Global Change Biology

Global Change Biology (2009) 15, 2803–2824, doi: 10.1111/j.1365-2486.2009.02109.x

The potential ecological costs and cobenefits of REDD: a critical review and case study from the Amazon region

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Abstract

The United Nations climate treaty may soon include a mechanism for compensating tropical nations that succeed in reducing carbon emissions from deforestation and forest degradation, source of nearly one fifth of global carbon emissions. We review the potential for this mechanism [reducing emissions from deforestation and degradation (REDD)] to provoke ecological damages and promote ecological cobenefits. Nations could potentially participate in REDD by slowing clear-cutting of mature tropical forest, slowing or decreasing the impact of selective logging, promoting forest regeneration and restoration, and expanding tree plantations. REDD could also foster efforts to reduce the incidence of forest fire. Potential ecological costs include the accelerated loss (through displaced agricultural expansion) of low-biomass, high-conservation-value ecosystems, and substitution of low-biomass vegetation by monoculture tree plantations. These costs could be avoided through measures that protect low-biomass native ecosystems. Substantial ecological cobenefits should be conferred under most circumstances, and include the maintenance or restoration of (1) watershed functions, (2) local and regional climate regimes, (3) soils and biogeochemical processes, (4) water quality and aquatic habitat, and (5) terrestrial habitat. Some tools already being developed to monitor, report and verify (MRV) carbon emissions performance can also be used to measure other elements of ecosystem function, making development of MRV systems for ecological cobenefits a concrete possibility. Analysis of possible REDD program interventions in a large-scale Amazon landscape indicates that even modest flows of forest carbon funding can provide substantial cobenefits for aquatic ecosystems, but that the functional integrity of the landscape's myriad small watersheds would be best protected under a more even spatial distribution of forests. Because of its focus on an ecosystem service with global benefits, REDD could access a large pool of global stakeholders willing to pay to maintain carbon in forests, thereby providing a potential cascade of ecosystem services to local stakeholders who would otherwise be unable to afford them.

Keywords: carbon, climate, deforestation, degradation, ecosystem services, land-use and land-cover change, REDD, tropical conservation, watersheds, Xingu River basin

Received 14 August 2009 and accepted 26 September 2009

Introduction

Approximately 17% of global greenhouse gas emissions are estimated to come from the clearing and degradation of tropical forests (IPCC, 2007). In an attempt to reduce these emissions, negotiators within the UN Framework Convention on Climate Change (UNFCCC) are designing a mechanism for compensating developing (largely tropical) nations that succeed in reducing carbon emissions from deforestation and forest degradation, known by the acronym from reducing emissions from deforestation and degradation (REDD) (Gullison *et al*., 2007). This proposal is criticized, however, for its narrow focus on carbon and the concern that noncarbon

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ecosystem services (e.g., the provision and regulation of pure freshwater, biodiversity conservation, and the maintenance of soil resources) and social issues (e.g., poverty reduction and the protection of land and human rights) will be neglected or affected detrimentally (Daviet *et al*., 2007; Brown *et al*., 2008; Dooley *et al*., 2008). These important concerns require careful consideration, especially given the poor performance of previous global initiatives to protect tropical forests (Winterbottom, 1990).

One of the great challenges of environmental conservation is to reconcile human needs for food, fiber, and fuel – whose production often transforms natural ecosystems – with the need to maintain and restore ecological processes and watersheds. In the face of this complexity, recent decades have seen the development of a series of strategies largely centered on biodiversity, with an emphasis on the protection of pristine and relatively uninhabited (or low human population density) areas, explicitly avoiding managed landscapes. These strategies include those focusing on priority species (e.g., flagship: Dietz *et al*., 1994; umbrella: Wilcox, 1984; indicator: Noss, 1990; Lindenmayer *et al*., 2000; landscape: Sanderson *et al*., 2002) or geographic areas (e.g., protected areas: Terborgh, 1999; hotspots: Myers, 1988, 1990; ecoregions; Olson & Dinerstein, 1998; Olson *et al*., 2001) whose protection was thought to help maintain entire ecosystems. Increasingly, however, it became apparent that these strategies did not address problems of water supply and quality, flood control, pollination, and a host of other ecosystem services (Daily, 1997) that were impacted by large-scale landscape change, urbanization, mining, and other human activities. To address these problems, an ecosystem service-focused approach emerged, triggering a wave of efforts to assign monetary value to these services (Costanza *et al*., 1997; Daily, 1997) and establish markets or programs to pay individuals or communities to actively protect and restore these services (Landell-Mills & Porras, 2002; Wunder 2007; Engel *et al*., 2008). However, ecosystem-service-focused approaches have not gained broad penetration in large part because formal markets for the services have been difficult to institutionalize, among other challenges (Wunder, 2007).

Forest carbon has the potential to both increase the scale of tropical forest conservation and significantly reduce greenhouse gas emissions. The storage and sequestration of carbon by tropical forests may be the first ecosystem service with the potential for a broad, global-scale market. REDD could establish a flow of revenue to tropical nations that is far greater than existing flows of international conservation funding, but that is performance-based and only accrues to participating nations as they rigorously demonstrate their success in maintaining or expanding their forests. The largest pool of carbon that is susceptible to release to the atmosphere through land use change is in trees, which are also the dominant organisms in forests; thus, the emerging REDD regime has the potential to greatly increase the carbon density of the world's remaining tropical forest by slowing forest clearing and degradation.

By attracting revenues from the rapidly expanding carbon market, whose transactions already reached \$125 billion in 2008 (Point Carbon, 2009), REDD has the potential to protect other services for which no market or other funding of this scale exists. Funding for international conservation (an average of <\$1 billion annually in the 1990s and declining even further in the early 2000s; Molnar *et al*., 2003; Wunder, 2006) represents only a small fraction of the value that the global carbon market has already achieved. This market is projected to reach over \$600 billion by 2013 (SBI, 2009). With early funding for REDD already surpassing \$6 billion (FCPF, 2009; GON, 2009), and an expanding role for REDD as a possible cost containment mechanism in national cap-and-trade policies designed to lower greenhouse gas emissions (e.g., USHR 2454, 2009), financial flows associated with REDD have already surpassed conventional conservation funding. REDD's great potential, then, lies in the possibility of engaging stakeholders from nations around the world in a strategy that conserves and restores tropical forests, benefiting those who depend on these forests as it reduces greenhouse gas emissions.

In this paper, we review the potential for REDD to provoke ecological damages and to promote ecological benefits beyond the maintenance and enhancement of tropical forest carbon stocks. We address three questions: (1) What are the potential ecological costs of the specific interventions that may be carried out under the aegis of REDD and how can they be mitigated? (2) What are the potential ecological cobenefits of REDD interventions? (3) How can these potential costs and cobenefits of REDD be measured and incorporated into monitoring programs? We then examine these potential ecological cobenefits through analysis of possible REDD program interventions in a large-scale Amazon landscape. Finally, we discuss the possible integration of ecological cobenefits into the REDD architecture. Together with safeguards to prevent socioeconomic and cultural harm and provide benefits to local communities and indigenous peoples, there is considerable scope for improving the positive ecological impacts of REDD with small changes in program design and monitoring.

Management interventions under REDD

This review focuses on the types of land management interventions that are most likely to be promoted under the REDD regime that is currently under negotiation within the UNFCCC. We focus on REDD because it is potentially the most far-reaching and powerful policy instrument for influencing tropical forest carbon stocks and, hence, tropical forest conservation. It should eventually link emissions reductions from tropical deforestation and forest degradation to the UNFCCC framework in which dozens of nations adopt binding targets for reducing their greenhouse gas emissions (Gullison *et al.*, 2007), with industrial nations reducing the costs of achieving these targets in part through offsets to tropical nations with REDD programs. Industrial nations could also contribute to reduction of emissions from forests in developing nations through contributions to a voluntary fund, or possibly through a levy on emissions trading. In the near term, the flow of revenue into programs designed to reduce emissions from tropical forests could be greatest from individual states (e.g., California) and nations (e.g., the United States) that are unilaterally developing legislation that would impose caps on greenhouse gas emissions that could be partially realized through international offsets, including payments to tropical nation programs based on REDD design principles (USHR 2454 2009, GCTF 2009). These offset programs will probably follow many of the design features of REDD, providing a further rationale for structuring this review around the most likely elements of the UNFCCC REDD regime.

