How Much Is the Amazon Worth?

The State of Knowledge Concerning the Value of Preserving Amazon Rainforests

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Abstract

This paper surveys the current state of knowledge concerning the value of the Amazon rainforest, including a survey of work to date to quantify changes in economic values when the rainforest cover changes. The focus is on local and regional impacts of forest loss or protection, including both gross values of forest protection and opportunity costs of converting the forest to other uses including agriculture. Important gross value items surveyed are timber and non-timber product extraction from a sustainably maintained rainforest; local values of eco-tourism; biological resources including bio-

prospecting; a range of hydrological impacts including watershed protection, hydropower production, and changes in rainfall patterns; and impacts of forest fires and their control. Mapping such values in geographical space is of high value for implementing efficient and effective (Reducing Emissions from Deforestation and Forest Degradation) programs for protecting the remaining forest. The current data basis for such mapping is found to be quite weak and in need of improvement for all value elements.

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1. Introduction

1.1 The Need to Measure the Gains from Forest Protection

The question in the title of this paper (“How much is the Amazon worth?”) has intrigued scholars and investors for quite some time. Over the past decades, this topic has been subject to considerable debate, in particular within Brazil, but also in other countries in the Amazon region, which encompasses parts of Peru, Colombia, Bolivia, Ecuador, Venezuela, Guyana, Suriname, and French Guyana. One concern relates to conflicts over alternative land uses – a debate that is key to the overhaul in progress of the Brazilian national Forest Code: *i.e.*, what is the relative value of the Amazon forest, standing or cut down for alternative use? The implication here is that both protection values and opportunity values (e.g., in the form of agricultural production on deforested lands or managed timber extraction) must be correctly accounted for. A wider issue is which parts of the Amazon forest have particularly high protection value and the extent to which some parts of the forest area should rather be used for alternative activities such as agriculture. A connected concern is the implementation of programs to protect threatened forest, such as REDD+ within which implementation of Payment for Environmental Services (PES) programs is contemplated. The potential interest of parties outside the region (e.g., donors) to contribute to Amazon forest protection is also likely to be influenced by the magnitude of demonstrated protection values.

Economic valuation must concern itself with the issues of “for what?”, “to whom?”; “when?”; the geographical range (“where?”) over which such values are expected to apply; and the degree to which (and how) such values differ across geographical space. The full set of values that must go into an assessment of Amazon protection measures is global; it is values accruing to the local and regional (Amazon, and wider South American region), and the rest of the global population (outside this region). This paper discusses only local and regional values, and not the additional global ones. Local protection values alone will in some cases be sufficient to outweigh the (usually, all local) alternative use values; but in other cases they may not. If not, conversion to other uses than forest would appear to be the preferable choice, were decisions based only on local opportunity values. In our view, decisions to protect or destroy ought to be based on an understanding of a full range of social and environmental benefits. This means taking into account global protection values (not accruing directly to the Amazon countries’ populations) in choices of whether and where to preserve the forest – though *how* such values are to be taken into account is a difficult political economy question in its own right, beyond the scope of this paper.

Global values are of two main categories. One is related to the carbon preserved when not deforesting (the values to the global society from reduced carbon emissions or of carbon stocks protected as a sink for greenhouse gas absorption). The other, less tangible, values are those

represented by preferences for saving Amazon rainforests in other countries (mainly, “passive use” values in higher-income countries manifested in willingness to pay for Amazon forest protection programs in such countries). Since these values always would come in addition to local protection values, they could (often) “tip the balance” in favor of protection, when forest conversion otherwise appears to be economically efficient. This raises new sets of problems and issues. One problem is that there are few means by which such global values, expressed by overseas beneficiaries, can readily transmit incentives to land users to forgo from deforestation in favor of conservation. While it is important to have a global perspective in mind when attempting to apply our results to practical policy, it will not play much of a role for our discussion in this paper, which is relatively technical and confined to the local and regional aspects.

Conversion of the rainforest to other uses in the Amazon (since 1970, approximately 18% of the total initial Brazilian rainforest area has been lost) has resulted from a complex of often interrelated causes, involving local, national and international drivers (Soares-Filho et al. 2008, Nepstad et al. 2006), with most of deforestation resulting in conversion into pastureland. The conflict between protection values and opportunity values in the form of agricultural production is thus essential. However, in the following discussion, one must be aware that finding “true” protection values of particular parts of the forest can be complicated and controversial (even when not taking any of the global values into consideration). This is in part due to strong interaction effects: the removal of rainforest from one area can have synergistic effects on other areas; or (when the removed forest area is large) even induce “tipping points” whereby catastrophic changes may be triggered.

On the other hand, one needs to recognize the great uncertainties involved in such exercises. Even when strong evidence is found that conversion of Amazon rainforest to other uses could be economically optimal, the process of forest conversion is essentially irreversible which places additional burdens of proof on such decisions. Important valuation elements may have been left out; and preferences (such as those for choosing the preservation alternative as such) may shift in favor of conservation. Thus, when we say that conversion of forest can be “optimal”; this is a conclusion that must be subjected to great scrutiny.

We will in this paper attempt to sum up and document some of the efforts that have been made, to put economic values on the service flows from the Amazon rainforest and how such values will be affected by forest losses. Most of what we document arises from studies at a local, project level, rather than to assess values at a regional scale. The objective is to serve as a starting point for a more systematic effort of this sort, where we intend to develop relevant and useful tools for policy makers.

Previous, but incomplete, attempts to value Amazon rainforests and their changes are found in Fearnside (1997), Andersen (1997), Torras (2000), Andersen et al (2002), and Margulis (2003). More recently, several more substantial efforts and projects have been undertaken, some of

which are currently ongoing, and some whose purpose is at least partly overlapping with ours. One is the TEEB program, sponsored by the European Commission, which involves a broad-based attempt to put economic values to biodiversity and other ecological losses when forest is lost within which work has also been done related to the Amazon (see Killeen and Portela, 2011). A Brazilian TEEB effort at a regional and local level is currently underway with sponsorship of the German technical cooperation program, while a national “EEB” effort is in the scoping phase.

Among other initiatives we mention WAVES, an international effort to embed ecosystem services and natural resources, including those represented by forests, in extended national accounts, led by the World Bank but in close coordination with the UN Statistical Office Environmental Accounts program. WAVES works on a national basis: a program for Colombia is already ongoing (see World Bank, 2013). UNEP has also sponsored a national-scale effort in Brazil to value attributes that add local and regional value to public protected areas in Brazil, where Amazon forest preserves are central due to the considerable area under such conservation units in the region (Medeiros, et al, 2011). Yet another similar project, by WWF-Netherlands, cites certain values of the standing forest from the literature as a basis for decision-making (WWF, 2009).²

1.2 Background to Recent Forest Developments in the Amazon

The South American region, most importantly Brazil, has long been considered as a potential “breadbasket of the world”, due to the size and potential productivity of its currently undeveloped land, and its already proven agribusiness prowess. But the complex and fragile ecosystems of the Amazon do not respond well to enhanced nutrient inputs. This contrasts to the much better-structured soils of the Brazilian Cerrado region. Also, few parts of the Amazon are well suited for mechanized agriculture due to terrain constraints (Nepstad et al., 2007; Nepstad et al., 2009; Soares-Filho et al., 2010). Researchers have thus repeatedly described the region as more valuable as a provider of regional and global forest services than as a new frontier for food or fiber commodity production (Chomitz & Kumari, 1995; Fearnside, 1997).

Nevertheless, the “laterite” hypothesis, that would transform Amazon soils to a “red desert” on clearing, has not always been proven true in practice (Goodland & Irwin, 1975). The Amazon is

² Another potentially applicable tool is InVEST, a means for geographic mapping of biologically-based values, developed by researchers at the Universities of Minnesota and Stanford, and operated with leading environmental NGOs (TNC and WWF); see e.g. Tallis and Polasky (2009), Kareiva et al (2011). Spatial valuation models have been developed under the auspices of the Amazon Scenarios Program (a policy-oriented initiative led by UFMG, WHRC and IPAM), and applied in several studies described below. Information about protection or production values from these programs will, clearly, support and add to the information documented here.

a mosaic of different landforms and soil types, within which there is space for more intensive crop and animal production in areas of *cerrado* and *terra roxa*. Although the area dedicated to these activities is proportionally limited at the biome scale, it is quite substantial, including agroforestry and silvipastoral systems operated by smallholders, managed timber and non-timber forest product extraction, and some reforestation, and perennial crop plantations whether with native or exotic species.

