DEFENDING PUBLIC INTERESTS IN PRIVATE FORESTS: LAND-USE POLICY ALTERNATIVES FOR THE XINGU RIVER HEADWATERS REGION OF SOUTHEASTERN AMAZÔNIA

By

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To my mother and my father, who gave me the courage, the inspiration, and the freedom to pursue my dreams even when they took me far away

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DEFENDING PUBLIC INTERESTS IN PRIVATE FORESTS:
LAND-USE POLICY ALTERNATIVES FOR THE XINGU RIVER HEADWATERS
REGION OF SOUTHEASTERN AMAZÔNIA

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When native vegetation is cleared to establish agricultural lands, damage to ecosystem services such as air, water, and climate can outweigh the substantial benefits of agricultural production. Brazil has created ambitious laws and regulations for the purpose of regulating land use on private lands in Amazon forests. This dissertation analyzes the performance of the central piece of Brazilian environmental legislation in the Amazon region: the Forest Code. In the wake of escalating deforestation and international pressure in the mid-1990's, the Brazilian Government modified the Forest Code, increasing from 50 to 80% the required area of each private landholding in the region that had to be maintained in native forest. I analyzed (1) the level of compliance with both the old and new Forest Code, the change in compliance over time, the costs of compliance, and the ecological services provided under the old *versus* the new regulations; (2) the potential for hybrid regulatory-economic policies (tradable forestland development rights and land-use zoning) to reduce the opportunity costs of the modified Forest Code while protecting ecosystem services and ecological integrity; and (3) the

potential of the emerging forest carbon market to complement Forest Code and land-use zoning protection of public interests in Amazon forests. As a case study, I used the 178,000 km^2 Xingu River headwaters region in the southeastern Amazon Basin. I developed a spatially-explicit land-cover simulation model in conjunction with a river discharge model and maps of potential economic rents under soy, cattle ranching, and logging, to conduct these analyses.

When the Forest Code's "legal reserve" increased from 50 to 80% in 1996, compliance dropped immediately from 92 to 72%, then declined further to 46% by 2005. The regulatory change imposed approximately nine billion dollars in forgone profits from forest conversion to soy and cattle ranching. The Mato Grosso state zoning plan, if implemented, would potentially provide 4000 km^2 more agricultural and pasture land, reducing the opportunity costs of strict compliance with the 80% legal reserve by one third, while protecting ecosystem services similarly well. Emerging carbon markets, if expanded to fully and fungibly include forest carbon could offset much of the opportunity cost of forest conservation in the Xingu region, increasing the viability of forest conservation.

CHAPTER 1
INTRODUCTION

In addition to producing food, fiber, and fuel for the world's growing population, the agricultural sector is an important source of income to support infrastructure development and the provision of public goods throughout the world. However, when native vegetation is cleared to establish agricultural lands, damage to critical ecosystem services such as air, water, and climate can outweigh the substantial benefits of agricultural production. Nearly one fifth of world-wide, human-induced greenhouse gas emissions to the atmosphere originate from the cutting and burning of tropical forests (IPCC, 2007). On regional scales, silted rivers, deadly avalanches, mudslides and floods, displaced and decimated tribes of indigenous forest peoples, fragmented landscapes, and the loss of plant and animal species are among the negative consequences of large-scale forest conversion. In the face of projected land shortages, increased food prices (FAO, 2009) and the threat of climate destabilization (IPCC, 2007), the question of how to balance private and public gains from agriculture with the damages that often result from conversion of native ecosystems to support these activities.

A major policy response to rampant deforestation has been to set aside tracts of intact natural landscapes as parks, reserves, public forests, territorial reserves of indigenous and traditional peoples and other types of "protected areas" (Terborgh, 1999; Redford, 1994). In some places in the world, these protected areas have succeeded in defending public interests in forests (Bruner et al., 2001; Nepstad et al., 2006) while in others they have failed (Curran et al., 2004). A growing body of evidence demonstrates, however, that many of the ecosystem services provided by forests that

sustain human societies will require environmental conservation on both private and public lands, and not just lands controlled by government agencies (Soares Filho et al., 2006). An environmental conservation paradigm is needed that reconciles the defense of public interests in forest resources with the important social and economic benefits of forest conversion to agriculture and livestock expansion.

In this dissertation, I evaluate the effectiveness of legally mandated land-use restrictions on private rural property in protecting public interests related to forests and woodlands on the southeastern Amazon's agro-industrial frontier, using the Xingu River headwaters as a case study. I evaluated the economic and ecological trade-offs of the Brazilian Forest Code—along with alternatives adapted from this policy—in protecting native vegetation cover, while permitting agricultural expansion, in the study region. I then assess the potential of a range of forest carbon price signals to compensate landholders for opportunity costs and forest restoration costs that they incur under alternative policy scenarios

In response to international concern over the very high rate of Amazon deforestation in 1995, the Forest Code was modified to require that 80% of each individual property in rural Amazônia be maintained as native forest, increased from the 50% requirement that had been in place since 1965. This sudden decision threw a large number of property holders who had already cleared more than 20% of their properties into noncompliance, and was not accompanied by effective mechanisms and procedures by which these property holders could bring their properties into compliance. Policy instruments that were created to facilitate compliance with the new regulation were not implemented. These included a provision allowing landholders to

“compensate” the acreage of forest that they needed to comply with the 80% rule through the trade of land development rights, and the creation of socio-economic, ecological land-use zoning that relaxed the 80% requirement in zones of agricultural consolidation. Today, thirteen years after the historic Forest Code modification was made, there is little quantitative information regarding the level of compliance with this law, the potential economic and ecological impacts of full compliance, and the degree to which these provisions, if implemented, could help to advance cost-effective reconciliation of forest conservation with agricultural expansion.

