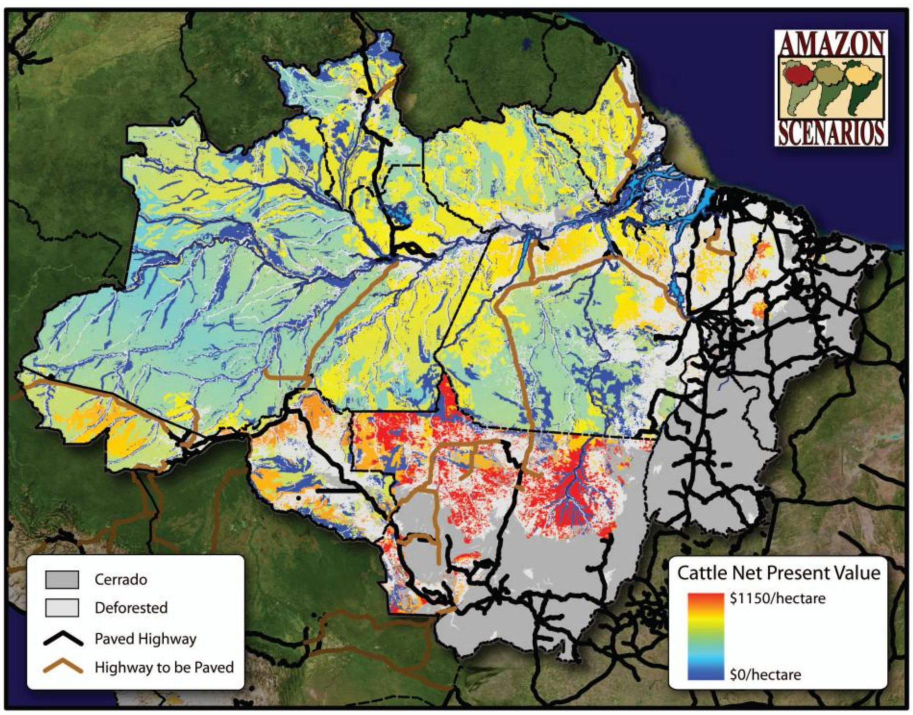


Figure 9: Potential net present value of cattle production in the Brazilian Amazon
Source: Reproduced from Nepstad *et al.*, 2007 *(http://whrc.org/Brazilcarbonsupplement)*

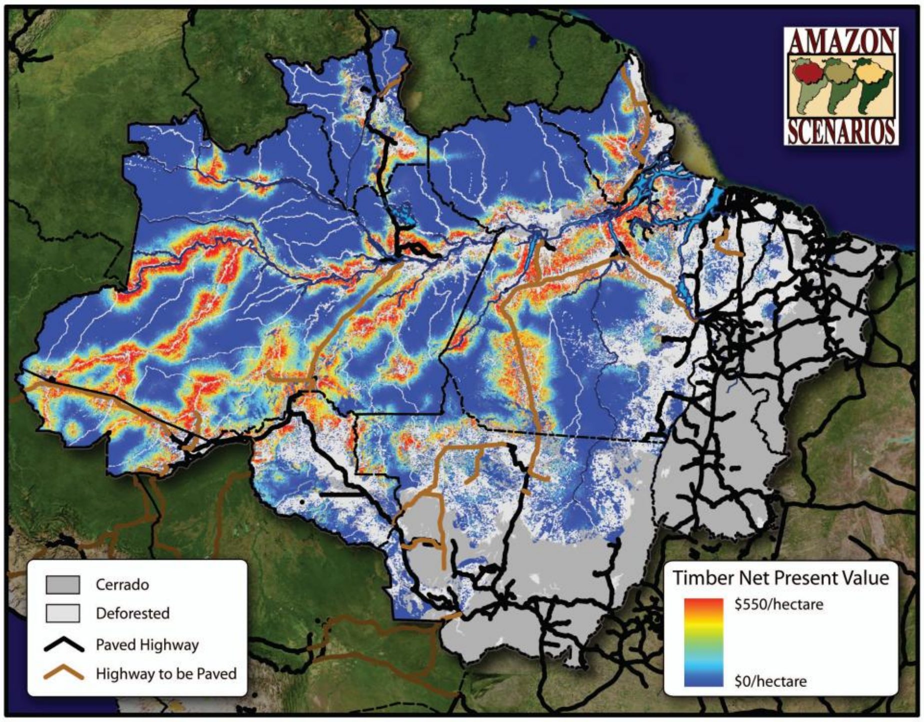


Figure 10: The potential net present value (2007–2037) of sustainable timber production in the Brazilian Amazon
Source: Reproduced from Nepstad *et al.,* 2007

Chomitz *et al.* (2007) contrast different types of agricultural or agro-forestry land use with the maintenance of secondary forest, sustainable use of primary forest and community forestry. The study shows how deforestation would be unprofitable in some parts of many countries and farming systems at very modest (<US$3/ton CO_2e.) carbon prices. Figure 11 provides a summary of likely costs of REDD for early adopters and includes Brazil and Indonesia.

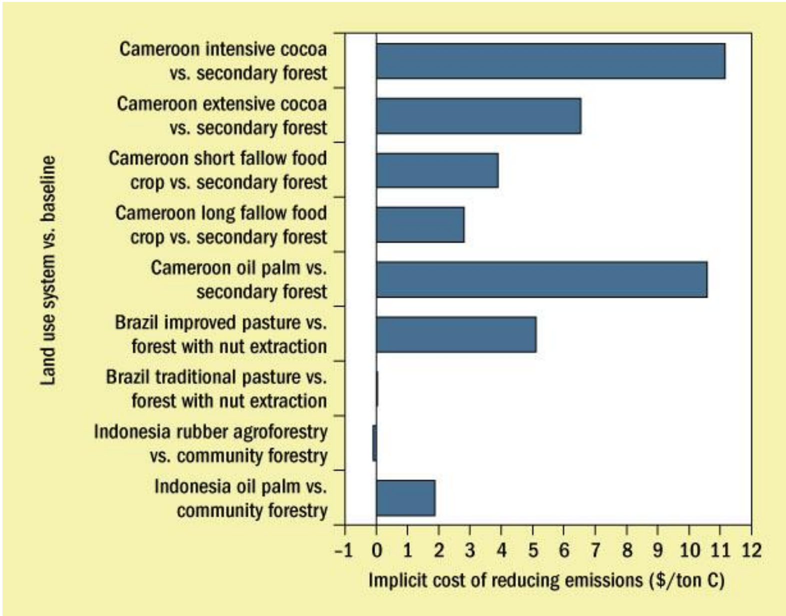


Figure 11: Deforestation would be unprofitable in many land systems at modest carbon prices
Source: Reproduced from Chomitz *et al.*, 2007

To sum up, the studies reviewed here suggest that at current carbon prices in both voluntary and compliance markets, REDD can compete with most prevalent land uses in the Amazon, namely logging and cattle ranching. REDD is somewhat less competitive with soybean production. Due to significant variation in deforestation rates on private land, net returns to different land uses and the carbon content of forest across the Amazon, the analysis of local level data is essential to accurately assess the costs of REDD.

2.7 Implementation and transaction costs

Table 8 summarises the types of implementation and transaction costs that may be associated with REDD and comments on the current status of REDD institutions and prerequisite conditions for the Amazon.

Table 8: REDD implementation and transaction costs and implications for REDD in the Amazon

Implementation and transaction costs category	Comments
1. Information and procurement	Currently, carbon markets are not prepared for large-scale REDD in the Amazon and carbon buyers have traditionally been reluctant to invest in carbon projects in the forestry sector. Procurement costs can therefore be expected to be significant.
2. Scheme design and negotiation	Large-scale REDD schemes may incur significant negotiation costs, especially if they contemplate payments from national government budgets that need to be negotiated with civil society.
3. Implementation	Existing organisations and institutions needed to be strengthened and systems like SLAPR implemented in all areas covered by REDD. Establishing and running payment mechanisms (especially in the case of direct payments to landowners) are likely to contribute the lion's share to this cost item.
4. Monitoring	In some states, rural licensing systems are in place that would allow annual deforestation monitoring at farm-level scales. The technology for satellite-based deforestation monitoring is relatively well developed and much more cost-effective than ground based monitoring.
5. Enforcement and protection	Enforcement costs might be considerably reduced by delivering payments only after verification of effectively avoided deforestation. Given weakly enforced property rights in large parts of the Amazon, enforcing theses rights (e.g. in and around protected areas) might prove crucial to assuring additionality of REDD and, hence, represent a relevant source of transaction costs.
6. Verification and certification (Approval)	These cost items have been shown to be an important barrier for small-scale carbon forestry projects (Cacho *et al.*, 2005), but are expected to decrease with project size.