The Brazilian Amazon contains approximately 5.2 million km² of land, within which several forest and non-forest ecosystems exist. In the Amazon forest biome, if managed within the constraints of the 1965 Forest Code as initially revised by provisional measure in 1997, 80% of private lands were maintained in legal reserves under original or restored forest. It is conceivable even with such restrictions that the potential for harmonious regional development of these resources could be ensured. Such land use restrictions were relaxed to 50% of forest cover with the adoption of statewide Ecological-Economic Zoning (ZEE), a constitutional requirement in Brazil. The more recent (2012) revision in the Forest Code maintained the same requirements for conserving both legal reserves and riparian forests, although relaxed the obligation of forest restoration for past deforesters. According to estimates by Soares-Filho (2013), the revision of the Forest Code reduced by 59% the obligation to restore forests on private lands in the Brazilian Amazon.

Although ZEE has been conducted in several Amazon states, its promise to serve as a guideline to direct land use toward areas with greater productive and infrastructure potential has often been gerrymandered to allow incompatible activities. Successive frontier expansions have breached the restrictions posed by the literature and ZEE. The Belém-Brasília highway along the eastern rim of the basin, and BR-364 cutting across Mato Grosso, Rondônia and Acre to the southwest, served as principal corridors for migration and logistics. Econometric analyses have shown the close relationship between road expansion and deforestation in Amazon municipalities (Soares-Filho et al., 2006; 2010). Extensive cattle ranching is also closely related to deforestation, benefiting from the initial nutrient pulse provided by biomass burning, followed by low stocking rates associated with unimproved pastures, and in some cases abandonment. According to EMBRAPA & INPE (2008), 15 million hectares (21% of converted areas) are in abandoned secondary regrowth, mostly following pasture degradation.

Studies of the process of frontier expansion have noted its cyclical character (Schmink & Wood, 1992). Initial occupation occurred in many areas due to the biological provenance of rubber and related extractive species. With the decline in the natural rubber trade as plantations were established outside the Amazon, former tappers continued this activity at a lower level for modest trade, while engaging in small scale subsistence agriculture. Timber extraction began with these populations along riverbanks, but with increasing demand for wood as the Atlantic Forest was rapidly exhausted in the 1960s and 1970s, timber operations turned to the Amazon as a ready source of timber with lightly enforced restrictions to occupation and conversion.

At the same time, the military regime offered up the region as “land without people for people without land”, a region that should be “integrated rather than handed over” to foreign interests (*integrar para não entregar*). The geopolitical imperative was at the bottom of policies for infrastructure development – the Belém-Brasília and later BR-364 through Cuiabá to Rio Branco would serve as principal corridors for attracting settlers and enterprises to subdue and appropriate wealth from the region. This infrastructure was later integrated with that in the Peruvian, Bolivian and Venezuelan Amazon, finding new corridors to Europe, North America and the Pacific Rim. Recently, the infrastructure integration plan was revived by IIRSA (Initiative for the Integration of the Regional Infrastructure of South America) that aims to invest in large energy, transport and communications infrastructure projects across South America (Kileen et al., 2007).

Later cycles attracted more sophisticated extraction activity. As Bunker (1985) describes, the underlying logic of the rubber boom never really subsided, with discovery of seemingly inexhaustible mineral wealth (gold, diamonds, cassiterite, bauxite, magnesium, iron and gas) and hydroelectric potential, timber wealth and pasture, the sack continues.

In sum, forest conversion to pasture and cropland in the Amazon has been driven by diverse causes, including government programs that provided perverse incentives for frontier expansion, infrastructure improvement and expansion, land speculation in the absence of land titling and governance, logging, and cropland and cattle herd expansion in response to the growing demand for biofuels and for food. Internal migration associated with spontaneous settlement and smallholder settlement programs on forested land also contributed to these pressures. Hence the causes of deforestation are complex and frequently interrelated (Soares-Filho et al., 2008).

1.3 Applying Forest Valuation: Payments for Environmental Services

The underlying interest in valuation is in large measure driven by policy concerns – when is it optimal and efficient to preserve the forest – and if forest preservation is found to be the preferable policy, how is forest protection implemented? Both issues are closely related to rainforest valuation.

A fundamental issue in this regard, often neglected by conservationists, is that protecting the forest is not necessarily or always optimal. Opportunity values (in the form of potential agricultural outputs on currently forested lands; or conversion to other commercial uses) are often high and cannot be neglected. There are also often real and obvious conflicts over whether to preserve or develop a given forest area, between preservationists, authorities, and developers. In most cases, an optimal choice is not evident; only an appropriate valuation procedure (considering all aspects of the issue) can then reveal the case for alternative uses.

Secondly, should forest preservation be found economically optimal, the issue of implementation remains. Since parties that stand to gain from forest conversion to other uses, often do not bear all or most of the related costs, leaving the decision to save or convert the forest up to market forces alone is generally inefficient and sometimes catastrophic. Payments for ecosystem services (PES), to parties with power to convert forest land to other uses, may then be necessary. A further key issue is that, as long as the forested land is not acquired by government or conservation organizations, payments to those who face opportunity costs may be required on a continuous basis. Since long-term continuous payments of this type are unrealistic, a PES scheme for forest protection should usually be viewed as temporary; and as shown by Harstad (2013), to provide no guarantee against the loss of forest in the future.

In cases where payments are needed for forest preservation, for example, to recognize traditional peoples' historical roles in protecting such forests, obtaining precise values of net forest protection is in many cases extremely helpful, not least to prevent PES implementation payments from being either excessive or insufficient to bring about protection. The main point is that more precise valuation can provide clues to minimum payments that are necessary for the forest to be preserved; alternatively, the maximum payments that are efficient to make to save it.

There are many project-level initiatives underway that attempt to implement PES schemes or in other ways to harness market mechanisms (Wunder et al, 2008), so as to place appropriate economic values on standing forests. Perhaps the most important initiative being considered is Reducing Emissions from Deforestation and Forest Degradation (REDD) in developing countries (hereunder in the Amazon region), and the role of conservation, sustainable management of forests, and enhancement of forest carbon stocks (REDD+). In effect, REDD+ goes beyond PES projects as it focuses on creating both an institutional framework and international financial mechanisms for developing countries to reduce CO₂ emissions from deforestation and forest degradation.

The PES literature (Engel et al, 2008; Wunder et al, 2008) and forest carbon policy studies (*e.g.* Grieg-Gran, 2008) often treat opportunity cost arising from agricultural production parameters as the minimum of what society must be willing to pay for rainforest conservation, so it would serve as a floor cost rather than as a proxy for services precluded. Important issues have here been raised by several authors about the failure to account for other joint benefits of forest conservation that go above and beyond maintaining carbon stocks in forests (Stickler et al., 2009). The danger is then that PES payments will in such cases not be sufficient to prevent forest losses in particular in cases where opportunity costs are “close to” local and regional protection values; payment for the other services provided by the forest may then also be necessary. Borner et al. (2010) estimated a range of opportunity costs throughout the Brazilian Amazon, based on production statistics and knowledge of typical land use cycles. On the other hand, Muradian et al. (2011) suggested that forest occupants at the frontier may be willing at fairly low cost to bring themselves into conformity with a more firmly enforced land-use code, so as to avail themselves of access to institutional resources and avoid penalties. Pereira (2010) discusses two specific PES

schemes within the Amazon (in Bolivia and Brazil) and concludes that fully effective conservation programs must consider the full impacts on forest dwellers which may be beyond most standard PES arrangements. Additional costs of REDD+ implementation, including administrative and transactions costs, could in some cases be substantial (Nepstad et al., 2009), resulting from the need to develop local capacity to monitor and enforce land-use and zoning codes, and to support landowners' efforts to improve forest management and protection.