The central goal of the research presented here was to evaluate the effectiveness of legally mandated land-use restrictions on private rural property in protecting public interests related to forests and woodlands on the southeastern Amazon’s agro-industrial frontier, using the 178,000 km^2 upper Xingu River basin region as a case study. I evaluated the economic and ecological trade-offs of the Forest Code—along with alternatives adapted from this policy—in protecting native vegetation cover, while permitting development of important economic activities, in the study region. Specifically, I asked the following questions:

- Does the current legislation to regulate land-use (especially forest clearing) reach its stated objective in practice?
- Does current legislation balance public and private interests in conservation and agricultural production on private lands in theory? If not, could it be adapted to be more effective?
- How are future land-use/land-cover trajectories likely to vary in response to a suite of plausible future policy scenarios derived from the Forest Code?
- What are the environmental and economic trade-offs associated with those scenarios?

The Xingu River is one of the Amazon Basin's major southeastern tributaries. This landscape is three times the size of Costa Rica, larger than 90% of tropical nations, and provides a mesocosm of the range of actors and land categories encountered in the Brazilian Amazon, including modern agro-industrial farms, indigenous reserves, nature reserves, cattle ranches, and smallholder settlements (Jepson, 2006). In selecting a region defined by watershed boundaries, I was able to examine the linkage between land-cover and river discharge, which is strongly regulated by forests. The Xingu headwaters region also spans across two major biomes, with the closed-canopy forest in the upper (northern) two-thirds, and the cerrado woodland/savanna biome across the lower third.

Chapter 2 of this dissertation provides an assessment of the effectiveness of the Brazilian Forest Code in protecting forests on private lands in the Xingu River headwaters. In particular, I focus on the legal and ecological effectiveness of the change in the policy that took place in 1996. This study consists of two parts, a qualitative analysis and a quantitative analysis. First, I describe the characteristics of the resources that are the object of protection under the Forest Code and discuss the implications for effective policy structures for resources that represent both a private and a public good. Next, I describe the history and requirements of the Forest Code to gain an understanding of the policy's characteristics with respect to governing a common pool resource. Finally, I carry out a quantitative analysis focusing on three aspects of compliance with the Forest Code: (1) the level of compliance with the Forest Code for a landscape on the southeastern Amazon frontier; (2) the cost of compliance to landowners; and (3) the protection of ecosystem services and the conservation of

forested landscape integrity at three moments in history and under full compliance with the 80% and 50% legal reserve requirement in modeled landscapes. I interpret these quantitative results in light of the political process surrounding the creation and implementation of the decision to increase the legal reserve requirement to 80%.

Chapter 3 evaluates the potential for facilitating landholder compliance with the modified Forest Code without forfeiting the environmental protection potentially provided through two policy provisions that could substantially lower the economic opportunity cost of compliance: tradable forest development rights and state-level socio-economic, ecological zoning (ZSEE). First, I describe these two policy instruments and compare them with the Forest Code, analyzing the extent to which they represent economic instruments or exclusively regulatory instruments. Next, I carry out a quantitative analysis of each instrument, focusing on the balance between agricultural expansion and forest conservation. I compare modeled landscapes representing the land-cover consequences of full implementation of each of the three policy alternatives in terms of area available for agricultural activities, the opportunity costs and forest restoration costs of compliance with each theoretical landscape, and the extent to which ecosystem services are maintained. This chapter highlights the potential of hybrid regulatory-economic policies to reconcile public and private interests in private-land forests, but identifies the need for additional economic incentives to fully achieve this reconciliation potential.

Chapter 4 examines the potential of payments for forest carbon to complement the Forest Code and its hybrid provisions in promoting land use decisions that protect public interests in private forest while allowing for agriculture and ranching on a designated

area of land that is suitable for these activities. More specifically, I estimate the per unit price of forest carbon that would be necessary to compensate landholders for the costs (both opportunity and forest restoration) incurred in complying with land-use legislation under three policy scenarios: the reduced legal reserve (i.e., a 50% legal reserve in the forest biome compared to the current 80% requirement, the current Forest Code (80% legal reserve requirement, without tradable development rights), and the ZSEE scenario. I estimate the difference in forest carbon stocks (both remnant native ecosystems and enhanced carbon in restored forests) between each simulated policy scenario and three simulations of business-as-usual deforestation and calculate the carbon price using the costs that are associated with each policy scenario.

CHAPTER 2
DEFENDING PUBLIC INTERESTS IN PRIVATE FORESTS: COMPLIANCE, OPPORTUNITY COSTS, AND POTENTIAL ECOLOGICAL PERFORMANCE OF THE BRAZILIAN FOREST CODE IN THE SOUTHEASTERN AMAZON

Introduction

In 1995, deforestation in the Brazilian Amazon hit a record high of 29,000 km^2 cleared in that year (INPE, 2009)—shortly after Brazil hosted the heralded United Nations Conference on Environment and Development (UNCED) in 1992. News of the Amazon forest's accelerating destruction precipitated an international crisis, and the Brazilian government responded with widely publicized policy interventions[1], the most prominent of which was the increase from 50 to 80% forest cover of rural properties in the "legal reserve" mandated by the Brazilian Forest Code in the Amazon region. The effectiveness of this dramatic change in environmental policy for maintaining forests and the services they provide is still poorly understood even as the policy itself is hotly contested (Alencar et al., 2004).