Source: Reproduced from Börner and Wunder, 2008a.

Implementation and transaction costs are discussed in Section 1. A rough empirical estimate of US$1/ton CO_2e is used here based on a number of PES carbon sequestration and watershed management projects, GHG mitigation project implementation, simulations and the costs of implementing Sustainable Forest Management (SFM).

There is a little empirical data on current transaction costs in the Amazon, but not enough to derive an estimate specific to the Amazon. Transaction costs have been roughly calculated to be US$12 per hectare in Mato Grosso which is equal to US$0.04/ton CO_2e (Börner and Wunder, 2008b). In another study, Wunder and Alban (2008) estimate transaction costs in ongoing user financed PES schemes in the Brazilian Amazon, find high start up costs of US$17–186/ha and low recurrent transaction costs.

Adding US$1/ton CO_2e to the opportunity cost estimates discussed above increases the costs of avoided deforestation significantly. For the highest opportunity cost estimate above, high productivity forest timber harvest followed by ranching and soybean production, implementation and transaction costs increase the costs of REDD from US$6.1 to 7.1/ton CO_2e. Recall that transaction costs are likely to be greater for smaller projects than for bigger projects (assuming emissions cuts are correlated with project size) and also greater for a large number of small transactions versus a smaller number of larger transactions.

2.8 Capacity

Boucher (2008) assesses Brazil's capacity to implement and monitor a REDD programme or projects to be strong. Brazil has monitored deforestation based on remote sensing since the 1970s, and currently has a sophisticated system that can monitor short-term changes in deforestation rates. Furthermore, it has a strong scientific base with highly trained scientists, and good universities and research institutes. Politically it is a well-established democracy with the ability to regulate land use at state and federal level. And it ranks highly in terms of governance indicators, e.g. regulatory control and quality.

More specifically, in terms of technical capacity, the Project for Gross Deforestation Assessment in the Brazilian Legal Amazon (PRODES) monitors forest cover based on satellite and other remote sensing data. It has developed predictive models for land use change and is able to monitor near real time deforestation using coarse resolution (MODIS) (Hoare *et al.*, 2008). On the other hand, forest inventory data are scarce and have not been updated since the 1970s.

3 Case study 2: Costs of REDD in Indonesia

In 2007, Indonesia became the third largest emitter of GHGs globally. Roughly 85 percent of Indonesia's emissions are due to deforestation, forest degradation and forest fires. Indonesia is the world's top emitter of GHGs associated with the draining of peatlands which is responsible for over five percent of annual global emissions of GHGs from human activities (Joosten, 2009). The rate of deforestation has increased in recent years from 1.61 percent per year (1990–2000) to 1.91 percent per year (2000–2005) while the annual loss of primary forest has increased by 25 percent over the same period. Clearly, Indonesia's extensive tropical forest cover is threatened with rapid degradation and conversion.

3.1 Main drivers of deforestation

The direct causes of deforestation and forest degradation in Indonesia include logging for timber, the establishment of large-scale tree crop estates and industrial timber plantations, smallholder farming, internal migration and government-sponsored resettlement. Industrial timber plantations mainly supply the pulp and paper industry. Rising commodity prices have accelerated the conversion of forest for the production of cash crops, notably palm oil. Illegal logging is a significant problem, while forest fires destroyed over five million hectares of forest in 1994 and another 4.6 million hectares in 1997–98.

There are also important indirect drivers of deforestation in Indonesia. Over the last few decades, rapid economic growth has seen the emergence of a powerful class of private landowners whose interests are often in conflict with small-scale land users (Swallow et al., 2007). The fall in value of the Indonesian currency during the Asian financial crisis in 1997 provided additional incentives to convert forest to export tree crops, such as oil palm, rubber, cocoa and coffee. Competition between migrants, indigenous people and large-scale investors accelerates deforestation on islands with greater population density, e.g. Sumatra. Logging is a powerful driver of forest degradation on some other islands, e.g. Kalimantan.

3.2 Types of forest land – deforestation versus degradation

Including incentives to reduce forest degradation in REDD is particularly important for Indonesia, where forest degradation may be a larger source of GHG emissions than forest conversion. Indonesia contains one-half of the world's tropical peatlands, which are extremely rich in carbon. In recent decades, these ecosystems have been widely cleared and converted to oil palm, fast-growing tree plantations for the pulp and paper industry, large scale irrigated rice production and small scale agriculture. Large carbon emissions occur when peatlands are burned or drained. In a study of three provinces (East Kalimantan, Jambi and Lampung), Swallow *et al.* (2007) find that the economic returns from the conversion of peatlands are very low, while carbon emissions are very high. The conservation of peatlands is thus a very low opportunity-cost carbon abatement option, which has attracted wide attention as a priority for REDD investment.

3.3 *Opportunity costs*

3.3.1 Carbon content estimates

Table 9 presents empirical data from the Alternatives to Slash and Burn Partnership (ASB) study of above-ground carbon content by land use in Indonesia. According to this study, undisturbed forest contains 300 tons above-ground carbon per ha. However, according to the FAO's Forest Resource Assessment (FRA) the average carbon stock contained in living biomass in undisturbed forest in South and Southeast Asia is 89 tons/ha. The ASB estimate is more than three times the FAO estimate for Asia and is at least double the estimates used in the Brazilian Amazon (between 110 and 155 ton carbon/ha). There is no clear explanation why these estimates should be so different other than the ASB estimates are site specific for three provinces. Both estimates exclude soil carbon, but include carbon contained in living biomass, dead wood and litter.

Table 9: Above-ground time-averaged carbon stocks by land use in Indonesia

	Land use type	Time averaged carbon stock (ton C/ha)
1	Undisturbed forest	300
2	Logged over forest – high density	250
3	Logged over forest – low density	150
4	Undisturbed mangrove	200
5	Logged over mangrove	100
6	Undisturbed swamp forest	200
7	Logged over swamp forest	200
8	Home garden	21.8
9	Coconut	90.7
10	Damar agro-forest	114.8
11	Fruit-based agro-forest	116.1
12	Rubber agro-forest	62.1
13	Cinnamon agro-forest	60.0
14	Coffee agro-forest	17.2
15	Rubber	46.8
16	Oil palm	31.0
17	Tea plantation	7.5
18	Natural re-growth shrub	26.8
19	Sugarcane	12.5
20	Agriculture	11.9
21	Rice field	1.0
22	Grass	2.0
23	Settlement	4.1
24	Open peat	4.1
25	Cleared land	3.9

Source: Reproduced from Swallow *et al.* (2007).

As the opportunity cost estimates per ton carbon are highly sensitive to carbon content of forest, this paper uses both ASB (high carbon scenario) and FAO (low carbon scenario) carbon content estimates and looks at the implications for costs of REDD. The returns per ton carbon are adjusted and expressed as returns per ton CO_2e using the standard conversion factor (1 ton carbon = 3.67 ton CO_2e).

3.3.2 Some regional and local estimates

Appendix 1, Table A1.2 presents empirical estimates of financial returns to agricultural activities that cause deforestation and forest degradation in Indonesia. Roughly 30 percent of deforestation is due to the establishment of palm oil plantations, most of which are large scale. Smallholder rubber is responsible for another 30 percent, while rice and cassava production are responsible for 20 percent each (Grieg-Gran, 2006).

Per hectare opportunity costs are presented by land use. Returns to land are adjusted by the net change in carbon storage per hectare directly resulting from land use change. Carbon content data are taken from the ASB study (high carbon content scenario) and the FAO FRA (low carbon content scenario). The net change in carbon is defined as the difference in the carbon content of undisturbed forest and the carbon content of the alternative land use for the high carbon content scenario. For the low carbon content scenario, data on carbon content of alternative land uses is not available. It is assumed that the carbon content of alternative land uses is zero.