An important emerging literature deals with optimal implementation of REDD contracts in a number of situations: with lack of full contract enforceability, lack of full additionality, and under asymmetric information (where, typically, the land holder has private information about the opportunity value of deforested land). Much of this literature deals with the analytical and practical issues in providing effective preservation incentives to parties who effectively controls the land, without "overcompensating" landholders, given that buyers have limited financial resources (or limited willingness to pay) for such purposes. Salas (2013) shows that an optimal REDD enforcement contract implies no up-front payments with all payments contingent on ex post delivered protection values (such as carbon values), but that such contracts are more difficult to achieve and enforce when the opportunity costs are higher. Salas, Roe and Sohngen (2013) consider a similar problem under incomplete contract enforcement, and when opportunity values of protection are private. They show that a first best can still often be implemented given that forest conservation is sufficiently advantageous, but not when this value is smaller. Mason and Plantinga (2013) show that optimality can be enforced under wider circumstances, given a more sophisticated contract where the seller makes an up-front payment that is finally recouped on the average. Van Benthem and Kerr (2013), in studying a similar model, assume that such sophisticated contracts under asymmetric information are infeasible. Then some informational rent will flow to privately informed sellers, and some inefficiency will inevitably result when the buyer (reasonably) faces financial constraints. This is currently a very active research area which is quickly yielding new and interesting results; a full survey will not be attempted here. It is, in any case, useful to have the lessons from Harstad (2013) in mind: As long as forest land, not yet deforested, is not bought but instead only rented by the party that wants it preserved, protection against deforestation can itself only be rented and not bought; and there can be no guarantees against deforestation in the long run. Also, preservation typically requires that payments be made not just once, but continually flowing.

An issue for this literature, of particular relevance for our topic here, is the implications of pure uncertainty about actual protection values for the "buyer" or principal, concerned with implementing protection. This is likely to be a particularly serious problem under asymmetric information about opportunity values, and imperfect contract enforcement; as errors (in terms of not protecting valuable forest; protecting the wrong forest; and over-compensation of sellers) are then likely to be large. A better mapping of protection values could then substantially improve effective enforcement, and reduce enforcement costs. This would provide a tangible value of a more precise protection value mapping. So far little literature however exists on this topic.

A separate literature deals with the potential for sustainable agriculture or agroforestry to coexist with rainforest conservation (e.g. in buffer zones and/or as a means to protect agrobiodiversity). Such agricultures often rely on maintenance of the forest for pollination and pest control services, as well as to supply timber and non-timber products complementary to agricultural revenues. Pollination benefits have largely been studied in relation to specific perennial and some annual crops, including coffee (Ricketts et al. 2004; Olschewski et al., 2006; Priess et al., 2007), cashew, Brazil nuts, soybeans and others. The losses due to deforestation associated with pollination services can be substantial. Analysis of the pollination deficit in major crops suggests that productivity may be as much as 50% reliant on presence of animal pollinators (bees, bats, hummingbirds, etc.).

2. Elements in Local and Regional Valuation of Amazon Rainforest Preservation

2.1 Climate Regulation Services and Forest Carbon Emissions

The Amazon forests are home to about 20 percent of known terrestrial species (Raven, 1988), contain one-tenth of the global carbon stored in land ecosystems (86 ± 17 Pg of carbon, Saatchi et al. 2007), and account for one-tenth of global net primary production, sequestering 0.49 ± 0.18 Pg of carbon in an average climatic year (Phillips et al., 2008). As the world's largest tropical forest (5.4 million km^2), the Amazon also plays a central role in maintaining the global carbon balance and regional climate regimes; its forests cool the air by pumping about 7 trillion tons of water per year into the atmosphere via evapotranspiration (Moutinho & Schwartzman, 2005). In essence, the forest functions as a giant air conditioner that keeps the regional climate humid and rainy by cycling atmospheric water in the form of aerial rivers to the southeast and center of the South American continent (Fearnside, 2003).

Despite these important current roles of the Amazon rainforest, tropical deforestation has released significant net amounts of carbon to the atmosphere. During the 1990s, between 0.8 and 2.2 Pg of carbon were emitted per year from tropical deforestation, representing 10–35 percent of global GHGs (Houghton, 2005). Today this share has decreased to 12 percent, due to both increased fossil fuel emissions, and due to declining overall global deforestation rates (Van der Werf et al., 2009). The Amazon carbon stocks are also sensitive to shifts in climate. In the severe drought of 2005, the Amazon forest lost about 1.2–1.6 Pg of carbon biomass (Phillips et al. 2009), and in 2010 a more extensive drought resulted in a loss of about 2.2 Pg of carbon (Lewis et al., 2011).

In addition to deforestation, forest fires also influence global warming. Alencar et al. (2006) estimated that annual committed carbon emissions from fires in the Brazilian Amazon may amount to 0.094 ± 0.070 Pg in El Niño–Southern Oscillation (ENSO) years. However, this figure can be far greater in extreme ENSO years, such as the event of 1997–1998, when emissions from

forest fires in Mexico, the Amazon, and Indonesia totaled 1.6 Pg of carbon (Houghton, 2005)—the equivalent of 18 percent of current fossil fuel emissions worldwide (JRC, 2009). Not only do forest fires alter atmospheric composition, they also interrupt rain cloud formation (Ackerman et al., 2000), thereby reducing rainfall (Andreae et al., 2004) and increasing the average residence time of aerosols in the atmosphere (Ramanathan et al., 2001). These effects also have a significant negative impact on human health (Mendonça et al., 2004). For example, during the extreme drought in the southwestern Amazon in 2005—probably associated with the abnormal warming of the tropical North Atlantic—more than 40,000 people in the State of Acre sought medical care due to a persistent smoke plume, which stemmed from multiple fires that burned 300,000 hectares of forest in the region (Brown et al. 2006; Aragão et al. 2007). Moreover, direct economic losses from widespread fires in 2005 amounted to US\$50 million (Brown et al. 2006).

In sum, fires in tropical forests release globally significant amounts of carbon to the atmosphere and may increase in importance because of climate change. Among the Intergovernmental Panel on Climate Change (IPCC) scenarios that do not consider mitigation efforts, the A2 scenario is currently considered very plausible, given the steady increase in anthropogenic carbon emissions (Van der Werf et al. 2009). Under this scenario, temperatures are expected to rise between 2° and 5.4°C during this century. As a result, climate models predict a replacement of a large portion of the Amazon forest with savanna-like ecosystems by the end of the twenty-first century, turning the standing Amazon forests from a net sink to a source of atmospheric carbon (Cox et al. 2000; Collins et al., 2005; Salazar et al., 2007; Nobre et al., 2010). In addition, extensive deforestation may reduce rainfall over the Amazon (Salati et al., 1991; Sampaio et al., 2007; Da Silva et al., 2008; Coe et al. 2009; Nobre et al. 2009) and increase the length of the dry season (Costa & Pires, 2010), augmenting the risk of loss of a large portion of the forest to climate change–induced fires by as soon as 2020 (Golding & Betts 2008; Silvestrini et al., 2011). The synergy between drought, tree mortality, fire, deforestation and predatory logging practices may deeply impoverish the remaining Amazon forest, impairing essential ecosystem goods and services (Foley et al., 2007), with large socioeconomic consequences.

The synergy between deforestation, exacerbated by growing global demands for agricultural products and biofuels (Nepstad et al., 2006), and infrastructure investments in the Amazon (Soares-Filho et al., 2006), forest degradation by logging and fire, and an increasingly drier climate may boost fire activity in the Amazon. This vicious cycle could lead the remaining forests toward an impoverished state (Nepstad et al., 2001)—a tipping point that might be reached as soon as within the next two decades (Nepstad et al. 2008; Golding & Betts 2008). The degree to which this process will affect the Amazon forest is still uncertain and depends on other joint effects of climate change, such as the potential fertilization of vegetation from higher atmospheric CO₂ concentrations (Rammig et al., 2010) and the level of resilience of remaining forests (Soares-Filho et al., 2012).

2.2 Sustainable Timber Supply

Much of the literature on direct-use benefits of the Amazon forest is associated with extraction of forest products. Timber represents the largest value component of these (IBGE, annual; Wunder et al., 2008). It is recognized however that most timber extraction in the Amazon is accomplished using rotations insufficient to restore original productivity. In Brazil, IBAMA approved forest management plans require only 25-year rotations. According to research largely undertaken in the Eastern Amazon (Uhl, et al., 1991; Almeida et al., 1995; Barreto et al., 1998; Merry, et al., 2009), a rotation of at least 30 years, and the employment of reduced impact logging (RIL) techniques (Bacha and Rodrigues, 2006; Barreto et al., 1998; Killeen and Portela, 2011) are desirable to ensure both profitability and species conservation in the long run. Data for variation in biomass volumes across much of the Amazon (Saatchi et al., 2006) have made it possible to establish regional values for potential timber harvest (Merry et al., 2009).