Three critical assumptions are inherent in the change in the Forest Code as the pillar of the policy response. First, it assumes that the Forest Code is fundamentally the correct approach to protecting forests and the services that they provide. Secondly, and in conjunction, it assumes that the effectiveness of the policy is primarily a matter of adjusting the proportion of each landholding to be maintained in a private reserve. Thirdly, it assumes that there will be compliance with such a dramatic change in policy with no additional provisions or incentives to assist landowners in achieving compliance. The latter assumption may be one of the most important reasons for the heated debate

[1] The other major interventions were (1) the declaration of a 2-year moratorium on the harvest of mahogany, and (2) the prohibition of new clearing on properties already possessing "abandoned or under-utilized" areas or areas "used inappropriately with respect to the capacity of the soil."

over the policy change that has raged unabated since its inception in 1996 (Fearnside, 2008).

The change in the Forest Code brings into stark relief the challenge of balancing the protection of native vegetation and ecosystem services, while permitting development of important economic activities. From the perspective of (primarily) the agricultural lobby and its supporters, the new legal reserve requirement makes it economically unviable to maintain a working ranch or farm in the Amazon biome and jeopardizes the regional economy and the provision of public services (Camarga, 2008). For environmentalists and social justice advocates, the new policy is necessary to counter the alarming increase in deforestation in recent years and its possible consequences for biodiversity, regional climate stability, greenhouse gas emissions, air pollution, soil degradation, local and regional water quality, and in turn, human health. The critical question is what can be done to balance the numerous private and public gains from agro-industrial production with the damages borne by society as a result of these activities.

Typically, the approach to protecting public interests in forests has been to put them under state ownership for either strict protection or limited use for timber harvest. State control over timber harvests on public lands has frequently led to severe degradation of forests, as state oversight has often been inadequate to prevent open access regimes from developing, and as cronyism and corruption have influenced the allocation of forest concessions (Barbier and Burgess, 1997). The effectiveness of parks and other protected areas in controlling deforestation has been mixed. Examples from Brazil (Nepstad et al., 2006) and Costa Rica (Sanchez-Azofeifa et al., 2003) indicate

that protected area status is associated with decreased deforestation rates, whereas parks in Indonesia have been subject to massive deforestation (Curran et al., 2004). Increasingly, forest ownership or usufruct rights by indigenous and other traditional communities have been officially granted in common property arrangements that recognize not only the customary rights of these groups but also assume that these groups will more effectively protect forests if they implement customary institutions for regulating resource use. The resulting indigenous lands, extractive reserves, and community reserves encompass almost one third of the forests of the Brazilian Amazon and large percentages of the remaining forests of most tropical nations, although the enforcement of these reserves varies greatly.

Although strongly advocated by economic theory (and more famously, Garrett Hardin (1968) in *The Tragedy of the Commons*), private property arrangements for forests in the tropics are more rare: only 10% of tropical forests are held by firms or individuals (White and Martin, 2002; Agrawal et al., 2008). Proponents of individual property rights in forested frontier areas argue that secure tenure arrangements should induce more rational forms of exploitation of natural resources and a reduction in deforestation rates (e.g., Southgate and Whitaker, 1992). Establishing individual property rights does not automatically lead to reduced deforestation, however, especially when the profitability of agricultural and ranching activities outweighs that of forest management (Jaramillo and Kelly, 1997). Restricted individual property rights have been suggested as one way to protect forest resources on private property (Jaramillo and Kelly, 1997). Most Latin American countries place restrictions on tree cutting and require permits for any type of forest use. Many also require that riparian

forests and forests on steep slopes be strictly protected. Brazilian and Paraguayan legislation goes one step further in requiring private landowners to maintain a certain percentage of the native forest or woodland vegetation in a type of private reserve.

In Brazil's Legal Amazon, 29% of the region is designated as indigenous land and protected areas, about 45% is considered public untitled land (*terra devoluta*), and private lands make up approximately one quarter of the total area (Lentini et al., 2003). In contrast, in Mato Grosso, private lands comprise more than half of the total area, whereas indigenous reserves and state or federal protected areas constitute a total of 18% (Congresso Nacional, 2000). Thus, forest and woodland protection on private lands is of paramount importance in this region. However, it is unclear whether the Brazilian Forest Code—the primary instrument for protecting forests on private lands—is adequate to protect common pool resources, given the huge incentives to remove native vegetation in the face of agro-industrial growth, particularly following the increase in the proportion of private reserve to be maintained by each landholder.

In this chapter, I present the results of an assessment of the legal and ecological effectiveness of the revised Forest Code. This assessment consists of two parts, a qualitative analysis and a quantitative analysis. First, I describe the characteristics of ecosystem services and discuss the implications for effective policy structures for ecosystem services. Next, I describe the history and requirements of the Forest Code to gain an understanding of the policy's characteristics with respect to governing ecosystem services. Finally, I carry out a quantitative analysis focusing on three aspects of compliance with the Forest Code: (1) the level of compliance with the Forest Code for a landscape on the southeastern Amazon frontier; (2) the cost of compliance to

landowners; and (3) the protection of ecosystem services and the conservation of forested landscape integrity at three moments in history and under full compliance with the pre-1996 and post-1996 legal reserve requirement in modeled landscapes.