Table 10 summarises opportunity costs by land use expressed in US$/ton CO_2e. The highest opportunity cost for forest land is associated with palm oil production with estimates ranging from US$0.49/ton CO_2e for small holder farming in Sumatra up to US$19.6/ton CO_2e for the conversion of degraded forest land to palm oil in Indonesia. The opportunity cost of palm oil production on degraded land is very high expressed in carbon equivalent units because the carbon content of degraded land is low and hence the opportunity cost per hectare (large number) divided by carbon content lost (small number) is a large number. It should be noted that the highest estimates are from the Indonesian submission to the World Bank Forest Carbon Partnership Facility (FCPF) and are based on 2008 palm oil prices which were almost 50 percent *above* long term average palm oil prices for 1990–2000 and almost double palm oil prices in the first quarter of 2009. This study does not adjust downwards these estimates due to uncertainty regarding the future evolution of palm oil prices. It should also be noted that the estimates of the Indonesian submission to the FCPF have not been reviewed here in detail as it was not possible to obtain the report itself – figures attributed to the FCPF submission are collected from media reports (Thomson Reuters News, 04.03.09). The carbon content estimate for peatland, 277 ton/ha, is low and is in large part responsible for the high opportunity cost estimates. If the carbon content of peatland is increased from 277 ton C/ha to 600 ton C/ha, the opportunity cost of converting peatlands to palm oil falls from US$4.29/ton CO_2e to US$1.85/ton CO_2e, even when taking into account the carbon stored in palm oil plantations (30 ton C/ha in Table 9).

Table 10: Opportunity cost estimates in Indonesia by land use

Land use		Opportunity cost $/ton CO_2e			
		Low carbon content (a)		High carbon content (b)	
		Low	High	Low	High
Oil Palm	**Large scale**		6.3		2.1
	Supported growers		5.1		1.7
	High yield independent		4.4		1.5
	Low yield independent		1.8		0.6
	Smallholder		0.5		0.2
Rubber		0	4.2	0	1.6
Subsistence agriculture		0	1.53	0	0.47
Logging		3.82 (Sumatra)	7.96 (SEA and Pacific)	1.65 (Sumatra)	3.44 (SEA and Pacific)

Source: See Table A1.2 in Annex 1

Lower estimates of the opportunity cost of palm oil production (Vermeulen and Goad, 2006 based on 2005 data; Tomich, 2005 based on 1997 data) made in years with palm oil prices *below* long term average palm oil prices have been adjusted upwards to ensure that any bias in the estimates is upwards, i.e. opportunity cost estimates presented here would be on the high rather than the low side. Large scale, supported growers and high yield palm oil production have opportunity costs below US$3/ton CO_2e in the high carbon scenario and range from US$4.42 to US$6.3/ton CO_2e in the low carbon scenario.

Logging (unsustainable) is the next most profitable land use. Prices for hard and softwood have not been adjusted in the analysis as log prices have been historically stable and at the time of data collection of the various studies were 8–27 percent *above* the three year average of 2004–2007. Assuming carbon content of undisturbed forest at 300 ton/ha, the implicit cost is US$1.65/ton CO_2e for commercial logging in Sumatra and US$3.44/ton CO_2e for unsustainable commercial logging in Southeast Asia and the Pacific. These estimates are low compared to global estimates produced (Boucher, 2008) and compared to the costs of non-forest based carbon abatement (see Beinhocker *et al.,* 2008; Vattenfall, 2007; McKinsey & Company (2009). However, assuming the carbon content of undisturbed forest is 89 ton/ha and the carbon content of the cleared land is zero, the cost per ton CO_2e rises to US$3.82 for commercial logging in Sumatra and US$7.96 for Southeast Asia and the Pacific.

Returns to rubber production have been increased by 35 percent to compensate for low rubber prices at the time of opportunity cost estimation. The profitability of rubber plantations appears to depend on whether improved genetic material is used. Data from Tomich *et al.* (2005) indicates that the opportunity cost of smallholder agroforest rubber production without the use of clonal planting material is close to zero, but US$1.55/ton CO_2e with clonal planting material (high carbon scenario). In the low carbon scenario, assuming 89 ton carbon/ha forest increases the opportunity cost for rubber plantations with improved planting material to US$4.18/ton CO_2e which is higher than logging in Indonesia.

Subsistence agriculture and cattle ranching generate very low rates of return; most estimates are close to zero (and negative in some cases) due to the low per hectare returns and the low carbon content of these land uses. Assuming 89 ton carbon content of undisturbed forests raises the highest estimate for a mixed shifting cultivation and small scale agriculture to US$1.53/ton CO_2e.

The sensitivity of the results with regards to the carbon content estimates used for both undisturbed forest and for land use following deforestation underscores the need for project based REDD to assess local level carbon stocks as there is significant variation even within forests. The importance of the carbon content of the changed land use also highlights the opportunities for win-win land use change (Tomich *et al.,* 2005). For example, the use of undisturbed forest for sustainable logging and agro-forestry may result in relatively small losses of carbon and is associated with the highest opportunity costs (per hectare returns are divided by a small change in carbon stock). The impact on biodiversity and other ecosystem services may be more significant.

3.3.3 Province level estimates from the ASB study

ASB (Swallow *et al.,* 2007) provides detailed site-specific information on the opportunity costs of forest conversion for three provinces in Indonesia. The study presents spatially explicit analyses of tradeoffs between carbon sequestration and economic returns. The three sites are characterised by different population density/forest cover ratios to assess the "forest transition" hypothesis of an inverted-U relationship between deforestation and population growth; population growth accompanies deforestation and economic development for some time, and then levels off before a stage of slow but continuous afforestation.

To derive the cost of abating a ton of CO_2e, the ASB study looks at changes in land use to quantify the changes in economic returns per hectare and net changes in carbon sequestered (Table 9). The study finds that while the land use changes make financial sense to those making them, expressed in terms of tons of CO_2e, the economic gains associated with deforestation are very low. In the three provinces, 6–20 percent of the area where emissions increased due to land use change have generated returns of less than US$1/ton CO_2e and 64–92 percent of the area generated returns of less than US$5/ton CO_2e. The results of the ASB report are sensitive to the inclusion of peatlands due to their extremely high carbon content. It is argued that peatlands should be given special attention in negotiations and design of REDD. Traditional slash-and-burn systems on peat soils produce very small economic returns (US$0.10–0.20/ton CO_2e). Site specific information is presented below.

3.3.3.1 East Kalimantan Province

The province of East Kalimantan represents a highly forested, low population density area. East Kalimantan covers an area of 220,400 km^2 on Borneo and is sparsely populated. A network of rivers still provides the basis of the transport system. The province is 79 percent forested, and logging and harvesting are the dominant extractive uses. While a few oil palm plantations have been established, most have been abandoned due to conflict between investors and local communities. Subsistence agriculture is based on crop-fallow rotations. Rubber agro-forests are not common (as they are in other parts of Indonesia).

Figure 4.5: Summary of land use change in East Kalimantan Province, Indonesia for 1990, 2000 and 2005

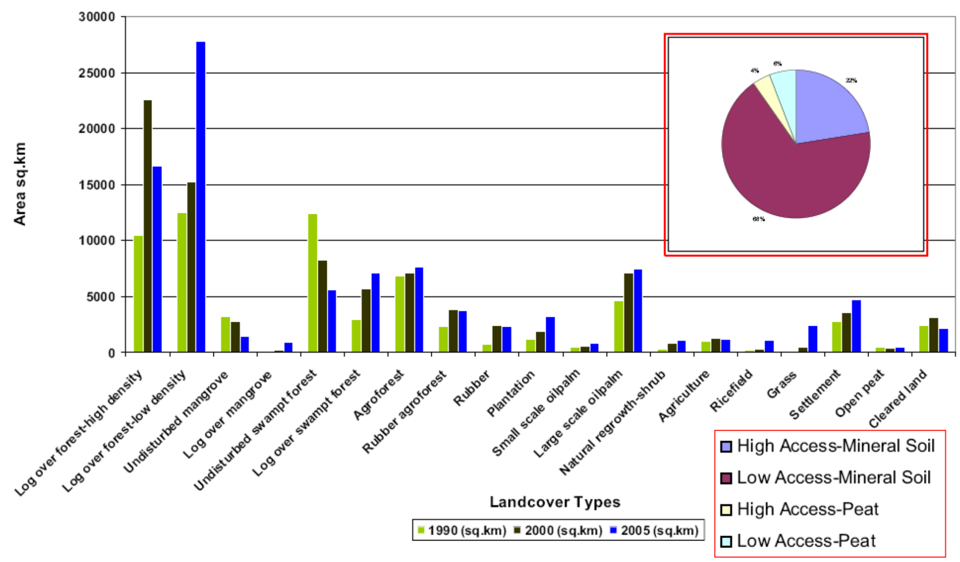


Figure 12: Summary of land use change in East Kalimantan Province, Indonesia, 1990, 2000, 2005

Source: Reproduced from Swallow *et al.,* 2007

Most carbon emissions in East Kalimantan are due to logging. However, because of low accessibility to transport infrastructure and markets, returns are relatively low at less than US$5/ton CO_2e (Figure 13). The results are shown using both private and social prices. In this review, we are concerned with private (financial prices) as the objective it to have a more precise idea of the financial costs of REDD.