For the Madre de Dios región in Peru, Giudice et al. (2012) developed a rent model that estimates optimal harvest fees, compares scenarios of timber harvest for the next 20 years, and calculates potential revenues to the State. They concluded that annual forest revenues to the State could increase from the current US\$ 1.0 million to an average of US\$ 23.4±1.4 million per year if the fee structure suggested by the study were adapted. The study also concluded that timber harvesting in Madre de Dios could increase by up to 200% over the next 20 years, without jeopardizing the integrity of conservation areas.

An attempt at an overall valuation of sustainable timber production potential in national and state forests in the Brazilian Amazon, by Medeiros et al (2011), concludes that this could generate net revenue on the order \$1.2-2.2 billion at current timber prices.

IMAZON has also conducted detailed studies of the timber frontier, viability of sustainable harvesting, transport costs, etc. It will be difficult, however, to predict the future value of sustainable timber supplies. It may be assumed that scarcity coupled with more effective regulatory control to keep loggers out of protected areas, will increase this value, but substitution by plantation stocks may make natural timber management less attractive.

IMAZON has surveyed all functioning enterprises in timber extraction poles in the Brazilian Amazon every five years since 1998 (including 2004 and 2009), reporting that although some consolidation has occurred, there were still about 2,225 timber operations throughout the region at the end of the last decade, mostly small in scale averaging 3,400 m³/yr processed volume (Barreto et al., 2005). Although there has been a general production decline and reduction in the number of logging enterprises, the average value of gross timber revenues is higher per m³. Volume declining per firm suggests depletion of accessible timber stock, which may be related to the decline in deforestation activity over the past 7 years.

With decline in higher valued timbers, overexploited in managed areas, lesser species begin to be utilized, due to recognition that many species have desirable mechanical properties. This has broadened the value base of the timber resource and reduced the area necessary for management for a given revenue target. Spatial analysis (Merry et al., 2009; Guidece et al., 2012) has revealed the potential for substantially superior timber revenues to be generated from managed areas in the Brazilian and Peruvian Amazon.

2.3 Non-Timber Forest Products, Game, and Fish

Considerable research and investment has been dedicated to the potential and limitations to development of value chains based on non-timber forest products (NTFP) (Peres, 2000; Anderson, 2002; Clay & Clement, 1998; Arnold & Ruiz-Perez, 2001; Belcher et al., 2005; Belcher & Ruiz-Perez, 2007, Muñiz-Miret et al., 1996; Nunes et al., 2012).

The panacea of NTFP as a “savior” of the Amazon forest, widely disseminated in the 1980s (e.g. Peters et al., 1989), has been widely called into question, due principally to the localized concentrations of such products, and their correlation with poverty and forest dependence (Southgate, 1998). Wunder (1990) carried out the most complete analytical review of NTFP production concentration in the Brazilian Amazon, based on national vegetal extractive statistics (IBGE, various years). A survey by May (2002), undertaken at the request of the FAO, included a review of information available on NTFP production, sustainability and profitability, in the Amazon, Cerrado and Atlantic Forest biomes of Brazil.

Recent work by scholars associated with Britaldo Soares-Filho in the Brazilian and Peruvian Amazon have identified areas in which the spatial concentration of Brazil nut resources could serve as a basis for local development, generating state and private revenues from value-added processing (Nunes et al., 2012). This study developed and used a spatially-explicit rent model of Brazil nut production to assess yields and potential rents from the Brazil nut concessions in Madre de Dios, Peru, under three scenarios of processing and management (unshelled, shelled, shelled-certified nuts). The authors estimated potential annual production in the region to be 14.1 ± 2.4 thousand tonnes of unshelled nuts, which at 2008 regional sale prices corresponds to rents of between US\$ $3.1 \pm 0.5 \text{ ha}^{-1} \text{ year}^{-1}$ for unshelled nuts to US\$ $8.4 \pm 1.4 \text{ ha}^{-1} \text{ year}^{-1}$ for shelled-certified nuts. Investments needed to scale-up certified production in all Madre de Dios concessions and approximately triple rents are in the order of US\$ $14\text{--}17 \text{ ha}^{-1}$.

The study also concluded that the Brazil nut industry in Madre de Dios presents a unique opportunity to harness conservation with sustainable development. If upgraded to certified-shelled production, the rents associated with Brazil nut collection combined with ecotourism and sustainable logging will compete with or trump rents of traditional agriculture in Madre de Dios (Table 1). Even so, the success of this industry depends on investments needed to expand processing, lower production costs, stabilize a market floor price, and command a price premium

for the environmental co-benefits from the sustainable management of the Brazil nut concessions, whose values are currently not attractive.

A similar study was carried out for the state of Acre, Brazil (Nunes et al., 2011). In this study, the authors used a spatially-explicit rent model of Brazil nut production to assess yields and potential rents from the major regions of Brazil nut collection in Acre, Brasil, under three scenarios of processing and management. They estimated potential annual production in this region to be 29,27±8,42 thousand tonnes of nuts with shell, which at 2008 regional sale prices corresponds to rents of between R\$ 0,28±0,08 ha⁻¹.year⁻¹ for nuts with shell to R\$ 7,8±2,39 ha⁻¹.year⁻¹ for shelled-certified nuts. Investments needed to scale-up certified production in the major collection regions, and boost the rents by more than 20 times, are in the order of approximately R\$ 22 ha⁻¹.

Table 1. Rents of Rural Activities in Madre de Dios, Peru

	NPV (US\$/ha)	Reference year	Source
Ecotourism	440	2005	Kirkby <i>et al.</i> 2010
Small landholder agriculture	294	2006	Kirkby <i>et al.</i> 2010
Cattle ranching	395	2006	Kirkby <i>et al.</i> 2010
Sustainable timber	544	2008-2009	Nunes et al. 2012
Improved, shelled-certified Brazil nut	320	2005	Nunes et al. 2012

NPV: Net Present Value over 25 years; discount rate of 7.35%. (Nunes et al., 2012)

For açaí, there is a potential for such a product to capture enthusiasm, given the expressive growth in niche demand, whose production chain is estimated to generate over US\$ 1 billion in net revenues annually (Brondízio, 2008). Nevertheless, Homma (2010) predicts the demise of NTFPs and their substitution with cultivated agroforestry products based on the same genetic material. This is the case of rubber tapping in Acre state, where the activity only survives due to economic subsidies from the government (Jaramillo et al., in preparation).

Thus, deforestation could inhibit sustainable harvesting but not necessarily growth in volume or value of such products, as they become fashionable in global markets. Agroforestry potential in forest buffer areas is associated with the use of NTFP as part of a diversified strategy for forest use.

Extractive activities have been progressively replaced by others more profitable per unit area, market intelligence and wider commercial networks. Their persistence is due to structural and situational factors: 1) continued existence of pockets of absolute poverty in rural areas; 2)

complementarity between family production and plant extractive activities in places where there is an abundance of native species with functioning markets in place; and 3) attempts to add value to niche and production chains associated with conservation and sustainable attributes. The major motivation is undoubtedly the persistence of poverty in tandem with plant genetic resource occurrence, since the number of undertakings in “sustainable” business based on NTFP, and their relatively limited success means that the audience reached is insignificant in relation to officially recorded production. The main NTFPs of regional importance in Brazil, in decreasing order of average annual value in current US\$, are described in Table 2, below.

Table 2. Value of Principal non-Timber Forest Products from the Brazilian Amazon Rainforest, 1985-2008. Current US\$ (IBGE, various years).

Years	1985	1990	1995	2004-08 (average)
Erva Mate	24,918,384	92,110,484	34,875,137	39,328,250
Açaí fruit	28,554,855	45,831,745	34,815,441	37,824,060
Babaçu (oil kernel)	40,563,277	30,111,716	38,372,301	47,989,670
Coagulated natural rubber	78,087,627	19,986,261	6,909,760	3,520,907
Piaçava fiber	20,362,390	69,270,337	13,660,332	37,994,790
Heart of palm - açaí (1)	5,406,838	16,327,742	13,136,006	3,792,167
Brazil nuts	19,378,986	7,224,062	5,688,986	19,436,001
Carnauba wax	6,273,611	12,299,941	2,648,493	7,000,228
Total (8 products)	223,545,968	293,162,288	150,106,456	196,886,073

(1) This amount refers only to palm heart coming from the Amazon, derived from *Euterpe oleracea* (açaí).