Governance of Common Pool Resources

The fundamental premise of the Forest Code is that forests and forest resources constitute a common good and therefore, the rights of land owners to dispense with the land as each sees fit must be balanced with a set of responsibilities to protect this common good. At the time the first version of the Forest Code was drafted and signed into law in 1934 (described more extensively below), there was widespread concern with erosion, loss of soil fertility, and the sedimentation and degradation of water bodies (Dean, 1995). In addition, when the legal reserve was first introduced, its original objective was to ensure that a minimum reserve of forest resources would be available on each property to supply local firewood, charcoal, and timber markets (Azevedo, 2009). Early on, the Forest Code also cited concern for the conservation of native species of flora and fauna. The major innovation (particularly for its temporal context) of the Forest Code is that it defined forests as a common good in relation to the services that flow from them because these services, including the prevention of sedimentation, the maintenance of water quality, and the conservation of native ecosystems, benefit people in society other than the private landholder. As a consequence, the expansion of agro-industry in the Amazon constitutes a “dilemma of the commons” (Hardin, 1968; Ostrom et al., 1999), in which the costs and benefits of the use of natural resources common to many are not equitably distributed among the users and inhabitants of the region. The costs include the loss of ecosystem services due to the large-scale conversion of native vegetation to pasture and cultivated lands that were not yet

understood at the time the Forest Code was drafted, including increased carbon emissions and potential destabilization of regional hydrological and climate systems (Werth and Avissar, 2002; Alencar et al., 2004), and the displacement of traditional peoples and small producers (Hecht, 2005; ISA, 2005).

The defining features of common-pool resources, such as the resources and services provided by forests, are that they are non-excludable and rival resources. An excludable resource is one whose ownership allows the owner to use it while simultaneously denying others the privilege (Daly and Farley, 2006). In contrast, when no institution or technology exists to make a good or service excludable, it is known as a non-excludable resource. Whereas the individual trees constituting the forest may be considered as excludable, the services that flow from them (e.g., maintenance of hydrological and climate regimes, clean air and water, habitat) are non-excludable. Rivalness on the other hand, is an inherent characteristic of certain resources whereby consumption or use by one person reduces the amount available for everyone else, or reduces the quality of the resource. Thus, a rival resource is one whose use by one person precludes its use by another, and a nonrival resource is one whose use by one person does not affect its use by another. Unlike excludability, rivalness is a physical characteristic of a good or service and is not affected by human institutions. Obviously, if trees are harvested for timber, they cannot be used by another person in the same way. However, ecosystem services that are maintained by forests are not traditionally considered to be rival, as the use of air by one person, for example, does not diminish the amount of air available for use by another. However, the quality or quantity of a service may be diminished by the use of the rival resource (trees, in this case) from

which the services flow. As such, it is useful to consider the services maintained by forests as rival, as well.

The combination of these characteristics means that substantial free-riding is likely if people follow their own short-term interests exclusively, as assumed by Hardin (1968). “Free-riding” refers to natural resource use that maximizes private gain to the detriment of the common good and takes the form of (1) overuse without concern for the negative effects on others, and (2) a lack of contributed resources for maintaining and improving the common pool resource itself (Ostrom, 1990). To avoid these outcomes, effective rules to limit access and to define users’ rights and responsibilities regarding the resource must be established, and incentives for users to invest in the resource instead of overexploiting it must be created (Ostrom et al., 1999).

In this paper, I consider the extent to which these two conditions of limiting free-riding (effective rules and incentives) are met with respect to protecting forests. I specifically discuss the rules and issues related to the Forest Code and its application in Mato Grosso in the subsequent section. However, it is useful to point out some general issues regarding environmental governance in Brazil and the Amazon that affect the implementation and effectiveness of the Forest Code as well as other environmental legislation.

Environmental Governance in Brazil

In Brazil, the use of regulatory instruments has been the dominant policy approach to environmental problems. Regulatory (often referred to as “command-and-control”) instruments typically define performance-based or technology-based standards with which producers or landowners are required to comply by law. Regulation can take a variety of forms, including the outright prohibition of an activity or substance (e.g., the

use of the pesticide DDT), or setting limits for the amount of a pollutant that may be produced (e.g., arsenic in drinking water), or defining the terms by which natural resources can be extracted (such as the length of the fishing season or the type of equipment that may be used in the fishery (Sterner, 2003). Failure to comply with regulations generally involves fines or other penalties. Thus, monitoring and enforcement are critical components of direct regulation. This may be relatively easy in some cases, for example, where regulators must simply check that a firm or individual has installed a required piece of equipment. However, monitoring becomes more complicated in cases where compliance with regulations is more difficult to observe (e.g., non-point source pollution). Moreover, if adequate numbers of trained personnel for monitoring and enforcement are not available, and if the force of law cannot be brought to bear on violators, adequate or sufficient monitoring and enforcement may become unviable.

Critics argue that environmental policy based on regulation tends to be inefficient because it generally does not capitalize on the rent-seeking behavior of resource users (Sterner, 2003). Direct regulation is accused of being too prescriptive, and not providing enough flexibility to firms and individuals to innovate and meet standards in the most cost-effective way possible. Regulations are often ineffective, as well, because of budget short-falls and the complicated architecture of carrying out monitoring and enforcement, as well as sometimes sizeable economic incentives to circumvent legislation. Moreover, regulations may also create perverse incentives. For example, when ambitious legislation is not accompanied by adequate enforcement, legal resource users are, in effect, penalized relative to the fraudulent resource users who

incur lower costs of production (Carter, 2001; Tietenberg, 1996). Nevertheless, direct regulation may be the most appropriate method for integrating the biological requirements of some natural resources into environmental policy. For example, fish should not be caught during spawning season and timber harvesting should not eliminate the seed stock of tree populations.