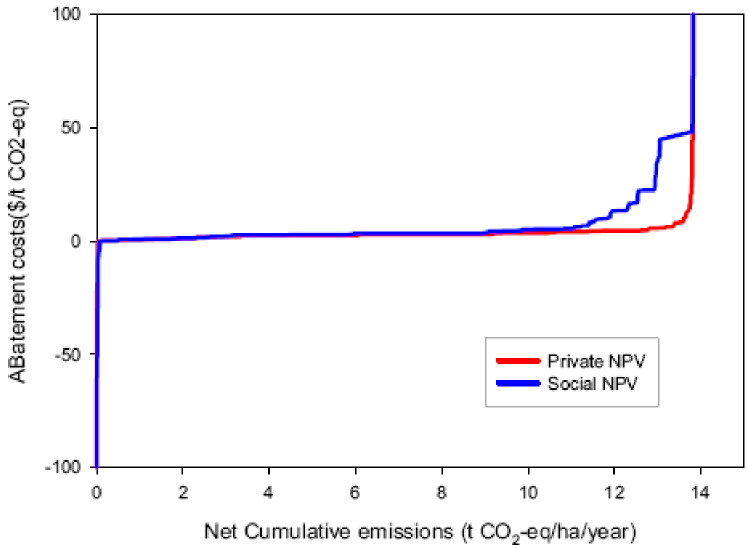


Figure 13: Abatement costs with private and social NPV for East Kalimantan
Source: Reproduced from Swallow *et al.,* 2007

3.3.3.2 Jambi Province

Jambi province in central Sumatra covers 55,000 km². Population density is 39 people per km² and 45 percent of the province is easily accessible by water or road. There is significant and ongoing conversion of forest to high value commercial crops like oil palm and rubber. There is both government sponsored and spontaneous transmigration occurring, further increasing forest conversion to agriculture and settlement. Only 34 percent of the province is still forested.

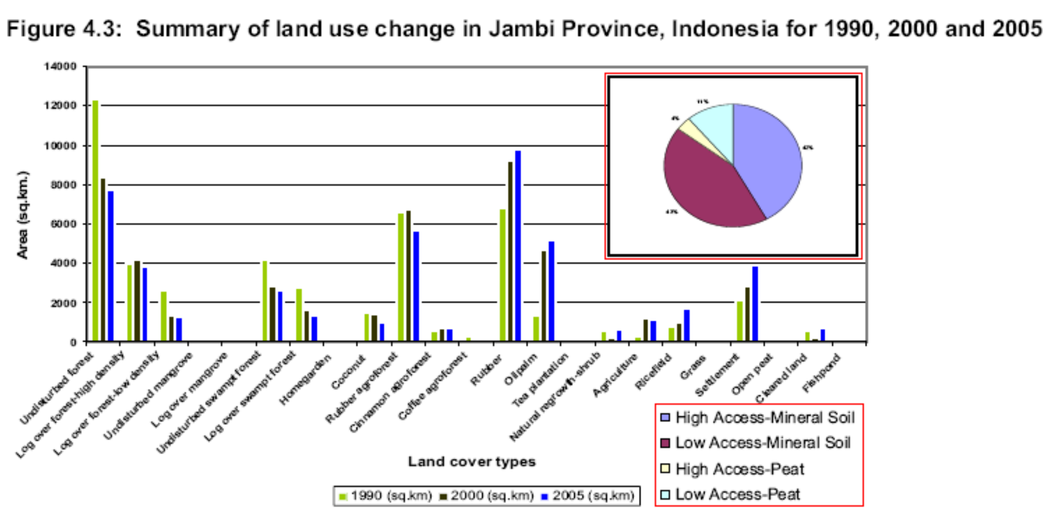
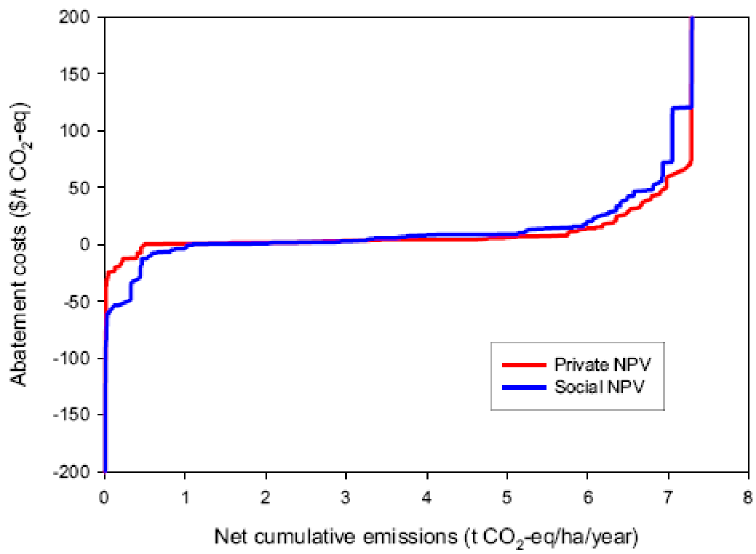


Figure 14: Summary of land use change in Jambi, Indonesia, 1990, 2000, 2005
Source: Reproduced from Swallow *et al.,* 2007

Figure 15 indicates that, excluding peatlands, land use change in Jambi has produced economic benefits. Accessibility to roads and markets is an important factor. The conversion of logged over forest to palm oil generate financial returns less than US$5/ton CO_2e in areas with low accessibility, but higher than US$5/ton CO_2e in highly accessible areas. However, the inclusion of peatlands dramatically reduces the opportunity cost per ton CO_2e with a greater percentage of carbon emissions associated with returns lower than US$2.50/ton CO_2e. (Unfortunately it is difficult to read the numeric values of the figures due to the scale used in the original paper).



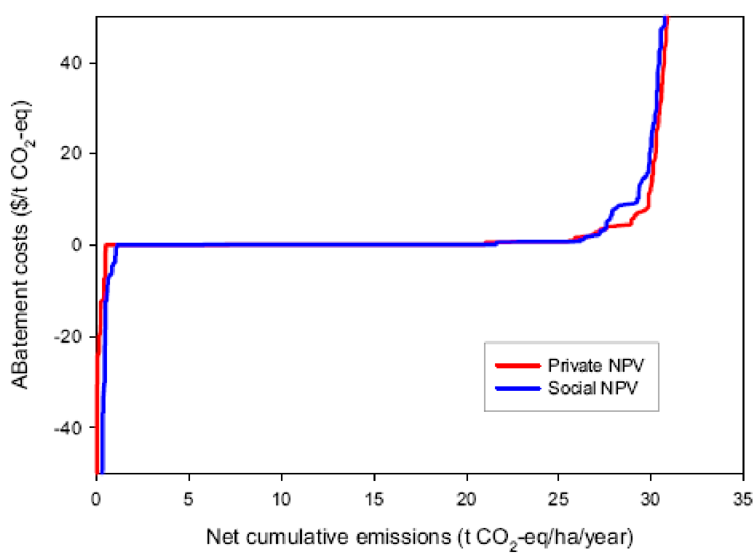


Figure 15: Abatement costs with private and social NPV for Jambi (a) without emissions from peat and (b) with emissions from peat
Source: Reproduced from Swallow *et al.,* 2007

3.3.3.3 Lampung Province

Lampung province in southern Sumatra has only eight percent of its area still under forest due to government sponsored and spontaneous migration from Java and large scale commercial logging in the 1970s, followed by conversion to industrial plantations of sugar cane and pineapple. Cassava is the main crop in the lowlands. In the foothills, coffee has increased along with international coffee prices. Indigenous Lampung people still cultivate semi-permanent food crops on flooded river banks. Shifting cultivation is no longer practised. Migrants grow rice, but due to drought and soil erosion, people are leaving the area.