The potential ecological costs and cobenefits of REDD will depend upon the types of land management interventions that this regime will eventually allow, which is the focus of considerable debate within the UNFCCC. REDD negotiations have converged on the need to maintain or enhance forest carbon on lands that are forested or that once were forested, but have acknowledged that the REDD regime should ultimately be compatible with emissions reduction protocols that are under development for the agricultural sector (Meridian Institute, 2009). Since the official launching of forest carbon talks within the UNFCCC in Montreal, in 2005, negotiators have expanded beyond their initial focus on reducing the rate at which tropical forests are cleared to include reductions in emissions from forest degradation (the second 'D' of REDD), forest conservation (i.e., the creation and management of forest protected areas), and the 'enhancement' of forest carbon through forest restoration or regeneration. This suite of activities is referred to as 'REDD + ' (Angelsen & Wertz-Kanounnikoff, 2008; AWGLCA5, 2009). Allowable interventions can therefore be categorized as reductions in *negative changes* in tropical forest carbon that are achieved by slowing the rate of clearing and degradation, and *enhanced positive change* in forest carbon, through increases in the area of forest (forest restoration) and increases in the carbon density of tropical forests (rehabilitation, forest restoration, and sustainable management of forests) (Angelsen & Wertz-Kanounnikoff, 2008; Meridian Institute, 2009).

Negotiations have also dealt with the issue of REDD program scale, with some nations favoring a REDD regime that is restricted to nation-wide emissions reductions and others preferring subnational programs. Our review is written with the assumption that REDD will eventually compensate nation-wide reductions in carbon emissions from deforestation and forest degradation, as originally conceived (Santilli *et al.*, 2005), because of the risk of 'leakage' from subnational programs. We acknowledge, however, that many tropical nations may begin their REDD programs with subnational approaches.

We evaluate the potential ecological costs and cobenefits of five different land management interventions by which a nation might pursue reductions in carbon emissions that would qualify it to receive REDD funding. First, REDD could be achieved by slowing deforestation – the clear-cutting of mature tropical forest – that is associated with forest conversion to crops and livestock, and is the largest source of carbon emissions to the atmosphere from tropical forests (Houghton, 2003). Second, nations can slow or restrict selective logging, or decrease the impact of logging through 'reduced impact' harvest practices (Holdsworth & Uhl, 1997; Holmes *et al.*, 2002; Putz *et al.*, 2008). Third, nations can reduce the incidence of fire in standing forests, which can release a similar amount of carbon to the atmosphere as deforestation during dry years (Page *et al.*, 2002; Alencar *et al.*, 2006; Hooijer *et al.*, 2006). Although the reduction of forest fires is not explicitly identified as a land management intervention within the REDD or REDD + mechanisms under negotiation, we anticipate that nations will attempt to reduce the incidence of forest fire to protect the forest carbon that is favored by other allowable interventions, thus addressing one important aspect of permanence. Fourth, within the 'REDD + ' regime (AWGLCA5, 2009), nations can increase regeneration and restoration of native forest, sequestering carbon in regrowing forests in approaches like those developed by Costa Rica (Sanchez-Azofeifa *et al.*, 2007). Finally and most controversially, nations may tap REDD funding through the expansion of their tree plantations, which is the main avenue by which some nations (e.g., China) may participate in REDD.

The potential ecological costs of REDD and their mitigation

Most of the interventions that should be favored by an eventual international REDD regime present little or no direct threat to natural ecosystems. Slowing the rate of deforestation, slowing the rate and per-area damage associated with selective logging, declines in the incidence of forest fire, and increases in the speed of forest regeneration on degraded land should provide substantial ecological cobenefits, as reviewed below, with few, if any, ecological costs. One potential negative effect could occur in fire-adapted ecosystems, such as tropical woodlands and savannas, where fire suppression and biomass accumulation could lead to the local disappearance of plant and animal species that depend upon periodic burning (Moreira, 2000; Oliveira & Marquis, 2002; Hoffmann *et al*. 2003).

Perhaps the greatest threat of ecological damage associated with REDD is the displacement of forest clearing for livestock and grazing land away from high biomass forests into lower biomass ecosystems, a particularly detrimental form of leakage. Low-biomass, high-diversity native ecosystems, including savannas, woodlands, grasslands, and transition forests, could become inadvertent victims of REDD programs (Brown *et al*., 2008; Miles & Kapos, 2008). This threat operates at two different scales. Within a tropical landscape, the preferential conservation and protection of high biomass forests could deflect deforestation towards low-biomass native ecosystems, even if they are of higher value for biodiversity conservation, soil conservation, or water regulation. At the scale of regional and global economies, REDD could reduce the availability of land for agricultural expansion, pushing food prices higher, increasing the demand for new agricultural and grazing lands in low-carbon ecosystems (Miles & Kapos, 2008; Nepstad & Stickler, 2008).

Another, potentially equally damaging, threat posed by REDD is the replacement of native ecosystems by monocultural tree plantations. For example, the species-rich cerrado woodlands and savannas of Brazil are already being replaced by plantations of *Eucalyptus* species, native to Australia, and at least one project to earn carbon credits from this process is already underway (Mansourian *et al*., 2005). The ecological effects of tree plantations vary depending upon the type of ecosystem that the plantation is replacing. Monocultures of tree species usually support fewer native plant and animal species than the native ecosystems that they replace (Barlow *et al*., 2007). They often require heavy machinery for establishment and management, fertilization, and pesticides, increasing the risk of soil degradation and chemical contamination (Bruijnzeel, 1990; Schwarzenbach *et al*., 2006). Under some conditions, can also provide important ecological cobenefits, however. For example, tree plantations established in already degraded landscapes can attract seed dispersal agents, catalyzing the regeneration of plant and animal communities (Parrotta *et al*., 1997). They can restore the high evapotranspiration rates that are typical of mature native forests, potentially reducing streamflow and the risk of flooding. It is clear that the expansion of tree plantations for the sake of carbon sequestration poses real ecological risks, especially when they displace low-carbon native ecosystems. It is important to point out that plantations are intended to be cleared in the future, thus not representing permanent habitat or protection for water resources. But a balanced appraisal of tree plantations must discern both benefits and ecological costs.

To mitigate the losses to biodiversity and other ecosystem services that might be brought on by the leakage effect described above and by the substitution of low-biomass native ecosystems with monocultural tree plantations, nations participating in REDD could be prohibited from clearing native vegetation or 'high conservation value' terrestrial ecosystems (HCV Network 2009) for agricultural expansion or for the establishment of plantation forests. Furthermore, participating nations should demonstrate that measures to protect areas of high biodiversity, regardless of biome, have been undertaken. The sister convention to the UNFCCC, the United Nations Convention on Biological Diversity (UNCBD), already issued a decision in 2008 calling on parties to the UNCBD, non-party governments, and international organizations to ensure that REDD support the aims and implementation of the UNCBD, provide benefits for forest biodiversity, and involve biodiversity experts in REDD program design (COP9 2008). However, parties to the UNFCCC bear the major responsibility to ensure that a tropical forest carbon incentive mechanism does not become a perverse incentive to endanger or diminish the ecological integrity and value of lower biomass ecosystems.

In the long term, the growing global demand for food that is driven, in part, by both population growth and increasing levels of affluency in emerging economies (Nepstad *et al*., 2006; Nepstad & Stickler, 2008) will exacerbate the tendency of REDD to displace agricultural expansion into low-biomass ecosystems, demanding a more systemic solution. For example, the intensification (i.e., increase in the yield per area) of agricultural and livestock production on existing cleared lands could allow growing global demands for food, fuels, feeds, and fiber to be met without expansion of the area of cultivation and grazing (Steinfeld *et al*., 2006), although intensification is not without its ecological costs, either. The long-term success of REDD and the mitigation of its potentially most damaging side effects may depend upon a more comprehensive solution to the growing global land shortage that is facing humanity. In the absence of such a comprehensive strategy, the negative effects of REDD could extend beyond the loss of low-biomass ecosystems to include greater hunger and rising food prices. However, it is important to remember that these same effects are implicit in any conservation strategy that protects native ecosystems from the expanding agricultural frontier.

Ecological cobenefits

Beyond these potential deleterious ecological effects of REDD are important potential ecological cobenefits that include the maintenance and restoration of hydrological functions, local climate regimes, soils, and native species assemblages through both direct and indirect effects (Fig. 1). Many of the cobenefits of REDD are best understood within the context of watersheds, the natural drainage units of the landscape. The output of water, energy, and minerals from a watershed is regulated by the ecosystems that occupy it (Bormann & Likens, 1979) and therefore strongly influenced by REDD interventions.