Although Brazilian legislation is considered to be among the most sophisticated and advanced in the world (Hochstettler and Keck, 2007), natural resource and environmental policies have been difficult to implement and enforce in practice. This can be explained in large part by the mutually reinforcing characteristics of (1) the Brazilian legal system—which follows the civil law tradition—and (2) the “command-and-control” nature of most environmental legislation in the country (Campari, 2005; Ames and Keck, 1998). In civil law systems, the legal code consists of ideals and principles written as rules of law which are typically changed only by legislative action, in contrast with the common law system in which laws and legal procedure are progressively constructed in decisions made by judges and juries in successive cases. For this reason, Brazilian environmental law tends to be idealistic and ambitious, but may be difficult to implement and enforce because the theoretical truths represented in the law may not be compatible with social, economic, and political realities among resource users. Furthermore, because laws are often perceived by their creators and supporters to represent moral ideals, the concept of providing incentives to groups or individuals to facilitate or encourage compliance may be seen as antithetical. Finally, the design of laws as principles in the abstract stands in direct opposition to the concept of negotiated

solutions which are often necessary in complex natural resource problems and are more characteristic of the common law system.

The “command-and-control” nature of legislation magnifies the importance of monitoring and enforcement, especially coordination among agencies responsible for carrying out monitoring and enforcement, clear and consistent communication with resource-users regarding the rules, and efficient and appropriate monitoring systems. Unfortunately, environmental regulation in Brazil has been marked by poor coordination among responsible authorities (Mello, 2003), lack of human and financial resources (Pasquis, 2004; Lustosa et al., 2003), and a pervasive perception of environmental policy as generally obtrusive to economic development and, thus, of significantly lower priority than many other policies (Ascher, 1999; Mello, 2003). Many key policy decisions that affect natural resources are still primarily the responsibility of other ministries, such as the Ministry of Agriculture, the Ministry of Transportation, the Ministry of Agrarian Development, and the Ministry of National Integration and Planning, which often leads to direct inconsistencies with environmental legislation. Moreover, the responsibility for developing and implementing specific environmental regulations is divided among federal, state, and municipal authorities, often without sufficient clarity about which agency is responsible for which issues and with inadequate communication and coordination between these authorities (Ascher, 1999; May, 1999). Even where responsibility is clearer, lack of personnel or funding for enforcement can hamper policy implementation. Ascher (1999) also observes that government policies frequently are not designed to encourage landholders or land-users to invest in good land

management, or instead provide incentives to invest in inappropriate technologies or methods.

Effective governance also requires that the rules of resource use are generally followed, with reasonable standards for tolerating modest violations. As already pointed out, Brazilian environmental legislation itself typically provides few incentives for resource users to comply and resources for enforcement are often lacking. Compounding these shortcomings, violations have tended to be overlooked, in large part because the judicial system is ill-equipped to prosecute violators (Ames and Keck, 1998). In 1998, the federal government signed into law a new Environmental Crimes Law which greatly broadens liability for environmental violators. The new law improves the ability of administrative agencies to apply administrative sanctions, establishes the responsibilities of corporations for environmental violations and damage, turns more environmental violations, such as illegal logging, into crimes with higher penalties (up to USD 16 million), and provides quicker judicial procedures for many environmental crimes (Brito and Barreto, 2006). However, Brito and Barreto (2006) conclude that the main obstacles to the enforcement of this law are difficulties in locating violators and the lack of effective communication among the agencies responsible for applying the law, which results in delayed prosecutions.

The Brazilian Forest Code

Current Requirements

The Brazilian Forest Code regulates the use and conservation of forests and other native vegetation types on rural properties through three principal mechanisms: (1) Permanent Preservation Areas (*APP—Áreas de Preservação Permanente*), (2) the Legal Reserve (*RL—Reserva Legal*), and (3) hillslope forests APPs are designed to

protect the most ecologically sensitive areas, and thus protect (1) the riparian vegetation around streams, rivers, lakes and artificial reservoirs, (2) slopes with greater than 45° inclination, and (3) hilltops. Vegetation in the APP zone may not be harvested or cleared, except in exceptional situations defined by law (for public utility or social interest). Where riparian vegetation has been cleared in excess of the limits defined within the Forest Code, the APP vegetation must be restored. The size of the APP zone within individual properties depends on the frequency and width of streams, springs, and on the topography of the property.

The legal reserve constitutes a percentage of the area of the property in which native vegetation must be maintained; the percentage is defined by the biome and region in which the property is located. In contrast with APPs, the RL may be sustainably harvested subject to an approved management plan that maintains the ecological function and composition of the native vegetation. Where native vegetation has already been cleared in excess of the limits stipulated by the Forest Code, property owners are required to restore the RL. The current Forest Code (modified in 1996 and officially adopted in 2001) determines that the legal reserve should constitute 80% of the area of the property for properties located in the forest biome of the Legal Amazon[2], 35% in the *cerrado* biome of the Legal Amazon, and 20% in other regions of the country.