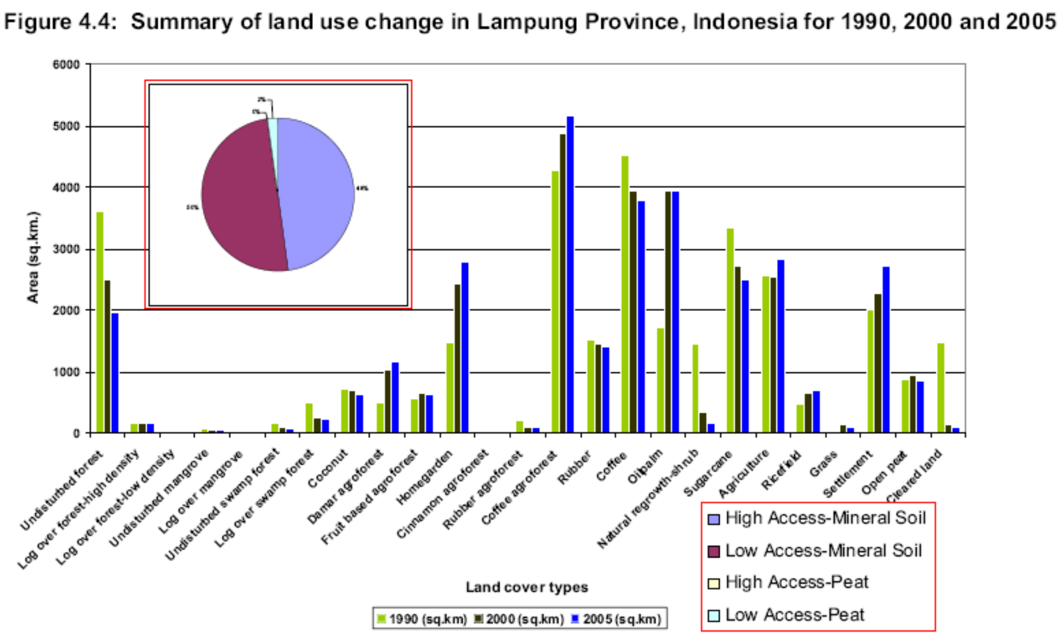


Figure 16: Summary of land use change in Lampung, Indonesia, 1990, 2000, 2005
Source: Reproduced from Swallow *et al.* (2007)

Figure 17 below presents the results for Lampung where the main source of deforestation is logging and conversion to multi-strata coffee. Opportunity costs are low at less than US$5/ton CO_2e. Illegal logging is a large problem here; between 1990 and 2000, 48 percent of forest conversion was illegal and between 2000 and 2005, 82 percent was illegal (Swallow *et al.,* 2007). There are strong economic incentives for claiming land depending on the price of coffee and oil palm (coffee has a relatively high price elasticity and oil palm extremely high). However, because the only forest land remaining is in the national park, there is high pressure on this protected area.

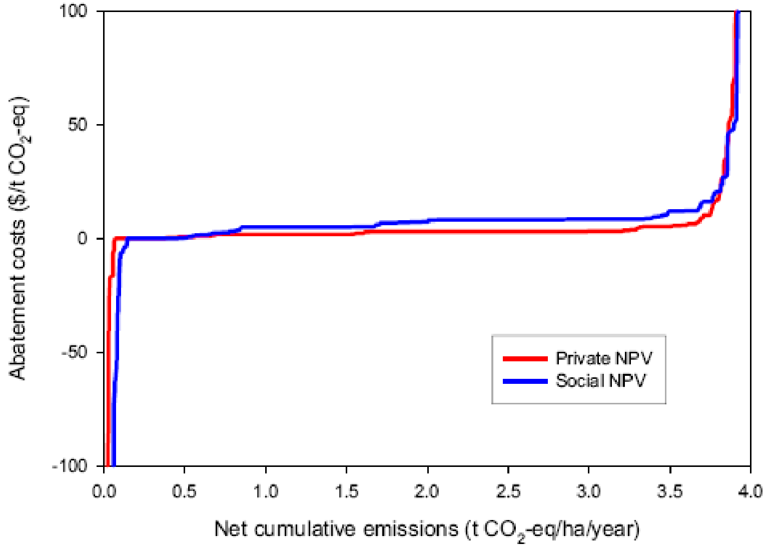


Figure 17: Abatement costs with private and social NPV for Lampung
Source: Reproduced from Swallow *et al.,* 2007

3.3.4 Other opportunity cost estimates

Research at the World Bank also shows clearly the low opportunity costs of forest conversion in Indonesia (Chomitz *et al.*, 2007). Figure 11 presented earlier illustrates returns to changes in land use expressed in US$/ton CO_2e. The lower section of the table indicates that shifting from community forestry to rubber agroforestry or oil palm in Indonesia would be unprofitable even at modest (significantly less than US$1) carbon prices.

In short, most studies based on local data, indicate that the opportunity costs of REDD in Indonesia are generally less than US$5/ton CO_2e with opportunity costs of degraded land between US$5–10/ton CO_2e. The highest costs are associated with the palm oil, rubber and commercial logging. Subsistence agriculture has low returns. The section below looks at the implications of adding implementation and transaction costs on to the opportunity of forest conservation in Indonesia to move towards looking at the full potential cost of REDD.

3.4 Implementation and transaction costs

A rough estimate of US$1/ton CO_2e is calculated based on existing studies of PES schemes in Brazil, Costa Rica and Mexico, carbon abatement projects and global estimates of the cost of implementing SFM. Adding US$1/ton CO_2e to the opportunity cost estimates discussed above increases the costs of avoided deforestation significantly, in some cases. However, the costs of abating carbon emissions based on REDD activities remain below $10/ton CO_2e.

4 Costs of REDD and other climate mitigation options

Empirical evidence on the financial returns to alternative land uses on recently deforested land in Brazil and Indonesia suggests that avoiding emissions from deforestation may provide a cost-effective climate mitigation option. The financial returns to a number of land uses, expressed in terms of net profits per ton of CO_2 e, are below current market prices for carbon. In other words, forest carbon can provide attractive investment opportunities simply from a financial perspective. Moreover, due to the large variation in opportunity costs within forest-rich countries, there appears to be significant scope to achieve efficient outcomes by allowing trade in REDD obligations across land users, while focusing REDD interventions on avoiding the conversion of forest to low-return agricultural uses.

However, there are a number of studies that have estimated the abatement costs of REDD to be higher than the estimates reviewed here (Beinhocker *et al.,* 2008; McKinsey & Company (2009); UNFCCC, 2007). A number of these studies are based on global partial equilibrium models of the forest sector which simulate the dynamics of the world economy. There are three major global partial equilibrium models: GTM, DIMA and GCOMAP[4] which use the same underlying data as more micro-level models, but differ in which sectors they include, the dynamics simulated, interest rates and data sets on carbon content and deforestation rates. These models produce unit costs of abatement that are significantly higher than the on-the-ground empirical estimates reviewed here. These global models take into account the level of emissions abatement (Boucher, 2008).

Table 11 summarises average opportunity costs estimates from different sources using different methodologies. The table is adapted from Boucher (2008) who provides average opportunity costs as well as the range of estimates. We provide only a range of estimates reviewed in this paper as these estimates should be compared only very loosely due to different methodologies and assumptions in the calculations of NPVs by land use (discount rates, assumptions regarding carbon content of competing land use, etc). The Boucher study obtained the raw data from regional/empirical studies and redid the analysis in a standardised manner. It then compared the results to those of the Stern Review and the three main global models. Implementation and transaction costs are not included here.

[4] For GTM see Sohngen and Sedjo (2006), Sohngen and Mendelsohn (2006). For DIMA see Kindermann et al. (2006) and Marland and Obersteiner (2007, 2008). For GCOMAP see Sathaye et al. (2006) and Anger and Sathaye (2008).

Table 11: Opportunity costs for REDD from different sources

Approach	Land use	Opportunity cost estimate $/ton CO_2 e		
		Average	High	Low
Global models (various)		11.26	17.86	6.77
Stern Review (2006)		5.52	8.28	2.76
Regional, empirical (various)		2.51	4.18	0.84
This review:				
Brazil	Ranching		2.6	0
	Soybean		3.4	2.5
	Subsistence agriculture		1.1	0
	Timber+ranching+ soybean		6.1	3.9
Indonesia – high carbon scenario	Palm oil		4.29	0.18
	Subsistence agriculture		0.47	0
	Logging		3.44	1.65
Indonesia – low carbon scenario	Palm oil		19.6	0.5
	Subsistence agriculture		1.53	0
	Logging		7.96	3.82

Source: Adapted from Boucher (2008), including estimates from this review.