The long-term fate of many tropical forests and their carbon stocks – including many Amazon forests – will be determined by the speed with which near-term 'forest dieback' proceeds (Nepstad *et al*., 2008). Climatologists have presented evidence of a possible late-century shift in terrestrial biomes that is driven by the accumulation of heat-trapping gases in the atmosphere that could include the drying of a large portion of the Amazon Basin (Cox *et al*., 2000, 2008), described below. A more conspicuous forest dieback is already underway, however, driven by land use, fire, regional climate change, and their interactions. This dieback could lead to the substitution of large areas of tropical forest by fire-prone scrub vegetation and degraded forests by 2030, releasing large amounts of carbon to the atmosphere (Nepstad *et al*., 2008; Malhi *et al*., 2009, Fig. 1). Effective REDD programs could postpone near-term

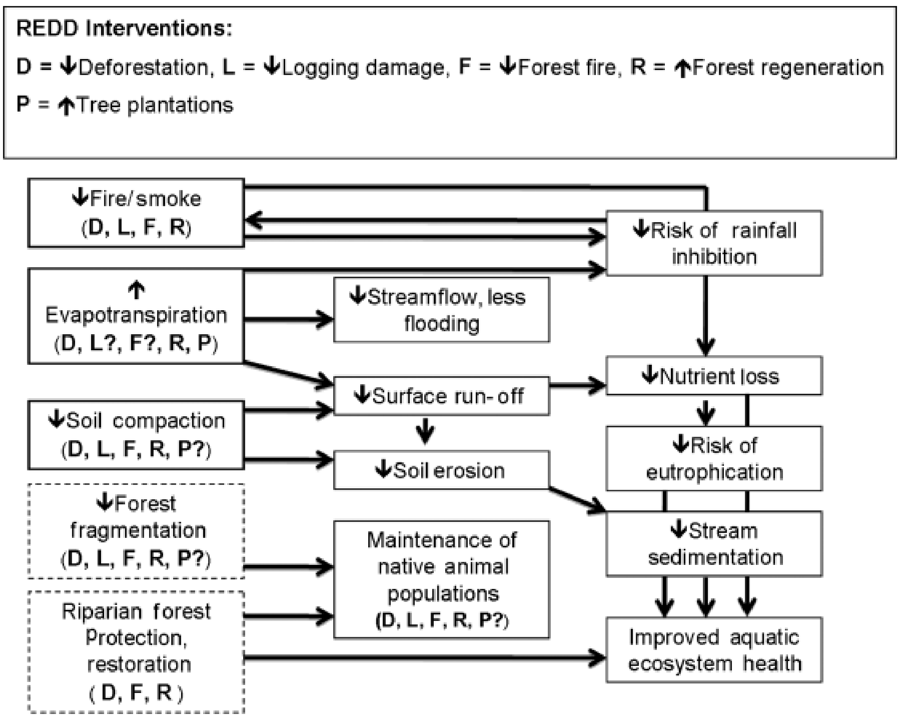

Fig. 1 Summary of potential ecological cobenefits of the five principle interventions that tropical nations could make to reduce carbon emissions from deforestation and forest degradation. Reductions in deforestation, logging damage, forest fire, and increases in forest regeneration will generally reduce the occurrence of fire in non-forest environments, increase vapor release to the atmosphere (evapotranspiration), reduce soil compaction and diminish forest fragmentation. REDD could quickly achieve benefits for stream health if it fosters regrowth or restoration of riparian zone forests. These changes will generally lead to declines in the risk of regional rainfall inhibition (relevant primarily for large forest blocks), lower annual stream discharge and flood risk, less surface run-off and associated soil erosion, and improved habitat for terrestrial and aquatic biodiversity. The loss of nutrients and sediments to streams should decline, increasing the health of these aquatic ecosystems and improving water quality. The role of tree plantations in these cobenefits will depend upon the type of plantation management practices, and the ecosystems that it is replacing.

forest dieback, since the positive feedback cycles that drive it in its early stages can be broken through management practices that reduce the risk of ignition sources in flammable landscapes (Goldammer, 1990; Nepstad *et al*., 2001; Vayda, 2006; Bowman *et al*., 2008). All REDD interventions, with the exception of tree plantations, decrease fire risk in tropical landscapes (Holdsworth & Uhl, 1997; Nepstad *et al*., 2001; Cochrane, 2003; Ray *et al*., 2005) and diminish the risk of regional rainfall inhibition by maintaining or restoring evapotranspiration.

Hydrology and water resources

Deforestation, selective logging, and forest fires affect watersheds and the streams that drain them by increasing runoff, river discharge, erosion and sediment fluxes (Fig. 1). These effects occur at the local scale and are influenced by the type of ecosystems that replace the forest and the ways in which these ecosystems are managed.

Land-use change influences the quantity of surface water resources by altering the partitioning of incoming precipitation and radiation among sensible and latent heat fluxes, runoff, and river discharge (Costa & Foley, 1997; Bonan *et al*., 2004; Li *et al*., 2007). Observations in watersheds from small ($<1\,km^2$) to medium ($1000s\,km^2$) spatial scales in the global tropics and extra-tropics show that in almost all cases deforestation reduces evapotranspiration and increases stream flow because of the reduced leaf area index, decreased root depth, and increased soil compaction that accompany forest replacement with less water demanding crops and pastures (Bosch & Hewlett, 1982; Bruijnzeel, 1990; Nepstad *et al*., 1994; Sahin & Hall, 1996; Moraes *et al*., 2006; Scanlon *et al*., 2007; Thanapakpawin *et al*., 2007; Chaves *et al*., 2008). The amount of increase depends on many local factors including the amount of rainfall, how much of the watershed is deforested, topography, soils, and the land use after deforestation, but observations indicate little effect with $<20\%$ of a basin deforested and a large increase in run-off ($200–800\,mm\,yr^{-1}$) with near complete forest removal. Tropical forest regeneration on abandoned lands restores evapotranspiration levels to that of mature forests (Hölscher *et al*., 1997; Jipp *et al*., 1998), indicating that tropical forest carbon enhancement through regeneration could help to restore hydrologic functions of the primary forest.

In addition to these water balance changes deforestation and conversion to agriculture alter the morphological and biogeochemical conditions of river systems through erosion and increased sediment flux, and can include the construction of dams that block species migration and water flow, the establishment of large cattle populations (Beaulac & Reckhow, 1982; Carpenter *et al*., 1998; McFarland & Hauck, 1999; Ballester *et al*., 2003), and the input of agrochemicals (pesticides, herbicides, fertilizers, and the chemical additives of the active ingredients) (reviewed in Nepstad *et al*., 2006). Agricultural and livestock expansion can provoke changes in vegetation and soil organic matter nutrient cycling (Neill *et al*., 1995, 2001; Markewitz *et al*., 2001; Biggs *et al*., 2004), and the development of urban populations (Vollenweider, 1971; Sonzogni *et al*., 1980; Howarth *et al*., 1996; Carpenter *et al*., 1998).

These local changes can have profound effects on the quantity, timing, and water quality of flows in even large rivers when integrated over entire watersheds. For example, analysis of discharge data in the $175\,000\,km^2$ Tocantins and $82\,000\,km^2$ Araguaia Rivers watersheds of eastern Amazonia suggest that land cover changes beginning after 1950 and culminating in the deforestation of about 50% of these basins by 2000, are associated with an approximately 25% increase in the annual mean discharge despite no significant change in precipitation (Costa *et al*., 2003; Coe *et al*., 2009). In the case of the Araguaia, a 28% increase in the sediment load was also observed and the geomorphology of the river has been fundamentally altered to more effectively transport the increased fluxes of water and sediments (Latrubesse *et al*., 2009).

Soil resources

Maintaining natural vegetation cover is one of the most secure ways of protecting soil resources. Soils not only store carbon (about 3000 Pg globally; Tarnocai *et al*., 2009), they also contain essential nutrients for plant growth, purify water, and serve as habitat for diverse flora, fauna, and microbial communities. Conversion of forest to agriculture can lead to varying degrees of soil erosion and degradation, depending on management practices and soil properties (Stocking, 2003). Deforestation need not always lead to the loss of soil and soil carbon if agricultural and pasture lands are properly managed (Neill & Davidson, 2000). Well-established soil conservation practices can minimize soil erosion in agriculture, but significant soil loss is common. We live today with the legacies of soil management and mismanagement, from ancient civilizations to recent times, with most examples of historical deforestation leading to soil degradation (Montgomery, 2007). Soil erosion is a global phenomenon, but some of the highest erosion rates have been observed in tropical regions, and the wet tropical climate is considered one of the most conducive for soil erosion (Lal, 1995). Hence, a likely cobenefit of REDD that goes beyond carbon stocks alone is the conservation of soil mass, nutrients, and habitat.