Finally, the Forest Code also limits the use of forest on slopes between 25° and 45° inclination. It stipulates that only sustainable forestry is allowed on these slopes,

[2] The Legal Amazon extends beyond the natural boundaries of the Amazon River basin to include the entire administrative jurisdiction of the states of Mato Grosso, Tocantins, Maranhão, Pará, and Amapá as well as those states that fall completely within the basin (Amazonas, Roraima, Acre, and Rondônia).

and that these areas *may not* be counted as part of the Legal Reserve, nor do they constitute part of the APPs. In theory, the legal reserve, the APP, and the hillslope forests are to be maintained separately and independently. For landowners having riparian and/or hillslope forests, the total area of land that must be set aside in restricted or sustainably managed reserves may sum to more than 80% of the property in the Amazon forest biome.

History

As noted previously, the fundamental premise of the Forest Code is that it defines forests and the services that flow from them as a common good and recognizes that private property ownership constitutes a bundle of both rights and responsibilities. The balance between rights and responsibilities has been the focus of intense and frequent debate between the farming and ranching lobby and environmental and social organizations in Brazil for at least 75 years. Although Brazil already had some regulations requiring the protection of forest resources on private lands prior to the 1930's, these were not formally codified until 1934. The Forest Code of 1934 (*Decreto-lei no 23.793*, January 23, 1934) for the first time made protection of forest resources a responsibility of land-holders; previously forests were seen as the object of utilitarian rights accruing to property-holders. The first Forest Code required that property-owners maintain "protection forests" serving a similar function as the permanent protection areas (APPs) of the modern Forest Code: to conserve hydrological functions, prevent soil erosion, support frontier defense, guarantee public health, protect sites of natural beauty, and provide protection for rare native species of flora and fauna. Cutting of trees was strictly prohibited in these forests (Zakia, 2005).

In 1946, the Brazilian Constitution was amended to allow expropriation of private lands for agrarian reform if properties were deemed not to fulfill their full social function. Although not formally enshrined in Brazilian law until 1946, the social function doctrine has been an important factor in property claims from the beginning of Portugal's colonization of its South American territory (Alston et al., 1999; Colby, 2003; Ankersen and Ruppert, 2006). Until the *Lei da Terra* (Land Law) was enacted in 1850, individuals could legally declare *posse* (*de jure*, usufruct rights; *de facto*, possession) simply by clearing land and establishing pasture or agriculture (Benatti, 2003; Ankersen and Ruppert, 2006). The justification for this claim was that the actions of the claimant helped the land to fulfill its social function. However, the fear of expropriation on this basis also put the social function doctrine in direct conflict with the 1934 Forest Code, creating a disincentive for compliance with the Forest Code.

To address this conflict, the Land Statute of 1964 (*Lei 4.504/64*) accorded an "environmental function" to private properties for the first time, expanding the notion of productive lands to include the conservation of natural resources. One year later, the modern Forest Code (*Código Florestal de 1965*, *Lei no 4.771*, September 15, 1965) was written into law. It introduced the concepts of the legal reserve (where sustainable timber harvests were permitted) in contrast with the permanent protection area (where strict protection is required, adapted from the "protection forests" of the 1934 Forest Code) and codified the notion that forests are a common good and as such impose limits on the rights of the property owner. Furthermore, the new Forest Code created restrictions on the use of private property that would be enforced through fines and other penalties. The 1965 Forest Code determined that the "Legal Reserve" (RL) should

constitute 50% of the area of the property for properties located in the Legal Amazon and 20% in other regions of the country. In 1989, the Forest Code was amended to reduce the RL to 20% in the cerrado and non-“primary” forest biomes in the Legal Amazon, to accommodate the agricultural expansion into the Center-West region of Brazil, particularly Mato Grosso (Azevedo, 2009).

In 1995, deforestation in the Amazon reached a record high of over 29,000 km^2 (INPE, 2009). Under international and domestic pressure, the government undertook several policy changes. First, it declared a two-year moratorium on the harvest of mahogany. Secondly, and more importantly, at the end of 1996, it adopted the provisionary measure (*medida provisoria*) MP 1.511, increasing the RL on rural properties in the Amazon forest biome from 50% to 80% and prohibiting new clearing on properties already possessing “abandoned or under-utilized” areas or areas “used inappropriately with respect to the capacity of the soil.” The MP 1.511 provoked an intense reaction from the agro-industrial lobby; over the next 5 years, the measure was modified repeatedly as the agricultural and environmental lobbies battled back and forth, leaving landholders in doubt regarding their legal obligations.

After repeated substantial changes, in 2000, the federal government re-edited the provisionary measure in favor of the National Environmental Council’s (CONAMA) recommendation, setting out the following requirements: (1) an RL of 80% for properties in the Amazon forest biome, 35% for those in the cerrado biome within the Legal Amazon, and 20% in all other areas of the country; (2) the area of APP could only be counted toward the RL in the 80% zones; (3) subject to specific conditions set out by state zoning plans (*Zoneamento Socio-Economico Ecologico*), the RL could be reduced

to 50% from 80% in the Amazon forest biome; (4) the tradable legal reserve rights scheme required that an RL be “compensated” in another area of equal extent and ecological function and character within the same micro-basin (Chomitz, 2004); (5) special conditions for small properties (less than 100 ha) allowed only smallholders to restore their RL using exotic species, allowed smallholders to subtract the area of APP from the RL when the APP exceeded 5% of the area of the parcel, and allowed smallholders to carry out “sustainable agroforestry management” activities in the APP. CONAMA’s text was maintained through MP 2.166 in 2001, at which point the National Congress passed a constitutional amendment limiting the power of the executive branch of the federal government to edit provisionary measures. Provisionary measures that were active at that time became law.