McKinsey & Company (2009) use global estimates to compare the cost-effectiveness of a range of carbon abatement opportunities across all sectors. Figure 18 presents a cost curve of abatement opportunities, focusing on abatement opportunities that are cost-effective relative to current market prices for carbon. The market price for carbon on the European Union Emissions Trading Scheme (EU ETS) was US$35/ton CO_2e in the first quarter of 2008 (Ecosystem Marketplace). The figure looks at abatement opportunities and costs in 2030 whereas this review focuses on current (roughly from 2000–2007, expressed in 2005 US$) abatement costs.

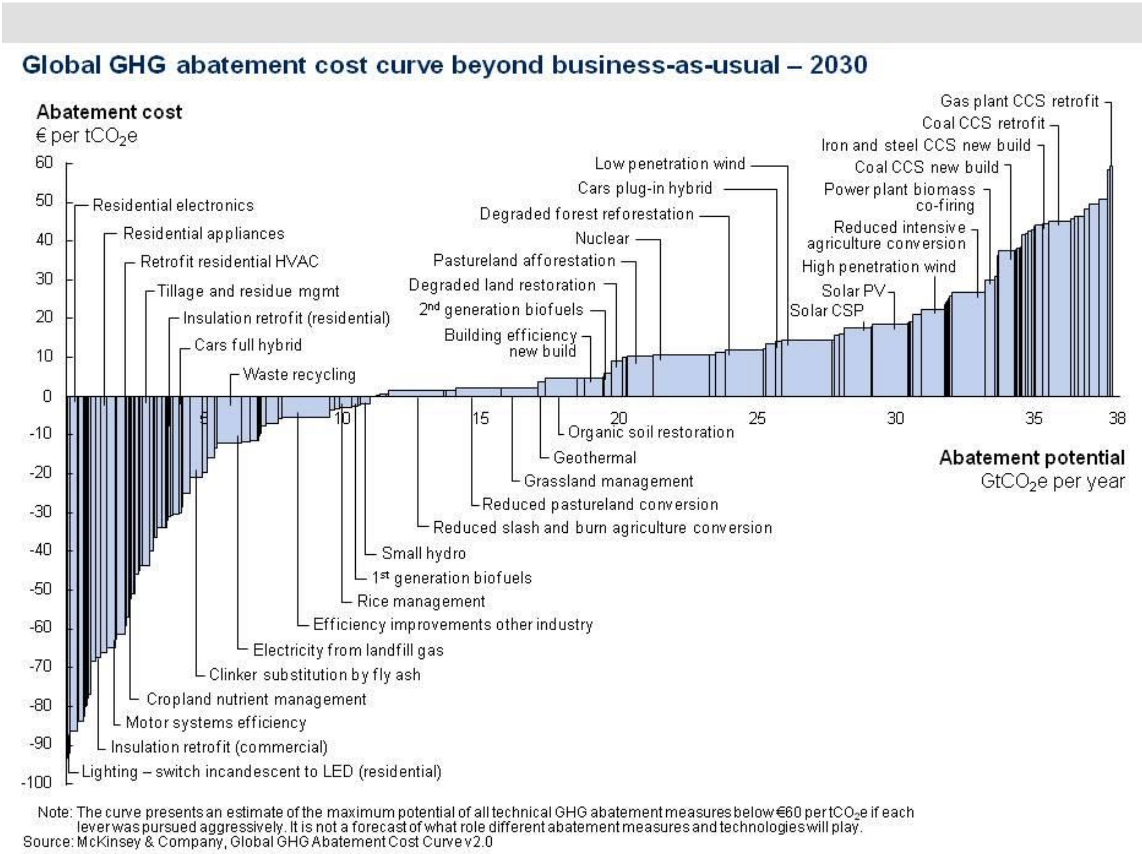


Figure 18: Global cost curve of GHG abatement opportunities beyond business as usual
Source: McKinsey & Company (2009)

The estimates reviewed in this study (US$ estimates are converted to euro at the 2005 exchange rate of US$1=1.25 EUR) are consistent with McKinsey & Company (2009) estimates for reduced slash-and-burn agriculture and reduced pastureland conversion at less than EUR 5/ton CO_2 e. These abatement options are more cost effective than many non-forestry sector abatement opportunities such as solar energy, wind energy, and carbon capture and storage.

Figure 19 looks more closely at the costs of abatement opportunities within the forest sector. Abatement based on reduced slash-and-burn agriculture and reduced pastureland conversion is more cost effective than other forest sector abatement options, e.g. the restoration of degraded land, afforestation of pastureland, and reforestation of degraded forest. In line with this review, McKinsey & Company (2009) find that the costs of abatement based on the reduced conversion of forest to intensive agriculture are higher and cannot compete with solar and wind power, for example. However, it must also be acknowledged that the data in Figures 18 and 19, and indeed many of the published estimates of abatement costs, do not measure risk consistently, i.e. the reliability of different abatement strategies.

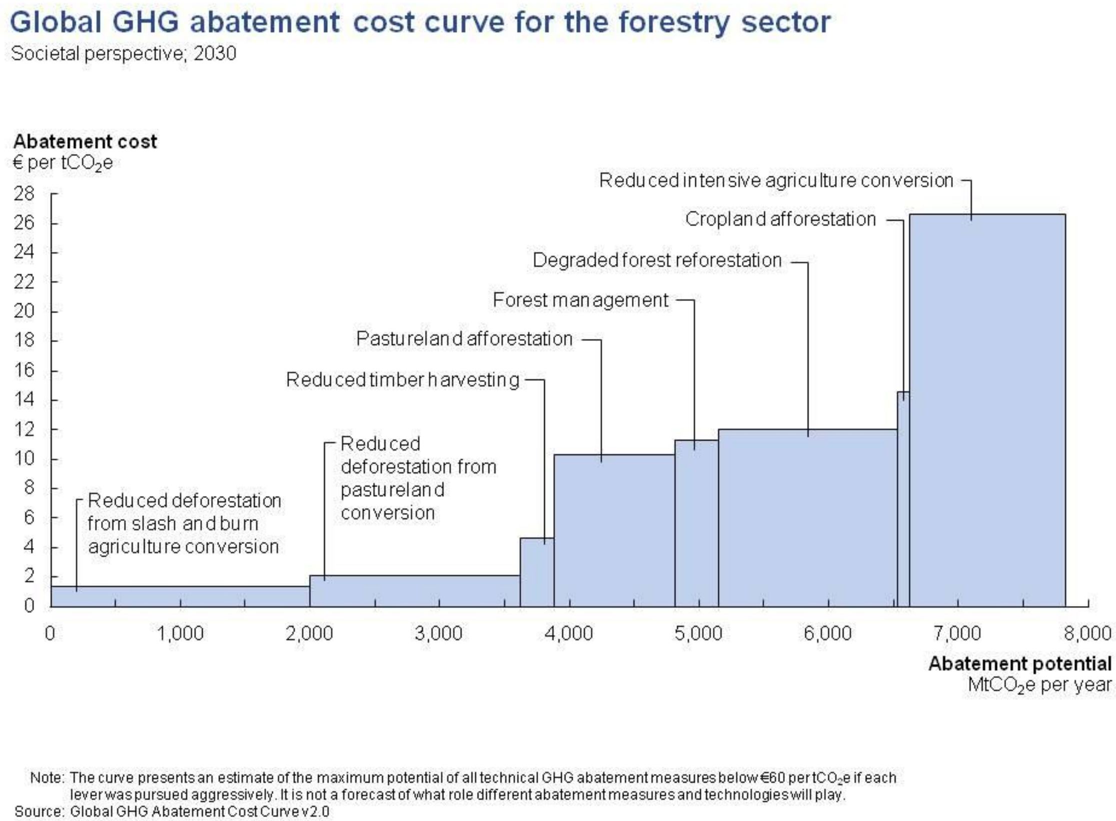


Figure 19: Abatement cost curve for the forestry sector 2030
Source: McKinsey & Company (2009)

There is a wide range of estimates of the costs of carbon abatement strategies in the forest sector and of REDD in particular. Much of the difference is due to the fact that micro-level estimates, based on particular local conditions, more accurately capture variation in local opportunity costs. This type of information is critical to guide public and private investors seeking to develop forest carbon projects and REDD activities in particular areas. For many stakeholders, global estimates and regional averages do not provide sufficiently accurate estimates of the relevant costs and risks.