Perhaps a special case of soil loss with deforestation is that of deep organic soils in southeast Asia, where about 12 million hectares of peatlands have been drained for agriculture and for oil palm plantations (Hooijer *et al*., 2006). Drainage allows oxygen to enter previously inundated soils, thus promoting aerobic decomposition of soil organic matter. Drying of the organic soil layer also renders it more susceptible to fire. Hooijer *et al*. (2006) estimated that current emissions due to decomposition and fire in drained peatlands of southeast Asia are on the order of $0.5\,\mathrm{Pg\,C\,yr^{-1}}$; higher emissions have been estimated during years of extreme drought (Page *et al*., 2002).

Some essential plant nutrients are lost and others are redistributed within terrestrial ecosystems when forests are harvested for timber or cut and burned for conversion to agricultural uses (McGrath *et al*., 2001; Davidson *et al*., 2004). Losses from the terrestrial ecosystem include harvest products, transport of gases, aerosols, and ash following fire, trace gas emissions from soils, soil erosion, and leaching to surface and ground waters. Redistribution includes incorporation of slash and ash material into soils and sedimentation of eroded soil in toeslope positions and streams. By far the largest of these losses of nutrient capital occurs during the initial phase of biomass removal through a combination of logging and/or clearing and burning. Fire is used both for site preparation and for subsequent weed control, resulting in significant loss of nitrogen (N) and phosphorus (P) and sometimes potassium (K) through emissions of aerosols and wind-blown ash (Kauffman *et al*., 1995; Hölscher *et al*., 1997). Significant N loss also occurs through volatilization as nitrogen oxides. Mass balance studies have shown that losses of N from Amazonian forests caused by site-clearing fires are 51–62% and 7–32% the aboveground biomass N and P, respectively (Kauffman *et al*., 1995). The large fraction of biomass N that is often lost during fires depletes the pool of actively cycling ecosystem N and provokes an N limitation after repeated fire (Davidson *et al*., 2007). This loss of nutrients can slow rates of regrowth of secondary forests (Zarin *et al*., 2005; Davidson *et al*., 2007).

Although the largest nutrient losses occur with initial and repeated fire, additional modest losses of nutrients following disturbance can occur through inputs to groundwater and stream runoff and through gaseous emissions from soils. In Amazonia, increased hydrologic export of N and P has been measured in association with deforestation in small catchments (Williams & Melack, 1997; Neill *et al*., 2001) and in meso-scale watersheds (Ballester *et al*., 2003; Biggs *et al*., 2004). These effects of deforestation on water quality are also mediated by soil type (Biggs *et al*., 2004; Davidson *et al*., 2004), indicating that these responses are likely to vary across regions.

Local and regional climate

The primary goal of REDD is to maintain and potentially increase carbon in standing forests, thereby reducing the release of substantial amounts of CO_2 to the atmosphere and slowing further climate change (Gullison *et al*., 2007; IPCC, 2007). However, Global Climate Model (GCM) simulations suggest that there may be an additional mechanism by which tropical forests directly influence regional climate in a way that is unrelated to CO_2 emissions to the atmosphere. Therefore, a more immediate potential cobenefit of REDD may be its positive contribution to the protection of near-term regional climate.

GCM simulations with scenarios of future tropical deforestation indicate that the replacement of large areas of forest with other vegetation types such as grass or seasonal crops, which have greater reflectivity and lower water-demands, leads to reduced net surface radiation, decreased atmospheric moisture convergence, decreased water recycling, higher surface temperature and reduced precipitation (Dickinson & Henderson-Sellers, 1988; Nobre *et al*., 1991; Costa & Foley, 2000; Costa *et al*., 2007; Sampaio *et al*., 2007; Malhi *et al*., 2008; Coe *et al*., 2009). The fundamental changes to the energy and water cycles caused by large-scale deforestation may feed back to the atmospheric circulation and climate altering not just regional but continental-scale rainfall patterns (e.g., Nobre *et al*., 1991; Pielke *et al*., 1998; Delire *et al*., 2001) and are expected to propagate to other regions of the globe (Werth & Avissar, 2002, 2005a, b). It has been suggested that these predicted changes in climate, induced by large-scale deforestation, could produce a new climate equilibrium in many locations in the tropics that is out of balance with the current forest distribution (Malhi *et al*., 2009) and therefore threaten the existence of tropical forests in general, including those in protected areas.

The threshold at which tropical deforestation could provoke a continental-scale change in climate – or whether or not this change will take place at all – is not yet known. A large number of GCM studies of deforestation feedbacks to atmospheric circulation have been done, particularly in the Amazon Basin. Results depend on the model used and the assumptions made and have suggested that significant regional, deforestation-driven climate change can occur with 30–60% of the basin deforested (Oyama & Nobre, 2003; Costa *et al*., 2007; Sampaio *et al*., 2007). However, the deforestation threshold of climate change is easily underestimated. Simulations rarely include a full suite of feedbacks including atmospheric aerosols, dynamic vegetation, and forest fires. Any one of these feedbacks or a combination may be large enough to significantly affect forest extent and feedback negatively to regional climate (Fig. 1). For example, recent evidence from remote sensing and atmospheric studies (Andreae *et al*., 2004; Williams *et al*., 2002) indicate that dense aerosol loading in the atmosphere during periods of high biomass burning can inhibit rainfall for weeks at a time by creating an excessive concentration of condensation nuclei in the atmosphere and by reducing net solar radiation at the land surface.

Terrestrial and aquatic biodiversity

The conservation of biodiversity – defined here to mean the native assemblages of plant and animal species and their populations – is an important potential cobenefit of REDD. Habitat loss, alteration, and fragmentation are the leading causes of declines in species and populations around the world (MEA, 2005; Gallant *et al*., 2007; Sodhi *et al*., 2008; Cumberlidge *et al*., 2009). REDD interventions involving native forests should help to conserve biodiversity, although tree plantations could play an important role in restoring biodiversity in degraded lands if certain conditions in their establishment and management are met and if other, more permanent habitat is easily accessible (Parrotta *et al*., 1997; Barlow *et al*., 2007). As discussed previously, the greatest potential ecological cost of REDD with respect to biodiversity would be incurred if it provides incentives to clear or degrade lower biomass vegetation that contains high levels of biodiversity.

The greatest potential of REDD cobenefits for biodiversity conservation is through slowing deforestation. However, the same absolute reduction in deforestation rates could have dramatically different ecological cobenefits for biodiversity conservation depending on the level of fragmentation of the residual forests. In general, the ratio of forest edge to forest interior is highest in small remnants and in large remnants that have long, narrow shapes (Turner, 1989; Cook *et al*., 2002; Fischer & Lindenmayer, 2006). Species with large area requirements and those that are interior habitat specialists, avoiding modified habitats, are generally the first to disappear from fragmented terrestrial landscapes (Tilman *et al*. 1994; Woodroffe & Ginsberg, 1998; Laurance *et al*., 2001). In addition, small fragments can only support small species populations (Turner, 1989; Baguette & Schtickzelle, 2003). With greater distances between fragments, populations of species become fragmented, with fewer opportunities for genetic exchange (Cook *et al*., 2002).

Selective logging adversely affects forest structure (Putz, 1991, Costa & Magnusson, 2002) and food resource availability (Fimbel *et al*., 2001), contributing to forest fragmentation and edge effects (Gustafson & Crow, 1996; Laurance & Bierregaard, 1997) and creating barriers for the movement of arboreal organisms (Johns, 1986; Crome & Richards, 1988; Laurance & Laurance, 1996; Putz *et al*., 2001; White & Tutin, 2001). Canopy gaps caused by logging contribute to changes in the composition of understory vegetation, reducing habitat quality for understory-dependent species (Thiollay, 1992; Plumptre, 2001). Sustainable forest management, including reduced impact logging techniques, which can reduce the carbon emissions associated with logging (Putz *et al*., 2008), have also been found to significantly reduce the impacts of logging on insect and vertebrate populations (Azevedo-Ramos *et al*., 2004). Reduced impact logging can also diminish the damage to the soil, which can affect soil dwelling organisms and tree root systems. However, in some regions, notably in central Africa, animals may face increased threats from hunting as any logging regime opens access to forests to hunters (Poulsen *et al*., 2009). This will be an important issue for nations to consider as bushmeat represents an important source of protein in some forest regions.

Fires in standing forests lead to injuries and death for sedentary species, including plants, soil dwelling organisms, insects, birds, and other vertebrates. The most vulnerable species at the time of the fire are those with low mobility, poor climbing ability, and reliance on cavity nests in trees (Barlow *et al*., 2002; Peres *et al*., 2003); subsequently, understory birds (Barlow *et al*., 2002) and mid-canopy and canopy bird and monkey species (Peres *et al*., 2003) also show declines, presumably because of changes in habitat and resource (e.g., fruit, insect) availability. Thus, any interventions that reduce accidental (non-natural) fire as a way of reducing carbon emissions would also be likely to benefit biodiversity.