Relevance of Other Policies

At the same time that the Forest Code was undergoing almost constant modification, environmental regulation was moving towards greater decentralization as part of an overall shift in governance structure in Brazil in response to a mandate set out by the 1988 Constitution (Azevedo, 2009; Weiss, 2000). In 1997, CONAMA[3] approved a resolution requiring rural environmental licensing to be developed and implemented by states as a means of ensuring that the Forest Code would be carried out; this was part of an overall legal decision that states should elaborate their own forest policies and regulate rural environmental licensing (related to ranching and agricultural production, especially deforestation, agrochemicals, and soil management). Although

[3] Under current Brazilian law, CONAMA must approve any environmental legislation or policies (in addition to approval by the executive and/or legislative branches) before these policies can be officially enacted and implemented.

the Federal Pact (*Pacto Federativo*), which turned greater decision-making and regulatory power over to states, was not signed until the beginning of 2000, Mato Grosso's state environmental agency—the *Fundação Estadual do Meio-Ambiente* (FEMA)—had already gained control over implementation of the Forest Code (among other environmental legislation) from IBAMA (the arm of the federal Environmental Ministry responsible for implementing and enforcing all environmental legislation prior to decentralization) by 1999 (Azevedo, 2009).

Simultaneously, FEMA began developing an environmental licensing system for private rural properties (*Sistema de Licenciamento Ambiental para Propriedades Rurais*—SLAPR) as a means of monitoring compliance with the Forest Code, and particularly, of differentiating between legal and illegal deforestation. Although INPE had been monitoring deforestation in the Amazon (PRODES) since 1988, the system was unable to discriminate legal and illegal clearing. The system was implemented in 2000 and 2001. Now in control of implementation and enforcement of the Forest Code, Mato Grosso determined that it would require landholders within the Amazon forest biome in the state to observe a legal reserve of only 50% rather than the 80% required by the federal Forest Code. FEMA cited Mato Grosso's state zoning plan as justification for this interpretation, despite the fact that the plan had not been legally approved by the state assembly yet (Azevedo, 2009). This provision was maintained until 2005, when the federal government's Operação Curupira extinguished FEMA and the federal government again exerted more control over environmental regulation in the state, including requiring establishment of a new state environmental agency, the State Environmental Secretariat (SEMA—*Secretaria Estadual de Meio-Ambiente*).

Additionally, landholders faced uncertainty regarding the definition of biomes and their boundaries, and thus, the proportion of a given property to be set aside, since the legal reserve requirement is biome-specific. The cerrado and forest biomes are frequently redefined in the Legal Amazon. That is, since the Forest Code specifies the percent of a rural property that must be maintained in a legal reserve of the native vegetation, and this percentage is not the same for all biomes (i.e., 35% for cerrado, 80% for forest), landowners, lobbyists, and politicians often pursue legal redefinition of a vegetation type in regions where the distinction between cerrado and so-called "transition" forests is not easily recognized and well-established. In general, the percent of vegetation required to be maintained is established in the political arena, with little (if any) input from scientific assessment. Many properties located in what is considered to be "transition" forest were registered with a legal reserve amounting to 20% of the property before 1996 since this vegetation type was not considered to be forest and was subjected the Forest Code amended in 1989. In some cases, properties may be located in more than one biome, creating additional confusion regarding the percent and location of the legal reserve on the property that must be maintained.

Application of the Forest Code on private rural properties is subject to conditions set out in the Agricultural Law of 1991 (*Lei no 8.171*, 1991). It established a term of 30 years for the property owner to restore the RL—where it does not meet the minimum percentage—on his land. Although its goal is to protect the landholder by assuring a generous period of time to comply with his legal obligations under the Forest Code, it also states, unequivocally, that the rural private properties must comply with the APP and RL requirements. The Agricultural Law also exempts landholders from paying the

rural property tax (ITR—*Imposto sobre a Propriedade Territorial Rural*) on the APP and RL areas of their land. Essentially, the Law thus assigns an economic value to the environmental function provided by these protected areas and is designed to serve as an incentive to reinforce the requirements of the Forest Code.

Materials and Methods

Study Area

The 177,780 km^2 Xingu River headwaters region is located in the northeastern corner of Mato Grosso state, in central Brazil (Figure 2-1). The region's soils, topography (100-300 m altitude, with flat interfluvial expanses) and climate are well-suited for soybean production and cattle ranching. Native vegetation types in the region are comprised of forests (tall evergreen, transitional semi-deciduous, and riparian) and savannas (*cerrado* woodland, mosaics of grassland, thickets, gallery forests). Ten indigenous territories are completely contained within the boundaries of the Xingu watershed in Mato Grosso (Figure 2-1). Indigenous territories cover approximately 42,200 km^2 within the basin, representing 24% of the total area of the headwaters region. Private landholdings comprise nearly 70% of the total area, and smallholder settlements[4] comprise less than 5% of the region. The streams and rivers of the major protected forest area that lies at the center of the region—the PIX complex, which alone comprises nearly 20% of the headwaters area—are under growing threats from sedimentation, agrochemical run-off, and associated fish die-off from the unprotected headwaters regions outside of the park boundaries (Sanches, 2002). The Xingu region

[4] Here, smallholder settlements are defined as government-sponsored settlement projects (*assentamentos*) consisting of contiguous, individual 100-ha lots which are distributed to individuals or families who meet a number of criteria, including not owning any other real estate nor having the means or prospects to own any real estate.

is representative of many areas along the Amazon's agricultural frontier, but faces a more acute and immediate threat because it lies between two major federal highways (BR-158, BR-163) that are partially paved, and lies in the pathway of the northward expansion of Brazil's grain belt.