Territorial concentration is the norm among this smaller group of products with significant value. They are primarily derived from “oligarchical” species (Peters et al., 1989), meaning that their occurrence is geographically concentrated in certain regions or ecosystems where such species dominate (e.g. babaçu forests in the Vale do Mearim in Maranhão, açaí forests in the Amazon estuary near Belém, piaçava areas in the Northern várzea and in coastal zones in the Northeast, etc. The dependence on a given region for a certain vegetal raw material tends to be very strong. More than 80% of the value for NTFPs in each state in the Amazon, for example, is concentrated in only one product (Wunder, 1999). Additionally, the principal products of vegetal extractivism (historically rubber and Brazil nut, and more recently açaí, babaçu and piaçava) concentrate a significant part of the total market value for the products recorded. Furthermore, roundwood or

firewood use greatly exceeds the value of NTFPs in nearly all areas of their occurrence, which partially explains their replacement by other land uses. Since these uses generally result in the permanent removal of the forest, the potential for NTFP to revert deforestation patterns is restricted. Tallis and Polasky (2009) report on the application of their integrated valuation tool, InVEST, tool to assess damages due to Amazon forest losses, and find substantial losses in the values of local forest products under a “business-as-usual” alternative up to 2020, most along current and proposed roadways and population centers.

According to Peres (2000) as many as 25 million game vertebrates, constituting 89,000 tons of biomass, are consumed annually by local Amazon populations in the form of bush meat, valued at replacement cost at \$197 million. Fish are a basic staple to Amazon residents, so it is no surprise that their production is also considerable in the region. Studies showed that commercial fisheries on the main tributaries of the Amazon river attained an annual catch valued at \$ 389 million/yr. (Almeida et al., 2003).

2.4 Eco-Tourism

According to Kirkby et al. (2011) eco-tourism should be defined as "travel to natural areas to admire, study, or enjoy natural landscapes and wildlife in a way that contributes to conservation and the welfare of local peoples". The same source puts global net rents from eco-tourism in the range of US\$ 28.8 billion yr⁻¹ (a bit over 4% of the global tourism market of nearly \$1 trillion). A key question here is how much of this currently consists of Amazon ecotourism, and how significant such activity could become.

A range of estimates for the value of eco-tourism as a means to justify standing forests have been generated. These include Andersen (1995), whose evaluation implied that the number of tourists visiting the Amazon region could be easily multiplied several fold, with expenditures on the order of \$1,699 per visit, of which 50% could be captured for local benefit. Studies have looked at the potential eco-tourism revenues in eastern Ecuador (Drumm, 1991), with results similar to those obtained by Andersen on a per-hectare basis. Of particular importance, the Madre de Dios region of the Peruvian Amazon is one of few currently extremely attractive regions in the Amazon for ecotourism, given in particular its close proximity to Cuzco (Kirkby et al., 2010). In other words, ecotourism is (at least, currently) economically important at a regional level only where integrated to a wider network of tourist attractions, such as the ones of the Cuzco region, which attracts tourists from all over the world.

Therefore, it is clearly not meaningful to provide average figures on this component for the entire Amazon biome, as tourism destinations are highly dependent on the presence of distinct landscape features or species, organized entrepreneurs and – most of all – ease of access, comfort and security. There is clearly a growing demand for such activity, but there are important caveats to the valuation: 1) “eco” tourism may not overall be very environmentally friendly: the more

popular it becomes, the less likely it is to protect the resources that are the object of visitation (exception made for Madre de Dios, where tourist concessions run by eco-lodges must and are conserving the forest (Kirkby et al., 2010), and 2) the more exclusive forms of such tourism tend to be more highly valued than those directed toward the general public, however their elasticity is also high.

The discussion in some of the literature (e.g., Andersen, 1995) has sought to consider total potential ecotourism values, for the entire Amazon biome. A more appropriate consideration is the value of ecotourism that could be lost when only smaller parts of the forest (and not all of it) are lost. It seems obvious that not all the rainforest will be used for ecotourism; which raises the issue of substitute sites when particular sites that currently are prized as destinations are lost or degraded. The implication would be to reduce the loss, relative to that calculated on basis of the average return to ecotourism for the entire biome. Empirical studies to illustrate this issue are currently lacking.

2.5 Pharmaceutical Values and “Bio-Prospecting”

Option values for forest protection based on pharmaceutical use potential remain controversial in the valuation literature. Although the probability is fairly high that active compounds with pharmaceutical value may be found, it is spurious to place a value on an entire biome or even representative ecosystems based on potential bio-prospecting “hits”. In most studies in the literature, a probability factor is included to reflect the number of species identified or estimated in a given biome, and the historical rate of activity found in random collection. Although it was often argued that probability could be dramatically increased by the use of indigenous knowledge, it turns out that other methods are available for screening in a collection protocol that are as or more effective than native intelligence. The current use for such purposes is poorly recorded in statistics, and there is widespread subsistence use that is difficult to account for (few valuation studies with traditional communities include attempts to value the benefits that accrue through use of medicinal plants).

Although the potential benefits may be quite high, the overall option value of the Amazon region for such use has been disputed in the literature, ranging from as low of \$21 ha⁻¹ (Simpson et al., 1996) to nearly \$10,000 ha⁻¹ (Rausser & Small, 2000) analyzing data from the same forest in eastern Ecuador. The former amount is similar to what Andersen (1995) reports for the Brazilian Amazon (\$30ha⁻¹ yr⁻¹). The value range arises from distinct assumptions about the number of endemic species that may be found in a given patch of forest, and the strategies used for collection and screening (Costello et al., 2006). Certainly, with deforestation and extinction, you could posit that lower probabilities of such finds would prevail. With a diminishing stock of endemic species, there would be lower chance of making a “hit”. The question is whether such valuation conveys the actual utility of forest conservation, or is purely hypothetical.

Considering that Brazil is a megadiverse country, the level of research and development activity involving prospecting the potential of this immense gene pool is either insufficient, or continues to be carried out with impunity by biopirates. In contrast, the regulatory procedures established by law and the considerable delays and transactions costs involved in securing permits have impeded the generation of royalties and taxes from potential bioprospecting discoveries and associated technical innovation. Cases of accusations of biopiracy, including apprehensions of collections and preventive arrests over a long period effectively paralyzed scientific work focused on native biodiversity with traditional populations. Business projects based on biodiversity have been similarly affected (of a total of 86 permits granted by the genetic research and development board CGEN as of August, 2009 (Figure 1), only 17 requests, of which 6 were renewals, were presented by companies); and only three private companies – Natura, Extracta and Quest Pharmaceuticals – had secured approvals for access. Embrapa – the national agricultural research corporation – submitted a similar number of requests for permission. At the same time, over 150 requests for such approvals were still on the docket as of November 2011 (May & Vinha, 2013).

Fig. 1. Biodiversity use licenses authorized by CGEN by purpose: 2003–2009.
Source: May & Vinha, in press; www.mma.gov.br, CGEN, *processos autorizados*

All evidence indicates that the complex procedures and the delay in granting permits have discouraged innovation in the biotechnology sector based on natural assets, which encourages more R&D based on biochemical synthesis. Other companies in the cosmetics and essential oils sector have expressed a lack of interest in subjecting themselves to CGEN administrative procedures, since going through such procedures for obtaining an access permit publicizes

interest in a given natural resource, and can result in unfair competition while a permit is being officially reviewed (the process of granting a permit can take as long as 5 years). Nevertheless, an evaluation of the potential for investment in development of plant-based health and cosmetics products in Brazil concluded favorably, due to the anticipated low cost of traditional knowledge that could be brought to bear on new product development (Fonseca et al., 2009). Here again, however, the prospect that random collection may be similarly efficient in collecting biologically valuable compounds, combined with the restrictions against accessing traditional knowledge would militate against such a result.