Finally, REDD could protect and restore landscape-level functions performed by species such as pollination (Ricketts *et al*. 2004) and seed dispersal. For example, in the tropics, up to 90% of all plant species are adapted for seed dispersal by vertebrates (Howe & Smallwood, 1982; Jansen & Zuidema, 2001). As a result, the ability of vertebrates to persist in and move around tropical forests is of great importance for natural regeneration processes, which themselves ultimately contribute to enhancing carbon in regenerating forests and maintaining carbon in native (primary or old-growth) forests.

REDD could promote dramatic cobenefits for aquatic biodiversity, especially where it leads to the maintenance or restoration of riparian zone forests and watershed function. Aquatic biodiversity may be the component of tropical biodiversity that is most vulnerable to land cover/land use change. The biodiversity of lower order streams is especially vulnerable due to its dependence on exogenous food sources and on environmental conditions created by the surrounding forest (Karr & Schlosser, 1978; Vannote *et al*., 1980; Gregory *et al*., 1991; Naiman & Décamps, 1997; Benstead & Pringle, 2004). Forest streams typically flow under the closed forest canopy and food chains develop from forest organic material (Goulding, 1980; Lowrance *et al*., 1997; Pusey & Arthington, 2003; Sweeney *et al*., 2004). Deforestation can increase solar loading, temperature, sedimentation, nutrient inputs, oxygen demand and turbidity of small streams, with important impacts on aquatic biodiversity, including sharp declines in fish diversity (Barton *et al*., 1985; Neill *et al*., 2001; Abell & Allan, 2002; Melo *et al*., 2003; Mendonça *et al*., 2005). In the tropics, vulnerability to local extinctions from forest clearing is heightened by the fact that there can be great variation in species composition between adjacent rainforest streams (Lorion & Kennedy, 2009a). The range of management interventions in streams, such as check dams, invasion by pasture grasses, direct cattle impacts, and agricultural chemicals, create physical and environmental barriers to the movement of aquatic species, altering species assemblages (Flecker, 1992; Pringle & Hamazaki, 1997). Even in larger streams and rivers, where external conditions no longer determine aquatic conditions and primary production within the aquatic ecosystem plays a more important role in the aquatic food-chain, removal of riparian vegetation can significantly reduce habitat quality for many fish species (Burcham, 1988; Bojsen & Barriga, 2002; Neill *et al*., 2006; Lorion & Kennedy, 2009b).

Monitoring ecological cobenefits of REDD

If REDD is to achieve its potential as a conservation intervention that carries with it numerous ecological cobenefits, a credible, cost-effective system for monitoring these cobenefits could increase the likelihood that they will be realized (Wunder, 2006; Engel *et al*., 2008). REDD programs will require rigorous, cost-effective approaches to monitoring of terrestrial ecosystem carbon stocks that will depend upon periodic estimates of the spatial coverage of each ecosystem type and the carbon density of each type (Meridian Institute, 2009). The measurement of these terrestrial carbon stocks is, itself, the topic of much debate under the theme 'Monitoring, Reporting, and Verification' (MRV; Achard *et al*., 2007; Gibbs *et al*., 2007; GOFC-GOLD, 2009). We assume here that REDD programs will be accompanied by high resolution mapping of vegetation cover, and estimation of vegetation carbon stocks.

In this review, we focus on two ancillary questions: (1) What additional ecological parameters could be monitored to determine the broader ecological impact of REDD programs? (2) Could an ecological cobenefit MRV system be designed and deployed and what are the tools and techniques that could be employed in such a monitoring system? Our review should not be interpreted as endorsement for a mandatory ecological cobenefit MRV system, and rather as an attempt to inform discussions of value-added proposals for REDD. Ecological indicators must be identified that can be monitored with a minimum of cost, but that reflect the ecological health of a landscape or water catchment. At the broadest scale, this monitoring system must emphasize indicators that can be detected remotely (through satellite sensors) supplemented by measurements in the field that (a) integrate the ecological status of much larger regions (e.g., stream or river characteristics that provide information about a watershed) or (b) help calibrate or validate remotely sensed monitoring approaches.

The most advanced and widely used system of standards for monitoring ecological cobenefits is the Climate, Community and Biodiversity Project Design (CCB) Standards (CCBA, 2008). The CCB standards evaluate land-based carbon mitigation projects, focusing on the integration of best-practice and multiple-benefit approaches to 'identify projects that simultaneously address climate change, support local communities and conserve biodiversity' (CCBA, 2008: 6). While these standards present an important first step towards the development of a monitoring, reporting, and verification system for the ecological performance of REDD programs, they omit many of the ecosystem-level cobenefits that are the focus of this paper. The CCB is designed primarily to address concerns about the impact of carbon projects on biodiversity. Monitoring of the potential ecosystem-level and species-level cobenefits of REDD could eventually be formalized in a standard set of monitoring protocols like the CCB standard but that is internationally coordinated and authored like the IPCC land-use guidelines. These could be adapted to fit national or regional (subnational) contexts. For example, the guidelines might recommend maintenance or restoration of riparian zones in all cases, but the protection or restoration of natural vegetation cover beyond those zones would depend on local or regional factors, such as slope and soil type.

Components of an MRV system for ecological cobenefits

The first step in developing an MRV system for the ecological cobenefits of REDD programs is to establish a point of reference for the indicators of each cobenefit before REDD interventions begin. This baseline will depend on existing knowledge of an area or on new surveys conducted in anticipation of REDD interventions. The challenge is to measure trends in water resources, climate patterns, and animal communities, which are generally more dynamic and difficult to monitor remotely than forest cover and ecosystem carbon stocks.

REDD programs will ultimately affect the entire territory of each participating nation, and direct field monitoring of the ecological impacts of REDD programs will become too costly. Measurements that could form the basis of systematic monitoring fall into two general categories: (1) those that can be measured remotely, and (2) field-based measurements that serve (a) to calibrate and validate the remote measurements, and (b) as indicators of a combination of processes (e.g., turbidity as a measure of surface run-off and soil erosion). These measurements can be supplemented with information provided by simulation models.

Remote measurements and monitoring

The considerable forest mapping that tropical nations must carry out to participate in REDD could become particularly useful, cost-effective components of an ecological cobenefit MRV system. REDD MRV will require high-resolution maps of forest cover, forest degradation (logging, fire), and estimates of associated carbon stocks updated at least every 5 years. The monitoring protocols that will be used in REDD are still under discussion (Herold & Johns, 2007; GOFC-GOLD, 2009), and are benefiting from technological breakthroughs, including high-resolution, cloud- and smoke-free imaging of forests using new radar and laser sensors (Kellndorfer *et al*., 2007; Asner, 2009; Goetz *et al*., 2009). Assessments of carbon emissions from selective logging and forest fire are particularly challenging because of the difficulty encountered in accurately mapping the area and intensity of these disturbances, but algorithms that permit the estimation of canopy thinning hold promise for overcoming this challenge (Asner *et al*., 2005; Oliveira *et al*., 2007). A great deal of ecological information can be derived from these products. For example, by overlaying forest cover and canopy density on maps of watershed boundaries, topography, and watercourses, the percent forest cover of catchments can be monitored, with the goal of maintaining forest cover above a minimum level. The protection and restoration of forests on steep slopes, areas subjected to periodic inundation, and riparian zones can be quantified. REDD forest maps can also be used to track the quality and quantity of both terrestrial and aquatic habitats by measuring forest fragmentation (including edge to interior ratios, fragment number and isolation, and proximity to water courses), and the continuity of streams (which are interrupted by dams, reservoirs, roads, and forest clearing).

The ecological impacts of REDD programs can also be assessed using remote sensing products that will not be part of each nation's forest carbon monitoring. Active fires in non-forest vegetation and smoke are already monitored using MODIS satellite algorithms (Justice *et al*., 2003). Bare soil, which is an indicator of erosion susceptibility, especially if overlaid on maps of soil type and topography, can be monitored with Landsat and other high resolution, optical sensors (Adams *et al*., 1995; Cochrane & Souza, 1998; Asner & Lobell, 2000). Techniques for remote monitoring of population-level plant biodiversity, including invasions, are also under development (Asner & Vitousek, 2005; Nagendra & Rocchini, 2008; Asner & Martin, 2009; Krishnaswamy *et al*., 2009). Remote monitoring of the temperature and turbidity of streams and rivers would provide an integrative measure of erosion and riparian zone coverage of streams (Torgersen *et al*., 2001; Sawaya *et al*., 2003), but requires very high resolution imagery and is not yet monitored systematically.