The key findings of this review may be summarised as follows:

- There is significant variation in per hectare opportunity costs in Brazil and Indonesia, reflecting differences in local conditions, land use and proximity to transport infrastructure and markets. National, regional and global averages are of limited usefulness in determining where REDD is most cost-effective.
- There is significant variation in the carbon content of forest land at national, provincial and local level. Moreover, there is some inconsistency between published estimates of carbon content, based on the application of different methodologies. It

is therefore critical not only to estimate local opportunity costs, but also to measure carbon content on a local basis.

- A review of empirical opportunity cost estimates suggests that REDD is competitive with most land uses in the Brazilian Amazon and many land uses in Indonesia at a carbon price of less than US$5/ton CO_2e. REDD is competitive with most land uses in Indonesia at US$10/ton CO_2e. Subsistence agriculture and most livestock production systems are characterized by very low returns in both Brazil and Indonesia. Logging and cash crops generally exhibit higher opportunity costs.
- While implementation and transaction costs add roughly US$1/ton CO_2e to opportunity costs, these additional costs are not so large as to make REDD (or other forest carbon activities) financially unattractive relative to non-forest sector carbon abatement options.

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Appendix 1

Table A1.1: Opportunity cost estimates by land use for the Brazilian Amazon

Region	Land use	Total net return US$/ha	NPV in 2005 US$/ha A	Adjusted (2) NPV in 2005 US$/ha C=A*(1+B)	Carbon content ton C/ha D	OC per ton C US$/ton C E=C/D	OC US$/ton CO2e (1) F=E/3.67	Source/methodology	Year of estimate	Commodity price in year of estimate, $ G	Long term avg price 2000-2009 $ H	Change in price from baseline year (%) B=(H-G)/G
Amazon	Beef cattle medium/large scale		390	461	110	4.2	1.1	Grieg-Gran 2006 uses Margulis 2003. Average of five representative farms in Para, Rondonia, Mato Grosso	2003	90	106	18%
Amazon	Beef cattle small scale		2	2	110	0.0	0.0	Grieg-Gran 2006 uses Lewis et al. 2002 (ASB Brazil)	2002	95	106	11%
Amazon	Dairy		154	194	110	1.8	0.5	Grieg-Gran 2006 uses Arima and Uhl, 1997. Cost of clearing assumed at US$100/ha.	1997	84	106	26%
Amazonas	Extensive cattle ranching		660	614	110	5.6	1.5	Börner and Wunder, 2008. Profitability taken from Margulis, 2004, assuming low end estimate for Amazonas.	2004	114	106	-7%
Mato Grosso	Extensive cattle ranching		684	636	110	5.8	1.6	Börner and Wunder, 2008. Profitability taken from Margulis, 2004, assuming high end estimate for Mato Grosso.	2004	114	106	-7%
Amazon	Cattle ranching		264	332	155	2.1	0.6	Vera Diaz and Schwartzman 2005.	2005	84	106	26%
Amazon	Traditional pasture		2	3	110	0.0	0.0	Grieg-Gran cites Tomich et al. 2005. Results of ASB matrix, returns to land.	1996	81	106	31%
Amazon	Improved pasture		789	1033	110	9.4	2.6	Grieg-Gran cites Tomich et al. 2005. Results of ASB matrix, returns to land.	1996	81	106	31%
South America	Cattle ranching - large scale		572	501	140	3.6	1.0	Robledo and Blaser 2008. Estimates range between US$10-660/ha.	2008	121	106	-12%

1/standard conversion rate of 1 ton C = 3.67 ton CO2e equivalent
2/adjusted by percentage value in column O to take account of commodity price changes

Region	Land use	Total net return US$/ha	NPV in 2005 US$/ ha	Adjusted (2) NPV in 2005 US$/ha	Carbon content ton C/ha	OC per ton C US$/ton C	OC US$/ton CO2e (1)	Source/methodology	Year of estimate	Commodity price in year of estimate, $	Long term avg price 2000-2009 $	Change in price from baseline year (%)
			A	C=A*(1+B)	D	E=A/D	F=E/ 3.67			G	H	B=(H-G)/G
Amazon	Soybeans		1899		155	12.3	3.3	Grieg-Gran 2006 uses Vera Diaz and Schwartzman, 2005. Returns to farmers in Mato Grosso. Assume US$100/ha clearing cost.	2005	281	240	-15%
Mato Grosso	Cash crops (soybeans)		1027		110	9.3	2.5	Börner and Wunder, 2008.	2008	453	240	-47%
South America	Commericial crops (soybean, palm oil, pulp)		2214		140	15.9	4.3	Robledo and Blaser 2008.	2008	453	240	-47%
Amazon	Soybeans		1924		155	12.4	3.4	Vera Diaz and Schwartzman 2005.	2005	281	240	-15%
Amazon	One-off timber harvesting	236			110	2.1	0.6	Grieg-Gran 2005. Average stumpage fee in Paragominas in 1995, Stone 1996 in Barreto et al 1998.	1995	258	213	-17%
Amazon	One off logging - high timber productivity scenario	1435			155	9.3	2.5	Vera Diaz and Schwartzman 2005. Logging revenue is based on average timber stumpage value ($1400) plus average land rent ($35). HTP scenario = 40m3 timber/ha.	2005	238	213	-11%
Amazon	One-off logging - low timber productivity scenario	185			155	1.2	0.3	Vera Diaz and Schwartzman 2005. Logging revenue is based on average timber stumpage value (150) plus average land rent ($35). LTP scenario = 15m3 timber/ha.	2005	238	213	-11%
Mato Grosso	Timber extraction	109-734			110	1-6.7	0.3-1.8	Börner and Wunder, 2008.	2008		213	
Amazonas	Timber extraction	24-791			110	0.3-7.2	0-2	Börner and Wunder, 2008.	2008	291	213	-27%

1/standard conversion rate of 1 ton C = 3.67 ton CO2e equivalent

2/adjusted by percentage value in column O to take account of commodity price changes

Region	Land use	Total net return US$/ha	NPV in 2005 US$/ha A	Adjusted (2) NPV in 2005 US$/ha C=A*(1+B)	Carbon content ton C/ha D	OC per ton C US$/ton C E=A/D	OC US$/ton CO2e (1) F=E/3.67	Source/methodology	Year of estimate	Commodity price in year of estimate, $ G	Long term avg price 2000-2009 $ H	Change in price from baseline year (%) B=(H-G)/G
Amazon	Tree plantations		2378		110	21.6	5.9	Grieg-Gran 2005 assumes same as for coffee-bandarra system in Lewis et al 2002 (ASB Brazil)	2002			
Amazon	Managed forestry		462		110	4.2	1.1	Grieg-Gran cites Tomich et al. 2005. Results of ASB matrix, returns to land.	1996			
Amazon	Coffee-bandarra		2172		110	19.7	5.4	Grieg-Gran cites Tomich et al. 2005. Results of ASB matrix, returns to land.	1996			
Amazon	Coffee-rubber		969		110	8.8	2.4	Grieg-Gran cites Tomich et al. 2005. Results of ASB matrix, returns to land.	1996	185	94	-49%
Amazonas	Cash crops (coffee)		618		110	5.6	1.5	Börner and Wunder, 2008.	2008	138	94	
Amazon	Manioc/rice		2		110	0.0	0.0	Grieg-Gran 2005 assumes same as for pasture. Negative is ASB report.				
Amazon	Perennials/ bananas		2		110	0.0	0.0	Grieg-Gran 2005 assumes that perennials, fallow and degraded land have same return as manioc/rice.				
Amazonas	Food crops (corn)		452		110	4.1	1.1	Börner and Wunder, 2008. Due to fallow periods, NPV for staple crops less than for cattle production despite equal average annual returns.				
Amazonas	Fruits (water melons)		374		110	3.4	0.9	Börner and Wunder, 2008.				
Amazonas	Fibres (malva)		292		110	2.7	0.7	Börner and Wunder, 2008.				
Amazon	Annual fallow		130		110	1.2	0.3	Grieg-Gran cites Tomich et al. 2005. Results of ASB matrix, returns to land.				
Amazon	Improved fallow		2285		110	20.8	5.7	Grieg-Gran cites Tomich et al. 2005. Results of ASB matrix, returns to land.				
South America	Small scale agriculture/shifting cultivation		323		140	2.3	0.6	Robledo and Blaser 2008. Based on US$/day income assessment on 1 ha of prodcutive area.				
South America	Fuelwood and NTFP gathering		231		140	1.6	0.4	Robledo and Blaser, 2008.				