2.6 Water Resource Values

2.6.1 Direct Values of Changed Water Flows and Precipitation

Among ecosystem services, the regulation of the water balance and river flow by forests (Guo et al., 2000; Costa et al., 2003) is of particular importance as they provide a series of economic values to agriculture, hydroelectric power generation, industry, urban dwelling (Guo et al., 2000), as well as many other use (*e.g.* fisheries) and non-use values (*e.g.* eco-tourism) (Fisher et al., 2009).

Previous simulations of interactions between biosphere and atmosphere (Shukla et al., 1990; Nobre et al., 1991; Hahmann & Dickinson, 1997; Sampaio et al. 2007; Ramos da Silva et al., 2008; Coe et al., 2009) indicated that high rates of evapotranspiration by the Amazon forest are partially responsible for the maintenance of local precipitation (Malhi et al., 2008). Thus, large scale deforestation over the basin may lead to reduced precipitation rates (Hahmann & Dickinson, 1997; Costa & Foley, 2000; Sampaio et al. 2007; Ramos da Silva et al., 2008; Coe et al., 2009). These impacts would be more intense during the dry season (Nobre et al., 1991; Sampaio et al., 2007) and could increase, as a result, the length of this season in some Amazon regions (Costa & Pires, 2010). In addition, local changes in the water balance due to deforestation could intensify droughts events during El-Niño years (Voldoire and Royer, 2004; Ramos da Silva et al., 2008), raising major concerns given that the frequency and extent of severe droughts may be increasing, especially across south and southwestern Amazon (Lewis et al. 2011).

Land-cover changes induce complex interactions between climate and land by altering precipitation, evapotranspiration and soil moisture. Consequently, it is hard to predict the outcomes of these interactions for river regimes (Costa & Foley, 2000). Deforestation at a local scale could cause a direct increase in surface runoff due to the reduction of evapotranspiration (Sahin & Hall, 1996) without any measurable effect on regional precipitation. However, at a large scale, climate feedbacks from land cover changes and atmospheric dynamics could also lead to a reduction in precipitation, which if large enough could result in a decrease in surface

runoff (Coe et al., 2009). Therefore, the net effect of land cover changes on river regimes depends on the heterogeneity, extent and spatial pattern of the deforested area (D’Almeida et al., 2007).

Various ecological impacts could result from either increasing or decreasing river discharge, depending on the scale of these changes. Considering the local effects of deforestation only on surface processes, the increased surface runoff together with soil exposure due to forest clearing would increase sediment loads and nutrient transported from terrestrial environment to river system (*e.g.* Walling & Fang, 2003; Latrubesse et al., 2009). Consequently, changes in water quality and riverbed morphology could alter aquatic ecosystem and fish community structures (Poff & Allan, 1995). Some studies have also shown that deforestation could cause increase in flood frequency and severity that may entail large economic losses (Bradshaw et al., 2007).

In turn, the complex feedbacks between deforestation, climate, and hydrology may also diminish annual and seasonal precipitation patterns. Climate data observations in Rondonia pointed out an 11-days delay on the onset of rainy season after a period of 30 years of intense deforestation (Butt et al., 2011). The increase in the length of the water deficit period could interfere on navigability of rivers, water supply and hydroelectric power generation (Brown, 2006; Marengo et al., 2008).

Even in the (fortunate) case where these local and regional hydrological impacts of forest cover changes are well understood, the issue of their economic value remains. In large measure this is related to agricultural and electrical power generation impacts, mostly in the vicinity of forests (lost, or remaining), but also in some cases far from the forest. Water flows are of course closely connected to precipitation patterns, which are bound to change with major changes in forest cover, with resulting impacts on agricultural outputs.

2.6.2 Impacts on Hydroelectric Potential

Regulation of river discharge is a fundamental ecosystem service that the Amazon forest provides, and therefore must economically be valued in order to draw attention to the need of reconciling conservation with sustainable development in the Amazon. Much of the valuation work associated with water flow benefits in the Amazon has focused on hydroelectric potential, since localized water availability has not historically been problematic (though the effect of the quality of such supplies on human health is an important issue, intertwined with child mortality).

Brazil and other South American countries facing future energy shortages are building hydroelectric dams in remote regions of the Amazon. However, hydropower production in the Amazon may depend upon its forests because of their role in maintaining the region’s rainfall. For example, Belo Monte complex, expected to be the world’s third largest hydroelectric complex, is a key component of Brazil’s energy strategy. In the next decade, energy demand in

Brazil is anticipated to increase by approximately 51%; the Belo Monte complex is expected to provide over 40% of this additional supply of electricity. Project engineers assume that rainfall patterns will be stable into the future. However, this might not be the case. Belo Monte's energy generation may be lower than expected if deforestation proceeds unabated (Stickler et al, 2013). Stickler and colleagues have attempted to model the impact of deforestation on the energy supply of Belo Monte; see Stickler et al (2013). According to these authors, if deforestation proceeds within both the Xingu and Amazon basins and climate feedbacks are taken into consideration, power generation potential would decline by almost 70% below maximum installed capacity and is likely to fall short of expected capacity by an average of 62% in all but 4 months. Similar analyses could be expanded to other river basins in the Amazon and the effect of deforestation on power generation of planned hydroelectric could be measured and monetized using the energy market prices (Mohd Shahwahid et al., 1997; Arias et al. 2011; Guo et al. 2000).

Nevertheless, one of the limiting factors of climate modeling using AGCMs is the coarse spatial resolution of the input data and of the model itself. Hence, changes in hydrology of small watersheds can be quite different from the overall result for the entire basin, but cannot be represented accurately by the model. Therefore, there is a need to refine the current hydrological models to incorporate both finer geomorphological features and regional rain patterns.

2.6.3 Watershed Protection

Irrigation is rare within the Amazon biome, but increasingly important on the forest fringe. Much of the intensive agriculture in the Amazon basin occurs in the floodplain (*varzea*), during the dry season. With climate change, such activity has become increasingly vulnerable to drought, flash flooding, and associated risks. The costs of e.g. crop and property insurance, measurable flooding or drought losses or changes in productivity might be ways of getting at these effects. Their magnitudes would depend on settlement patterns in the floodplain which are also changing with the realignment of transport corridors to the *terra firme*.

The Amazon forest keeps the regional climate humid and rainy by cycling atmospheric water in the form of aerial rivers to the southeast and center of the South American continent (Fearnside, 2003). Hence, the growing importance of irrigation in cerrado areas on the Amazon forest fringe would be affected by loss of localized water availability, or affected by the concentration of rainfall in a lesser part of the growing season. There is also a reasonable body of knowledge on the effects of erosion and sedimentation on economic infrastructure such as navigation, irrigation and hydroelectric generation. Yet most or all of these impacts are likely to be localized rather than generalizable to the region as a whole. There are more pervasive effects of climate change on drought-related losses in agricultural productivity that could be treated as avoided losses. The World Bank's Erick Fernandes (pers. comm.) reports significant declines in output due to the 2010 drought.

2.7 Evaluating Costs of Wildfires

Controlled fires in the Amazon have been used for hundreds of years, by indigenous peoples and then mostly in a sustainable manner (Pivello 2011). Today, forest fragmentation and the use of fire as a land management tool are the major drivers of fire in tropical forests. Disturbance from logging also contributes to forest fires, as it opens the canopy and increases light penetration that dries dead leaves on the floor, thereby decreasing understory humidity and increasing fuel loads and forest flammability (Cochrane, 2003). It is common to use fire to restore pasture productivity and to clear forested land for agriculture in the Amazon, and these activities are the major ignition sources for understory fires (Nepstad et al. 1999; Alencar et al. 2004). In these mixed agricultural landscapes, forest fragments become highly susceptible to fires that escape from nearby cleared areas, especially due to lower humidity and higher flammability at forest edges (Ray et al. 2005). Fire also begets fire; after an understory fire event, tree mortality produces a combination of increased dead organic matter on the forest floor and a more open canopy, thus increasing the chance of fire recurrence (Nepstad et al. 1999).

Another cost of wildfires in the Amazon is the impacts on agricultural lands on the forest fringe. Potentially, the damage related to such fires could be kept down via “Coasian bargains” between established agriculturalists and those responsible for rainforest wildfires. The evidence however strongly indicates that such impacts are not a concern of the latter; see Morello (2013). There is thus likely to be a substantive negative externality problem associated with such fires.