Field-based measurements

The effectiveness of remotely sensed indicators of REDD performance will depend upon rigorous field testing of the assumptions behind these indicators. Censuses, perhaps conducted in conjunction with biomass surveys, could be designed to test remotely sensed indicators and to factor out other influences on population status (such as hunting, in the case of game animals). These censuses could also assess the status of invasive organisms and disturbance-sensitive species or species assemblages. Field surveys could be facilitated by new technology, such as automated digital acoustic recording of animals that allow broader sampling at a lower cost (Acevedo & Villanueva-Rivera, 2006; ARBIMON, 2009). Because of the vulnerability of aquatic biodiversity to changes in land cover and land use, monitoring of aquatic ecosystem health is a priority. The effects of forest protection or restoration on stream physical, chemical, and biological characteristics are quite dramatic where it has been measured (Neill *et al*., 2001, 2006). Of particular importance is the identification of thresholds of stream alterations through land use beyond which disturbed stream segments pose barriers to the movement of fish guilds and other groups of aquatic organisms. The adaptation of approaches such as the biotic integrity index (Karr, 1991), which uses aquatic insect larvae as indicators of stream health, could help link satellite-based metrics with stream ecosystem health.

REDD cobenefit monitoring should also take advantage of existing field monitoring programs. Many tropical nations monitor the height and discharge of navigable rivers, for example. Where recreational, subsistence and/or commercial fisheries are monitored, the data collected could provide a low cost source of information on the status of the aquatic system, once fishing pressure and other impacts are accounted for (Almeida *et al.*, 2003). Furthermore, regular and consistent sampling can be achieved by involving local communities, sawmill workers, and farm hands, among others, in the monitoring effort (Azevedo-Ramos *et al.*, 2004). It is important to point out, however, that the type of monitoring discussed here does not necessarily allow thorough evaluation of the integrity of all ecological processes fundamental to forest functioning; rather they indicate the state of some of the elements critical to ecosystem function.

Decision-making support tools

A variety of different mapping and modeling tools that allow decision-makers to compare land-use alternatives and their potential effects on ecosystem services have been developed. Such tools include mapping, scenario development, simulation modeling, and outcome assessment components, all of which could play an important role in integrating monitoring data from space and field measurements to assess ecological changes that are difficult to measure directly. Models could help increase understanding of stream and river discharge (Coe, 2000), regional climate (Werth & Avissar, 2002, 2005a, b), vegetation responses to climate and land use (Foley *et al.*, 1996; Kucharik *et al.*, 2000; Moorcroft *et al.*, 2001), and the status of vertebrate populations in both aquatic and terrestrial ecosystems (Beissinger & Westphal, 1998; Akcakaya, 2000; Soares-Filho *et al.* 2006; Goodwin *et al.*, 2007). River discharge monitoring stations, for example, can be supplemented with automated technology to measure the temperature, height, and clarity of stream water, the temperature and pressure of the air, and precipitation. In addition, some integrated tools permit decision-makers and resource-users to compare and contrast the economic and ecological trade-offs of land-use alternatives (e.g., Natural Capital Project, 2007; Naidoo *et al.*, 2008; Nelson *et al.*, 2008; Nepstad *et al.*, 2007, 2009; Stickler, 2009). These tools typically involve mapping, scenario development (often participatory), and/or simulation modeling of both economic costs and benefits associated with different land-uses alongside modeling of the effects on ecosystem services.

Case study: evaluating alternative REDD plans in an Amazon watershed

Potential ecological cobenefits arising from REDD will depend upon the programmatic approaches that are adopted. Here, we present results of a case study in which a dynamic landscape simulation model of a large Amazon watershed, the Xingu River headwaters, is used to compare a range of ecological cobenefits under three approaches to the reduction of carbon emissions.

Study area

The Xingu headwaters region is representative of many areas along the Amazon's agricultural frontier, with expanding production of cattle and soy (70% of the area) surrounding smallholder settlements (3%) and largely-forested indigenous lands (approximately one quarter of the area) (Fig. 2). The stream and river ecosystems are under growing threats from sedimentation, agrochemical run-off, and associated fish die-off from the unprotected headwaters regions outside of the indigenous reserve, which is located at the core of the region (Fig. 2). The Basin supports *cerrado* (savanna woodland) in the south and dense humid forest in the north (Fig. 2).

The Xingu region is also an advanced laboratory for exploring the potential ecological cobenefits of alternative approaches to REDD plans. A REDD pilot project for the region is under discussion (IPAM, 2009). The project will be integrated into Mato Grosso state's plan for meeting the aims set by the Brazilian National Climate Policy, which has established a target of 80% reduction in deforestation by 2020 (GOB, 2008; Nepstad *et al.*, 2009). The region (177 780 km^2) is larger than 90% of the tropical nations that could seek participation in REDD within the UNFCCC. The Xingu is also the site of a 5-year multistakeholder campaign to protect water resources, particularly through efforts to protect and reforest riparian forests in the region (Y Ikatu Xingu, 2009). The campaign has been moderately successful, and the prospect of carbon funds represents an opportunity to continue funding and expanding efforts related to stream health and other ecosystem services that are important to the region's inhabitants.

Materials and methods

Scenarios

We compared the ecological cobenefits of two REDD scenarios that represent possible approaches to the implementation of the Brazilian National Climate Policy (BNCP) in the Xingu headwaters region. Each

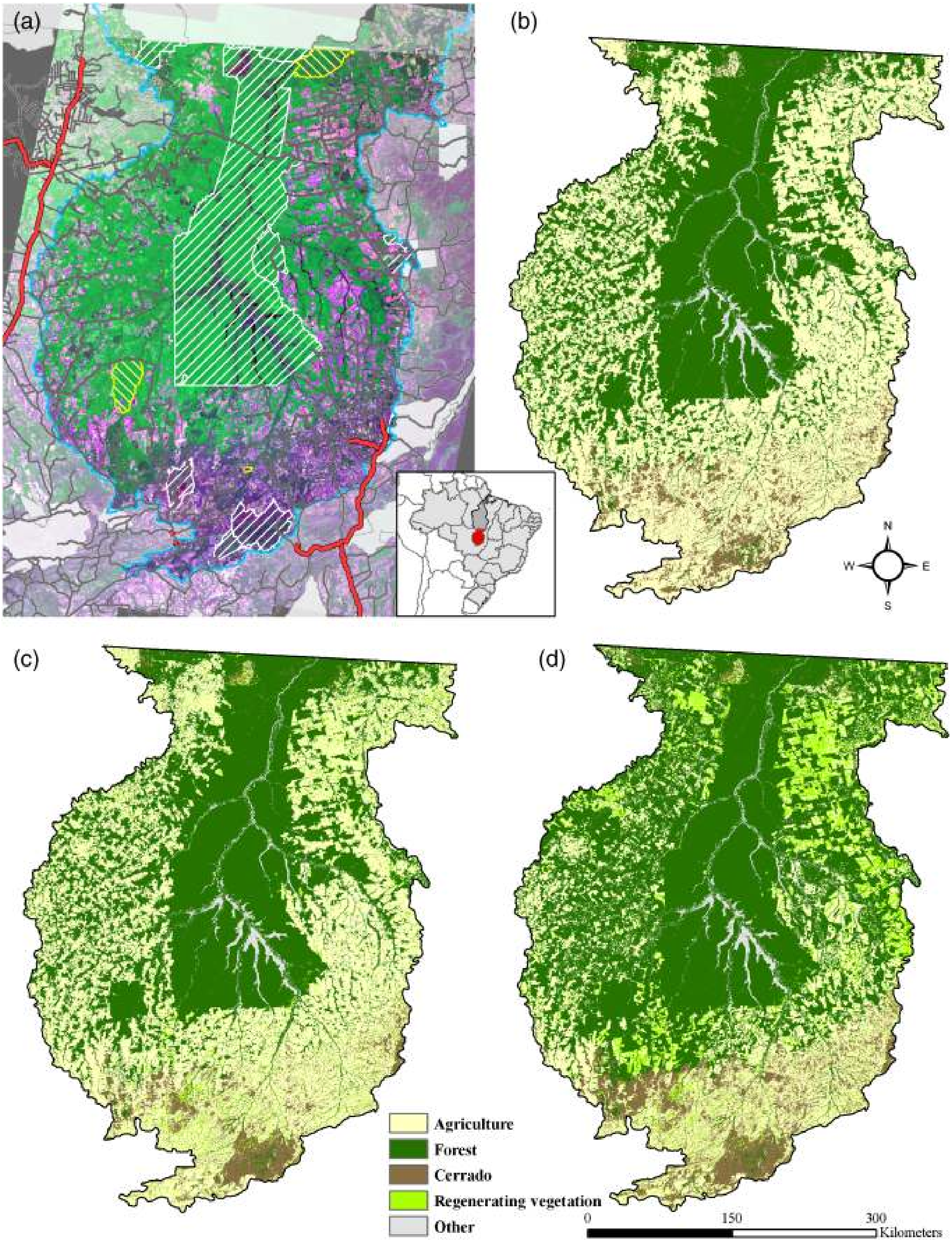

Fig. 2 (a) The Xingu River headwaters region (outlined in blue), showing federal and state protected areas (yellow), indigenous territories (white), paved roads (red), and other major unpaved roads (black). Land cover is shown for a Landsat 5 TM mosaic from 2008; greener areas indicate presence of more native vegetation or higher biomass regeneration, pinker areas indicate cleared areas or areas of low native biomass. (b–d) Comparison of alternative landscapes representing the outcome of 3 possible REDD scenarios for the Xingu River headwaters in 2020: (b) Business as Usual (BAU); (c) Protected Areas and Riparian Zone Only (PARZ); and (d) Integrated Landscape Conservation (ILC).