1/standard conversion rate of 1 ton C = 3.67 ton CO2e equivalent
2/adjusted by percentage value in column O to take account of commodity price changes

Region	Land use	Total net return US$/ha	NPV in 2005 US$/ha	Adjusted (2) NPV in 2005 US$/ha	Carbon content ton C/ha	OC per ton C US$/ton C	OC US$/ton CO2e (1)	Source/methodology	Year of estimate	Commodity price in year of estimate, $	Long term avg price 2000-2009 $	Change in price from baseline year (%)
			A	C=A*(1+B)	D	E=A/D	F=E/3.67			G	H	B=(H-G)/G
Amazon	Logging + cattle ranching (high timber productivity scenario)		1699		155	11.0	3.0	Vera Diaz and Schwartzman 2005. Logging revenue is based on average timber stumpage value ($1400) plus average land rent ($35). HTP scenario = 40m3 timber/ha.				
Amazon	Logging + cattle ranching (low timber productivity scenario)		449		155	2.9	0.8	Vera Diaz and Schwartzman 2005. Logging revenue is based on average timber stumpage value (150) plus average land rent ($35). LTP scenario = 15m3 timber/ha.				
Amazon	Timber harvest + cattle ranching + soybeans (high timber productivity scenario)	1435	3465		155	22.4	6.1	Vera Diaz and Schwartzman 2005. Logging revenue is based on average timber stumpage value ($1400) plus average land rent ($35). HTP scenario = 40m3 timber/ha.				
Amazon	Timber harvest + cattle ranching + soybeans (low timber productivity scenario)	185	2215		155	14.3	3.9	Vera Diaz and Schwartzman 2005. Logging revenue is based on average timber stumpage value (150) plus average land rent ($35). LTP scenario = 15m3 timber/ha.				

1/standard conversion rate of 1 ton C = 3.67 ton CO2e equivalent

2/adjusted by percentage value in column O to take account of commodity price changes

Table A1.2 : **Estimated land use returns in Indonesia using high carbon content estimates and adjusting for carbon content of new land use**

Country or region	Land use	NPV in 2005 US$	C in baseline scenario ton C /ha (b)	C in new land use ton C /ha (b)	Net change in carbon ton C /ha	OC per ton C US$/t on C (c)	OC US$/t on CO2e (a)(c)	C in baseline scenario ton C /ha (d)	OC per ton C US$/ton C (c)	OC US$/ton CO2e (a)(c)	Source/methodology
				high carbon scenario				low carbon scenario			
	Commercial agriculture										
Indonesia	Oil palm - large scale	2059	300	31	269	7.65	2.09	89	23.13	6.30	Vermeulen and Goad 2006 based on data from Zen et al 2005.
Indonesia	Oil palm - supported growers	1670	300	31	269	6.21	1.69	89	18.76	5.11	Vermeulen and Goad 2006 based on data from Zen et al 2005.
Indonesia	Oil palm - high yield independent	1443	300	31	269	5.36	1.46	89	16.21	4.42	Vermeulen and Goad 2006 based on data from Zen et al 2005.
Indonesia	Oil palm - low yield independent	592	300	31	269	2.20	0.60	89	6.65	1.81	Vermeulen and Goad 2006 based on data from Zen et al 2005.
Sumatra	Oil palm small holder	160	306	62	244	0.66	0.18	89	1.80	0.49	Tomich et al. 2005. Results of ASB matrix, private prices, 20% discount rate.
Indonesia	Palm oil on degraded forest on mineral soil	3595	50	31	19	189.21	51.56	50	71.90	19.59	Indonesia submission to World Bank FCPF cited in Reuters News 04.03.09.
Indonesia	Palm oil on peat land	3869	277	31	246	15.73	4.29	89	43.47	11.85	Indonesia submission to World Bank FCPF cited in Reuters News 04.03.09.
Indonesia	Palm oil on peat land	3869	600	31	569	6.80	1.85	277	13.97	3.81	Our estimate using data of Indonesia submission to World Bank FCPF, but increasing carbon content of peatland from 277 to 600 tC/ha.
SE Asia and Pacific	Commercial crops- palm oil, soybean, rubber, pulp, copra, groundnut, pineapple	2377	300	31	269	8.84	2.41	89	26.71	7.28	Robledo and Blaser, 2008.
Indonesia	Rubber - smallholder	48	300	47	253	0.19	0.05	89	0.54	0.15	Grieg Gran 2006 based on Tomich et al 1998 (ASB Indonesia).
Sumatra	Rubber agroforest	1.1	306	79	227	0.00	0.00	89	0.01	0.00	Tomich et al. 2005. Results of ASB matrix, private prices, 20% discount rate.
Sumatra	Rubber agroforest with clonal planting material	1366	306	66	240	5.69	1.55	89	15.35	4.18	Tomich et al. 2005. Results of ASB matrix, private prices, 20% discount rate.
SE Asia and Pacific	Cattle ranching (large scale)	314	300	2	298	1.05	0.29	89	3.53	0.96	Robledo and Blaser, 2008.

(a) standard conversion rate of 1 ton C = 3.67 ton CO2e equivalent

(b) carbon content data for high carbon content scenario are based on Swallow et al. 2007 above ground time-averaged carbon stock in Indonesia, except for Tomich 2005 (own estimates) and Indonesian submission to FCPF, 2008.

(c) OC is opportunity cost, (d) carbon content data for low carbon content scenario are based on FAO Forest Resource Assessment 2005 (above ground carbon/ha for South and SE Asia.

Country or region	Land use	NPV in 2005 US$	C in baseline scenario ton C /ha (b) high carbon scenario	C in new land use ton C /ha (b)	Net change in carbon ton C /ha	OC per ton C US$/ton C (c)	OC US$/ton CO2e (a)(c)	C in baseline scenario ton C /ha (b) low carbon scenario	OC per ton C US$/ton C (c)	OC US$/ton CO2e (a)(c)	Source/methodology
	Subsistence Agriculture										
SE Asia and Pacific	Small-scale agriculture/shifting cultivation	499	300	12	288	1.73	0.47	89	5.61	1.53	Robledo and Blaser, 2008. Mix of small scale agriculture with market access and shifting cultivation, $1/day income per day (CIFOR)
Indonesia	Rice fallow	26	300	1	299	0.09	0.02	89	0.29	0.08	Grieg Gran 2006 using Tomich et al 1998 (ASB Indonesia) Social prices, 20% discount rate.
Indonesia	Cassava monoculture	18	300	12	288	0.06	0.02	89	0.20	0.06	Grieg Gran 2006 using Tomich et al 1998 (ASB Indonesia) Social prices, 20% discount rate.
Sumatra	Upland rice-bush fallow	-62	306	37	269	-0.23	-0.06	89	-0.70	-0.19	Tomich et al. 2005. Results of ASB matrix, private prices, 20% discount rate.
Sumatra	Continuous cassava + Imperata	60	306	2	304	0.20	0.05	89	0.67	0.18	Tomich et al. 2005. Results of ASB matrix, private prices, 20% discount rate.
SE Asia and Pacific	Fuelwood and NTFP gathering	285	300	30	270	1.06	0.29	89	3.21	0.87	Robledo and Blaser, 2008. Based on high-priced, unsustainable harvesting of NTFP (important driver of degradation). CIFOR data.
	Wood extraction										
SE Asia and Pacific	Commercial non-sustainable logging (legal and illegal)	2601	300	94	206	12.62	3.44	89	29.22	7.96	Robledo and Blaser, 2008. ITTO data.
Sumatra	Commercial logging	1248	300	94	206	6.06	1.65	89	14.02	3.82	Grieg Gran 2006 using Tomich et al 1998 (ASB Indonesia) Social prices, 20% discount rate.
Sumatra	Commercial logging	1011	306	94	212	4.77	1.30	89	11.36	3.10	Tomich et al. 2005. Results of ASB matrix, private prices, 20% discount rate.
SE Asia and Pacific	Fuelwood/charcoal (traded)	95	300	30	270	0.35	0.10	89	1.07	0.29	Robledo and Blaser, 2008.

(a) standard conversion rate of 1 ton C = 3.67 ton CO2e equivalent. (b) carbon content data for high carbon content scenario are based on Swallow et al. 2007 above ground time-averaged carbon stock in Indonesia, except for Tomich 2005 (own estimates) and Indonesian submission to FCPF, 2008. (c) OC is opportunity cost
(d) Carbon content data for low carbon content scenario are based on FAO Forest Resource Assessment 2005 (above ground carbon/ha for South and SE Asia.

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