The potential consequences of forest fires have called into question our limited understanding of the science of fire in the tropics, and underscored the need to develop models of understory fire as tools to assess the impacts of forest fires in the face of a changing environment due to global warming and increasing anthropogenic forest disturbance. However, modeling fire in tropical forests is still at an early stage (Silvestrini et al., 2011). To date, the only process-based understory fire model developed for tropical forests is FISC (Fire Ignition, Spread and Carbon components) (Soares-Filho et al., 2012). This model was used to simulate current and future understory fire regimes and the associated carbon balance of forested landscapes of the Xingu Headwaters under a set of land-cover change, land-use management, and climate change scenarios. Of particular importance, simulations of fire regimes under modeled scenarios revealed that the major current and future driver of understory fires is forest fragmentation rather than climate change (considering that drought frequency remains constant, which tend not be the case). Fire intensity proved closely related to the landscape structure of the remaining forest. While climate change may increase the percentage of forest burned outside protected areas by 30% over the next four decades, deforestation alone may double it. Nevertheless, a scenario of

forest recovery and better land-use management would abate fire intensity by 18% even in the face of climate change.

FISC could be adapted and expanded to the entire Amazon basin. This effort will rely heavily on the prompt availability of basin-wide time-series maps of forest fire scars (e.g. Alencar et al., 2011). The economic impacts of future fire regimes in the Amazon will also rely on local estimates of the economic losses due to wildfires on crops, agro-forestry systems, and pastures, which might exist, although scattered in the literature. For example, Mendoza et al. (2004) estimated losses related to the release of carbon into the atmosphere, as well as losses due to respiratory ailments provoked by smoke from fires. Their study covered costs related to accidental pasture fires, fence losses, forest losses, carbon emissions, and impacts on human health. The authors, although using aggregated figures, estimated an average cost of around 0.2% of the region's GDP during the 1996–1999 period.

Fig. 2. The Amazon basin today and future fire risk. (left) Protected areas and major planned infrastructure. (right) The risk of fire by 2050 (Silvestrini et al., 2011) under business-as-usual deforestation (Soares-Filho et al., 2006) and climate change scenarios. (Davidson et al. 2012).

2.8 Effects of Climate Changes beyond the Region Including on Agriculture

To come to grips with such effects, one needs to sort them out from more general effects of climate change as such; this is admittedly difficult. Main potential such effects are likely to fall on agriculture. But one needs to be aware of the possibility of general equilibrium effects of various sorts, including via commodity markets. These could be of a global nature given that major commodity price changes are triggered by events and developments in the Amazon.

More recent concern is associated with the extra-regional climate effects of large-scale forest loss. A key issue here is that the Amazon forest influences the regional climate of areas such as the U.S. Midwest via climatic tele-connections (Avisar & Werth, 2003). There is an enormous accumulation of data from the LBA and associated NASA supported studies (Davidson et al. 2012), but little in the way of conclusive results to suggest the magnitude of rainfall availability in critical crop production regions that could be affected by a given proportionate loss in forest area. This is a line of inquiry that is attracting considerable attention from researchers connected with the UK funded ESPA program, as a basis for international payments for ecosystem services. How much more we can add to this kind of analysis is the question.

3. Opportunity Costs of Amazon Forest Protection

3.1 Agricultural Outputs on Cleared Land

Cattle herd expansion has been the single factor contributing most to Amazon rainforest destruction. The Brazilian cattle herd grew from 147 million head of cattle in 1990 to about 200 million by 2007. Eighty-three percent of this expansion occurred in the Amazon, and it is to be feared that this trend could continue as the industry bounces back from a recent agricultural downturn (Bowman et al., 2012). An estimated 18.5% of the Brazilian Amazon has been converted from forest to other uses, most during the past 30 years. Although 80% of this area is dedicated to beef cattle ranching on extensive pastures, its direct impact on deforestation is unknown.

Conversely, only 13–18 percent of deforestation is caused by conversion to soy crops (Morton et al., 2006; Lima et al., 2011), of which less than 6 percent can be attributed to biodiesel (Lima et al., 2011). However, there is a growing concern that soy expansion fuels deforestation elsewhere as it displaces cattle ranchers to inner Amazon frontiers, where land is still cheap (Nepstad et al., 2006; Arima et al., 2011). Soy crops are expanding specially on older frontiers; during the 2001–2004 period in Mato Grosso, more than one-third of the area where soybean plantings expanded was previously in pasture (Morton et al., 2006), and this increased demand for land plays an important role in driving the expansion of extensive ranching at the deforestation margin along with the more intensive cultivation of soy and other crops on older frontiers, both directly and indirectly (Bowman et al., 2012). Today, soy and sugarcane occupy large swaths of the southern and central-western Brazilian Cerrado and have expanded into the Amazon region despite the official ban on growing sugarcane there (Manzatto et al., 2009), raising concerns about environmental and social consequences.

This expansion increases land speculation and land concentration among larger-scale farmers with greater skills and capital, hence mobilizing deforesters who have capitalized on high rates of agricultural return at scale and spurring rural-urban migrations as land prices increase (Garcia et al., 2007). The magnitude of these indirect impacts of agricultural expansion on Amazon deforestation are yet unknown, in spite of attempts to model the phenomena (*e.g.* Lapola et al. 2010; 2011; Gouvello et al. 2010). However, it is clear that high rates of return to agriculture increase the opportunity costs of conservation (Grieg-Gran 2008; Soares-Filho et al., 2010), as well as the costs of enforcement (Nepstad et al., 2009), and put more pressure on the Brazilian government to soften environmental laws, such as the bill passed to reform the Brazilian Forest Code.

Recent studies for the Brazil Low Carbon Case Study sponsored by the World Bank (Gouvello et al., 2010) developed land use predictions for Brazil with repercussions in the Amazon region. These studies, in a partial equilibrium framework based on the FAPRI model, assume that growth in demand for Brazilian agricultural products, both with increasing incomes and population domestically, and rapidly growing demand from China and elsewhere for Brazilian grains, beef and biofuels, will stimulate horizontal expansion in significant areas of the country, including parts of the Amazon biome. The spatialized estimates incorporated in this model were developed by Soares-Filho and collaborators; the SimBrasil model includes micro-regional estimates of land use change to specific crops and pastoral mixes, causing direct and indirect land-use change effects on forestlands. Carbon emissions impacts are also estimated. The model indicates that cattle ranching intensification through pasture improvement from 1.0 to 1.5 AU/ha can permit the liberation of enough new land resources for agriculture, and that further deforestation could be largely avoided. Although the low-carbon study showed that agricultural expansion can take place in Brazil without further deforestation, this may not be a win-win scenario due to the indirect impacts on deforestation of the growth in the agricultural sector, as it capitalizes deforesters and increases land prices (Soares-Filho et al., 2012a).

Further studies on the impacts of pasture expansion (Bowman et al., 2012) argue that the Bank study's estimate of the investment needed to upgrade cattle intensity to liberate forest land from pressure (US\$250 billion over 20 years) would be far too costly and could not be bankrolled by carbon markets. The alternative would be to create institutional barriers and establish more stable property rights regimes over areas of remaining forest vulnerable to conversion for pasture.

A study by Mann et al (2009) indicates that the value of foregone ecosystem services in the Amazon exceeds agricultural outputs on 61% of lands used for soybean production; which strongly indicates the non-optimality of a large share of such agricultural conversion.

There has been an intense recent proliferation of opportunity and transport cost studies, including recent works by Nepstad et al. (2009), Chomitz (2007), Young et al. (2007) and Soares-Filho et al. (2010), referring to the cost of avoided or reduced deforestation. Opportunity cost curves and maps (see Figures 3 and 4 below) tend to demonstrate a fairly low and flat cost in CO₂ emissions

in terms of avoiding deforestation in extensive pastures, with net costs around US\$ 5 -10/t C, whereas areas that are under pressure for conversion to soybeans would have significantly higher costs, whose compensation would not be justified by the carbon market. On the other hand, a moratorium on direct supply of soybeans from deforested areas, in force since 2005, has created other disincentives to such conversion. Aside from an initial spurt in the first half of the past decade, soybean expansion into the Amazon has not been significant. Yet, scholars warn, the effectiveness of the moratorium could suffer, should commodity prices increase again substantially.