of the scenarios is based on existing legislation and/or on planned integration of that legislation with an emissions reduction program, and was compared with a business-as-usual simulation that assumes no REDD interventions. The basic assumptions underlying each scenario are as follows (Scenario assumptions and further methods are described in more detail in supporting information and Stickler, 2009):

- *Business as Usual (BAU)* assumes that the recent rate of deforestation (representing the average for the period 1996–2005, which coincides with the reference period set by the Brazilian government for compensation through the BNCP Amazon Fund) continues as does the current low level of compliance with environmental legislation, and thus serves as a baseline model against which to compare other options;
- *Protected Areas and Riparian Zone Only (PARZ)* represents the landscape under a carbon program that prioritizes protection of indigenous territories and other lands already designated as protected, as well as protection or restoration of designated riparian zone forests. This scenario also reflects real proposals for this region – as well as many others in the Amazon basin – since programs to compensate indigenous groups for forest conservation and private landholders for riparian zone restoration are being designed (unpublished, Rabobank);
- *Integrated Landscape Conservation (ILC)* represents the landscape under a program that takes an integrated approach to protecting carbon stocks and other ecological cobenefits, as well as targeting a broader range of land categories. This scenario is nearly identical to PARZ with the difference that it forces compliance with Mato Grosso state's zoning plan (SEPLAN-MT, 2009; Stickler, 2009), establishing a forest reserve requirement of 50–80% of private lands, adapted from the Brazilian Forest Code.

Model development

We developed a dynamic landscape model for the Xingu headwaters region that simulates forest cover under these alternative REDD scenarios (details in supporting information; Stickler, 2009). This spatial-statistical model of land-use change – developed using the Dinamica EGO modeling platform (http://www.csr.ufmg.br/dinamica/) – was derived from a land-use/land-cover change analysis and a GIS consisting of data related to the location and neighborhood attributes (e.g., distance to roads, distance to streams, slope, agricultural suitability) of four focal land-use transitions: (1) forest → agriculture (pasture or annual crops); (2) *cerrado* → agriculture; (3) agriculture-regenerating forest; and (4) agriculture → regenerating *cerrado*. The model simulates land-cover change over 14 time steps, beginning in 2007 and ending in 2020, using land-cover conversion probabilities and rates calculated from the 1996 to 2005 reference period.

Assessment of ecological cobenefits

We compared the final landscapes for each of the three alternative scenarios in terms of carbon stocks, river discharge, annual evapotranspiration, habitat quality, and water quality. We briefly describe how each indicator was assessed. Further details are presented in supporting information and Stickler (2009).

Carbon stocks under each scenario were calculated using a map of above- and belowground forest biomass developed and adapted for the entire Amazon basin (Saatchi *et al*., 2007 as adapted in Nepstad *et al*., 2007). We estimated the total deforestation-driven carbon emissions associated with each policy scenario by calculating the difference in carbon stocks between the initial landscape (2007) and each alternative scenario, correcting for the amount of carbon sequestration resulting from forest or *cerrado* regeneration (due to purposeful reforestation or to abandonment). Thus, we obtained estimates of total carbon emissions, of carbon emissions due to clearing of native vegetation, and total carbon sequestration represented by each policy alternative.

To investigate the impact of each scenario on the surface hydrology of the Xingu River in the absence of atmospheric feedbacks to precipitation, we carried out simulations with a land surface model (IBIS; Kucharik *et al*., 2000) and a river transport model (THMB; Coe, 2000). We carried out offline simulations (as described in Coe *et al*., 2009) for the final (2020) landscape maps for all scenarios, as well as for a scenario describing potential (historical) land-cover in the region, prior to settlement. We present the total volume and the percent change from the potential in annual discharge and annual evapotranspiration for each scenario.

The primary landscape measure associated with water quality is the presence of riparian zone vegetation. We present the amount of riparian forest cover and the percent of streams lacking forest cover as an indicator of the proportion of small streams that are likely to have higher temperatures and lower dissolved oxygen due to the lack of forest cover. We also present the percent of micro-watersheds having $<40\%$ forest cover as indicator of the proportion of small streams threatened with drying, sedimentation, and other changes due to landscape-level forest loss.

To evaluate differences in habitat quantity and quality among the scenarios, we calculated a series of simple landscape metrics for each landscape. We assessed quantity (total area for both cerrado and forest classes), degree of fragmentation (number of patches, mean patch size), habitat quality (total core area, total edge area, edge-to-core-area ratio), and connectivity (patch nearest neighbor distance).

Results

As expected, the more comprehensive *Integrated Landscape Conservation* (ILC) scenario achieves higher overall emissions reductions than the *Protected Areas and Riparian Zone* (PARZ) scenario when each is compared with the *Business as Usual* (BAU) projection (Table 1). This can be attributed primarily to the greater area of forest and *cerrado* woodland savanna that is maintained or restored under the *ILC* scenario. Although existing protected areas contain the highest biomass stands in the region, since both these and the riparian zones are strictly protected under both the *ILC* and the *PARZ* scenarios, the additional carbon stocks in the *ILC* are entirely attributable to protection and restoration of forest and woodlands on private lands. Perhaps most surprising is the relatively small difference in carbon stocks between the *PARZ* and *BAU* scenarios. For this region, the PARZ strategy would achieve 'avoided emissions' of only 1.6 Mt CO_2 Eq over a 10-year period, over 200 times less than the *ILC* strategy. Moreover, because the *ILC* strategy distributes forest cover more evenly across the landscape, this strategy better supports the protection of various cobenefits.

Hydrology and regional climate

All three scenarios show increases in stream discharge, ranging from 11% to 21% greater than that of the control landscape (Table 1). The *ILC* scenario has 8% less discharge than the BAU scenario, while the PARZ scenario has only 1 percent less. Mean annual evapotranspiration decreases as forest cover decreases in the scenarios, ranging from a 4% to 8% reduction from the control scenario.

Water quality

Whereas both the *PARZ* and the *ILC* had all of the riparian forests protected or restored, riparian forest cover was dramatically lower (30%) in the *BAU* scenario (Table 1). This suggests that *BAU* streams are more likely to be affected by sedimentation from point-source than those in the other two scenarios. Furthermore, in one-third of the *BAU* landscape's streams, water temperatures are likely to be have higher temperatures and lower dissolved oxygen levels (Neill *et al*., 2006), affecting species populations and assemblages. Some of these streams may also be subject to grass invasion (Neill *et al*., 2006).

Habitat

Overall, habitat quality and quantity are lowest in the *BAU* scenario, and highest in the *ILC* scenario (Table 1). The *ILC* landscape has the highest amount of total forest and *cerrado* cover, the greatest mean fragment size (more than three times as large as in the *PARZ* and *BAU* landscapes), as well as the highest amount of core or interior area (representing 89% of the total area for each landcover class). Perhaps most interestingly, the *PARZ* landscape has a higher proportion of its forest and cerrado cover (14% and 42%, respectively) in edge habitat than either the *BAU* and *ILC* landscapes. This can be explained by the higher amount of riparian forest, which necessarily increases the amount of edge throughout the region.

Conclusions

The spatial distribution of forest carbon that is protected or restored through REDD is vitally important. Although the PARZ scenario reduced emissions by <2% by 2020, it created connectivity across the Xingu headwaters through riparian zone forest restoration, which also provided shade and organic matter inputs to the streams. Protection of hydrological functions of watersheds and forest interior habitats was achieved only when severe restrictions were placed on agricultural and pasture expansion on private lands, as represented in the ILC scenario.

Discussion

The predominant potential ecological cobenefits of REDD are the maintenance and restoration of watershed function, local and regional climate regimes, soil resources, aquatic and terrestrial habitat, and improvements in water quality. In this review, we found that only one of the forms of a country's possible participation in REDD – carbon stock enhancement through tree plantations – could have negative effects on these ecosystem components, although REDD could provoke indirect negative effects by increasing the likelihood that low-biomass native ecosystems will absorb the agricultural and pasture expansion displaced from high-biomass forests (Miles & Kapos, 2008). Under all other circumstances, REDD would appear to do no harm, at a minimum, and in most cases, may carry with

Table 1 Comparison of alternative landscapes representing the outcome of three possible REDD scenarios for the Xingu River headwaters in 2020: (a) Business as Usual (BAU); (b) Protected Areas and Riparian Zone Only (PARZ); and (c) Integrated Landscape Conservation (ILC), in terms of carbon stocks, surface hydrology and regional climate, indicators related to water quality, and terrestrial habitat quantity and quality

	Scenarios		
Indicator	BAU	PARZ	ILC
Carbon			
Carbon stocks ($MtCO_2Eq$)	1536	1561	1926
Emissions since initial year ($MtCO_2Eq$)	461	460	122
Avoided emissions ($MtCO_2Eq$)	na	1.6	339
Regenerated carbon ($MtCO_2Eq$)	na	23	51
Surface hydrology and regional climate			
Mean annual discharge ($m^3 s^{-1}$) (% change from potential)	3303 (21%)	3258 (19%)	3055 (11%)
Mean annual evapotranspiration ($m^3 s^{-1}$) (% change from potential)	6379 (−8%)	6424 (−7%)	6627 (−4%)
Water quality			
Riparian forest cover (km^2)	10,859	15,497	15,497
Mean % vegetation cover per microbasin	58	59	77
% of microbasins with $>60\%$ vegetation cover	38	41	78
% of microbasins with $<40\%$ vegetation cover	35	34	10
Terrestrial habitat			
Vegetation cover (km^2)			
Forest	78,698	83,479	110,855
Cerrado	7731	10,955	15,234
Number of fragments			
Forest	34,421	33,223	13,777
Cerrado	18,197	22,064	15,571
Mean distance to nearest neighbor fragment (m)			
Forest	416	365	363
Cerrado	451	375	378
Mean fragment size (ha)			
Forest	229	251	805
Cerrado	43	50	98
Total interior habitat area (km^2)			
Forest	69,641	71,665	98,886
Cerrado	4904	6342	9762
Total edge habitat area (km^2)			
Forest	9057	11,814	11,969
Cerrado	2827	4613	5472

it substantial ecological cobenefits. This is particularly true for REDD programs that focus on slowing the destruction or degradation of old growth or mature tropical forests which are the major source of emissions that REDD is designed to reduce.

We have also established that it is possible to measure or monitor many of the ecological benefits beyond carbon storage, which makes the concept of developing MRV systems for those cobenefits a concrete possibility. This review does not explore the costs of implementing such ecological cobenefit MRV systems, and the degree to which these additional costs could exclude some tropical nations from participation in REDD. Some of the tools that are being developed to monitor, report and verify (MRV) emissions performance can also be used directly in the measurement of other elements of ecosystem function. Remotely sensed imagery can be employed to assess landscape metrics associated with indicators of ecological cobenefits (e.g., habitat fragmentation, fire occurrence and extent, soil exposure) efficiently and at a low cost. Improving remote monitoring approaches should be the target of a research and design agenda of the same level of urgency as the carbon MRV.

Finally, we have demonstrated for a large-scale Amazon landscape that REDD could foster improved watershed function, and increase habitat quality and quantity (e.g., through increased forest connectivity and reduced edge effects). The Xingu headwaters case study demonstrates that the ecological cobenefits of REDD are sensitive not only to the quantity of forests and woodlands remaining on the landscape, but also to their spatial distribution. Even small flows of carbon revenue properly targeted – for example, toward the conservation and restoration of riparian zone forests – could confer enormous ecological benefits for aquatic ecosystems. The results suggest that overall watershed function would be best protected under a more even distribution of forests and that REDD cobenefits could be maximized in the context of an integrated regional plan. However, the severe restrictions that this integrated landscape conservation scenario imposes on further agricultural and pasture expansion in the region is counterbalanced by the larger opportunity costs of foregone profits from these land-use activities (Stickler, 2009). These results also provide further support to a shift in conservation theory that posits that landscape-level conservation approaches are likely to secure ecosystem integrity better than those focused solely on protected or 'core' areas (Pickett *et al*., 1992; Poiani *et al*., 2000).

It is not clear how ecological cobenefits will be recognized and incentivized within an international REDD regime. In the full commoditization of carbon, REDD may face the same obstacles in attaching conditions to commodity trade as any other internationally traded commodity. For example, the World Trade Organization does not allow trade barriers related to environmental and social conditions (Sampson, 2000), which may make the preferential purchase of carbon credits deemed to maximize ecological cobenefits all but impossible under these rules. However, it is clearly possible to recognize and reward the ecological cobenefits of well-designed REDD programs using an approach that is under development within food commodity supply chains. Several multiple-stakeholder 'roundtables' are now finalizing the development of international standards and criteria for certifying the suppliers of soy, sugar, palm oil, and biofuel (Nepstad & Stickler, 2008). These certification systems will provide a premium, or at the very least greater market access, for the suppliers who comply with these criteria, and are reinforced as well by favorable credit status from finance institutions (Watchman *et al*., 2007). Using a similar approach, ecological standards and performance criteria for REDD programs could be developed, tested, and refined, positioning high-performing REDD projects or national REDD programs for carbon payment premiums. These standards and criteria could place greater emphasis on ecosystem-level performance measures, building upon the biodiversity-focused standards of the CCB.

Conclusions

Debate regarding the conceptualization and implementation of REDD has been vigorous and has focused primarily on determining costs (Stern, 2006; Eliasch, 2008; Kindermann *et al*., 2008), the architecture of the international agreement (Angelsen *et al*., 2008; Neeff & Ascui, 2009), the design of national REDD and distribution mechanisms (Johns *et al*., 2008; Busch *et al*., 2009), and the participation of and effects for forest peoples (Dooley *et al*., 2008). Relatively less attention has focused on how REDD might confer benefits beyond reducing carbon emissions (Peskett *et al*., 2007; Brown *et al*., 2008; Peskett *et al*., 2008). If well executed, the potential ecological cobenefits of REDD are numerous, and could improve water and air quality and wild game for low-income, rural populations. The protection of water resources, local and regional climate, soil resources, and biodiversity could contribute to the social benefits derived from REDD since they are ecosystem services on which local and regional populations depend. Because of its focus on carbon emissions reduction needed to stabilize the global climate system, REDD has access to a pool of nonlocal stakeholders who are interested in paying to maintain carbon in forests and thereby potentially provide a cascade of ecosystem services to local stakeholders who would otherwise be unable to pay for the benefits those services provide. Through the provision of ecosystem services provided directly and indirectly by conservation of forests, REDD could play an important role in maintaining or improving the quality of life of forest-dependent communities.

REDD's success both in maintaining the ecological integrity of landscapes and in achieving – or at a minimum, not detracting from – social goals will depend on adequately and appropriately addressing important issues beyond ecological cobenefits (Peskett et al., 2007). These include land tenure, equity in the design and execution of the program, and the extent to which REDD program design emphasizes multistakeholder participation and negotiated solutions. The latter might be partially addressed by engaging stakeholders in devising solutions for implementing general guidelines for protecting ecosystem services beyond carbon storage, which is likely to increase adherence to the standards (Ostrom *et al*., 1999; Dietz *et al*., 2003). REDD programs can be designed to confer important ecological cobenefits that include the provision of numerous ecological services on which local and indigenous populations rely in a way that past conservation strategies have failed to do. If the designers and implementers of REDD programs succeed in avoiding potential negative, indirect effects for low-carbon native ecosystems, this international policy instrument could successfully advance the goals of both tropical forest conservation and mitigation of climate change at an unprecedented scale.

Acknowledgements

The authors thank Tracy Johns, Nora Greenglass, Daniel Zarin, John Poulsen, and 2 anonymous reviewers for their comments and suggestions on previous versions of this manuscript, and Wendy Kingerlee for assistance with text editing. Financial support to the authors for portions of this study came from the US National Science Foundation (grants to the University of Florida and to the Woods Hole Research Center, DEB-074370), the US Environmental Protection Agency (CMS), the US National Space and Aeronautics Administration (LBA-ECO and Earth Systems Science Graduate Research Fellowship), the Gordon and Betty Moore Foundation, the David and Lucille Packard Foundation, and the Brazilian Conselho Nacional de Desenvolvimento Cientifico e Tecnologico (CNPq).

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Description of methods used to develop land-cover maps, dynamic landscape simulation model, and analyses used to compare alternative scenarios.

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