Note finally that there are indications that opportunity costs of forest preservation are particularly high in the Amazon region, viewed in a global perspective. A meta study by Phan et al (2012), where many of the currently available studies worldwide providing numbers for such opportunity costs were pooled, indicated that these costs may be particularly high in Latin America as compared to other regions (although admittedly the difference was not statistically significant as the sample of studies from the region was small). The study at least supports the notion that REDD implementation in the Amazon region is “hard”, and that much work lies ahead before a robust and clear path to effective and efficient REDD implementation has been found for the region.

Fig. 3. Carbon supply curve for Brazilian Amazon forests. This curve begins with carbon contained in forested parcels with the lowest opportunity costs on the left, building cumulatively to the right. (Nepstad et al., 2009).

Fig. 4. Opportunity cost map for the forests of the Brazilian Amazon region (CO₂eq.). Values indicate the forgone profits from soy or cattle ranching expressed as the net present value of the most lucrative activity divided by the difference in carbon stock of the forest and the soyfield or cattle pasture. Opportunity costs are reduced by the potential NPV from sustainable timber harvest. Amazon states are indicated by abbreviations: AC, Acre; AM, Amazonas; AP, Amapá; RO, Rondônia; MT, Mato Grosso; PA, Pará; RR, Roraima. (Nepstad et al., 2009).

3.2 Feedbacks between Deforestation, Climate Change, and Agricultural Expansion

The effects of agricultural expansion, even on agriculture itself, are in many cases far more complex than those found by a simple analysis as that offered in the previous sub-section. This is in particular the case when the area deforested, and converted to agricultural uses, is large.

The climate-driven Amazon forest dieback that is predicted to begin by midcentury (see Vergara and Scholz 2010) may have already begun through positive feedbacks between drought, fire-dependent land uses, and forest fires (Nepstad et al., 2011). A comprehensive conservation

strategy for the Amazon therefore requires careful consideration of the interactions between climate change, deforestation, and fire and relies heavily on international efforts to mitigate global warming.

With respect to agriculture, climate change can cause large geographic shifts in productivity, thereby reducing the productive area of grains (especially soy, due to the expected increased frequency of water stress events) in central and southeast Brazil by up to 40 percent (Assad et al., 2008). A net gain in cropland due to climate change in Brazil is likely to occur only for sugarcane, whose suitable lands are expected to increase by 100 percent. In general, considering nine crops evaluated by Assad et al. (2008), the economic losses from climate change are expected to be about US\$4 billion by 2020 and could reach US\$14 billion by 2070 under the IPCC-A2 scenario (Assad et al., 2008). They could be even higher, as more frequent and extreme weather events will impose a heavy burden on food security and may result in further cropland expansion.

Therefore, a positive feedback exists between climate change, decreasing crop yields, additional deforestation needed to accommodate those geographic shifts, and crop expansion due to improved levels and patterns of global food consumption.

In this regard, Oliveira et al. (2013) evaluated the resilience of the natural ecosystem and the productivity of soybean and pasture in the Amazon as a function of deforestation and increased CO₂ atmospheric concentration. In response to climate change for the year 2020, the author found changes in AGB (Above Ground Biomass) of -16%, in soybean yield of -10% and in pasture yield of -2%. With the introduction of physiological effects in simulations the changes were -7% for AGB, -4% for soybean and +1% for the pasture. In response to the change in land use, AGB declined in relation to control simulation to -37% when the deforested area was replaced by soybeans, and it decreased to 34% when the deforested area was replaced by pasture. The soybean yield and pasture productivity decreased to -18% and -29%, respectively. When all the effects were analyzed jointly, AGB declined to -37%, soybean yield to -21% and pasture productivity to -29%. For the year 2050 changes in AGB, in soybean yield and in pasture productivity were also higher in the simulations in which all effects were considered, with a 63% reduction in the AGB, 31% for soybean yield and 33% for pasture productivity in a Business-as-Usual deforestation scenario (Soares-Filho et al, 2006). The combined effects of climate change (due to changes in atmospheric composition) and change in land use caused a significant reduction in the productivity of the natural ecosystem and in the two crops analyzed. In general, the effect of deforestation was the most important factor for this reduction.

Coupling such model results with spatially-explicit land-use rent models will enable us to monetize the losses imposed by climate change and deforestation on land-use rents. For example, Bowman et al. (2012) developed a spatially-explicit rent model that incorporates the local effects of biophysical characteristics, infrastructure, land prices, and distance to markets and slaughterhouses to calculate 30-year net present values (NPVs) of extensive cattle ranching

across the Brazilian Amazon. The model is used to ask where extensive ranching is profitable and how land acquisition affects profitability. Average NPVs on deforested land range from 116 USD/ha in Amapá to 390 USD/ha in Mato Grosso. The highest net present values are in the cerrado of Mato Grosso, Tocantins, Rondônia, and around Rio Branco (AC), Manaus(AM), and Santarém (PA), and along the deforestation margin of already-established corridors such as the Belém-Brasília highway (BR-10/BR-153) and the Porto Velho-Cuiabá highway (BR-364) (Figure 5). Regions exhibiting negative net present values are only those that are completely inaccessible (*e.g.* remote regions of Amazonas and Pará). The model estimates fall within the range of existing estimates in the literature (annual profit estimates ignoring fixed costs such as purchasing animals and tractors are roughly 12-80 USD/ha), and support previous research that has found that ranching, if we ignore land investment or opportunity costs, is profitable, though only marginally so (Mattos and Uhl, 1994; Barros et al., 2002).

Fig. 5. Cattle ranching rents in the Amazon. (Bowman et al., 2012).

Vera-Díaz et al. (2007) has developed an interdisciplinary model of soybean crop rents, based on climate, soils, and economic variables. This model comprises a component of soybean yield that integrates the major climatic, edaphic, and economic determinants for soy crops in the Amazon Basin. Yield is simulated using a crop physiology model that captures the effects of climate and physical attributes on the development of soybean plantings, such as fertilizer applications, credit, and latitude. Coupled to the yield component is a rent model that deducts the costs of soybean production and transportation to exportation ports from its market price; the result is then multiplied by the expected productivity per hectare output by the yield component. More

recently, Nepstad et al. (2009) and Soares-Filho et al (2010) modified this model to simulate soybean profitability over a 30 year time-period based on variation in transportation costs due to road expansion and paving throughout the Brazilian Amazon region. Following, the authors constrained the soy rent model to produce positive rents only on land suitable for mechanized agriculture. The suitability map for mechanized agriculture takes into account four factors (Nepstad et al. 2007): The availability of flat land, appropriate soils, flood-free areas, and regions without climatic restrictions (Figure 6). See also Galford et al (2013) for a discussion of possibilities of agricultural expansion on the Amazon fringe, without substantial increased net carbon release.

Fig. 6. Soy crop rents in the Amazon. (Soares-Filho et al., 2010).

While there are obvious externalities associated with agricultural activity in the vicinity of the Amazon, it is not obvious that this always leads to serious allocation problems. Klemick (2011), studying fallow ecosystem services in shifting cultivation in the Eastern Brazilian Amazon, finds that the use of fallow is in many cases over-optimal from a private perspective, and closer to socially optimal for the wider community. This indicates that farmers in some cases may optimize returns for the entire local community and not simply for themselves. Klemick's analysis in any case indicates a large positive output elasticity with respect to fallow (around 0.15).

4. Conclusions

This paper has attempted to identify and document types of local and regional values involved in protecting the Amazon rainforest, and point out existing work to quantify such values. We have also, more briefly, considered opportunity values of converting the forest to other uses, which are mainly agricultural. As we point out, a documentation of such values is a necessary step in the process of efficient protection of the Amazon rainforest, or alternatively, efficient conversion to other uses. Crucially, for economic efficiency to have an influence in the process by which protection and conversion decisions are made, one needs to recognize all values related to protection, including those accruing to the global community outside the Amazon region. Thus values of the types documented in this paper, while forming one key basis for decisions to protect or not protect the rainforest, since they do not constitute all such values (in particular, the “global” values are left out), are not sufficient to make such decisions.

Our documentation shows that, even as reasonably good numbers exist for some value components, in some parts of the Amazon, our knowledge about these values is highly incomplete, in several aspects: in the geographical dimension; in terms of which value aspects are more or less important overall (and even perhaps what are all these value aspects); and in terms of the average basin-wide values attached to them. In addition to pinpointing where some knowledge already exists, we identify avenues for subsequent research to identify further such values, which will be challenging.

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