Land Cover Maps

I developed maps of observed landscapes for 3 dates (1996, 1999, 2005) and 2 theoretical landscapes reflecting land-cover under pre-1996 and post-1996 Forest Code requirements. I used these maps to assess compliance with the requirements of each of these 2 versions of the Forest Code: (1) pre-1996, requiring 50% legal reserve in the forest biome (hereafter, referred to as "pre-1996"), and (2) post-1996, requiring 80% legal reserve in the forest biome (hereafter, referred to as "post-1996"). I also used these maps as a basis for assessing rate and proportion of legal and illegal deforestation following the change in the Forest Code. Furthermore, I calculated both the area available for conversion and in need of restoration relative to the requirements of each version of the Forest Code. Finally, I also combined them with maps of net present value for the region to estimate the costs of compliance.

Observed landscapes

I developed maps of land-cover for 3 dates of significance for the change in legislation to assess the extent to which the change in the legal reserve slowed deforestation in the region: 1996, 1999, and 2005. 1996 is the year in which the legal reserve requirement in the Amazon Forest biome was raised from 50% to 80% of private property through a provisional measure. Mato Grosso's state environmental agency took over responsibility for monitoring and enforcement at the beginning of 2000. Since only dry season imagery from this region is cloud-free, it was more

conservative to use images from July/August 1999 than from the same period in 2000. 2005 is the year in which the federal government dissolved the Mato Grosso state environmental “foundation” (subsequently reorganized as the state environmental secretariat) and mandated that landholders in Mato Grosso follow the requirements of the Forest Code to the letter of the law, rather than as interpreted by the state government since 2000. Maps with 4 classes (forest, cerrado, agricultural lands, other) were classified as described in Appendix A.

Theoretical landscapes

Two theoretical landscapes corresponding to the requirements set forth by the Forest Code before 1996 and after 1996 were developed using a spatial landscape simulation model. The basic architecture and function of the simulation model is described in Appendix B; details regarding the modeled landscapes follow here. The assumptions underlying each of the two scenarios differed only in the percent of native vegetation that was to be maintained or restored on each private land-holding in the headwaters region. Since a complete map of property boundaries for the region is not available, I used micro-basins representing individual stream reaches (1:1,000,000 scale) as proxies for individual properties. The mean, range, and distribution of sizes of the 2881 microbasin ($\bar{x}$ = 5981 ha, 4-70,766 ha) is comparable to that of private properties in the region (Jepson, 2006; Fearnside, 2005; Appendix B). Furthermore, the mean percent clearing in the current (2005) landscape is comparable among microbasins and properties for which property limit data are available, indicating that microbasins are suitable substitutes for individual properties in terms of sampling the population of properties in the region. It was necessary to use a spatial unit with

complete coverage of the study region to simulate the distribution of vegetation across the landscape corresponding with the requirements of the pre- and post-1996 versions of the Forest Code.

For both scenarios, full compliance with the law was assumed. All indigenous reserves and state and federal protected areas were strictly protected. Furthermore, a 50-m riparian buffer zone surrounding each stream and river visible in a map derived from a thematic stream layer obtained from the Mato Grosso State Regional Planning Secretariat (SEPLAN-MT) was strictly protected. The Forest Code stipulates that a riparian buffer zone of at least 50 m be protected around every natural body of water, and that the size of the buffer zone (up to 500 m) is dependent on the width of the stream. However, as stream width is difficult to assess and no official map of riparian buffers exist, the estimation of variable-width riparian buffers was not possible and I assumed, conservatively, that all water bodies were surrounded by a 50-m wide buffer. Where necessary, vegetation was restored to this riparian buffer zone so that each theoretical landscape had 100% native vegetation cover within the boundaries of the riparian zone. The original 1965 Forest Code (pre-1996) required that the legal reserve be maintained in addition to the riparian zone (i.e., the riparian zone could not be counted toward the percent of vegetation to be maintained in the legal reserve). Various versions of the post-1996 provisionary measure alternately had the same requirement or allowed the riparian zone to count towards the legal reserve. The final version of the provisionary measure that was adopted in the federal legal code required that each be maintained independently and completely, although subsequent interpretation at federal and state levels varied. To facilitate analysis of compliance with the legal reserve

requirement separately from the riparian zone requirement under the two iterations of the Forest Code, I did not allow the riparian zone area to count as part of the amount of legal reserve required in each microbasin in either the pre- or post-1996 scenario.

To calculate the amount of deforestation or restoration that could or should take place, respectively, outside the riparian zone to meet the legal reserve requirements under each of the two scenarios, I classified each micro-basin according to biome (cerrado or forest). The cerrado-forest biome map was obtained by merging a map of forest-non-forest derived from INPE Prodes maps with a map of biomes from the IBGE RADAM vegetation thematic map. Of 2881 micro-watersheds in the Xingu River headwaters region, 34 straddled both biomes. To facilitate model design and processing, each of these microbasins was assigned to the biome representing more than 50% of that micro-watershed's area. At time step 0, the model calculates how much of each watershed's area consists of cleared area. If this area is greater than the allowed amount (as determined by each version of the Forest Code) after the area of the riparian zone is subtracted from the total watershed area, the model reforests the area up to the allowable amount of cleared area. If the cleared area is less than the allowable area, the model begins to deforest the watershed based on where the highest favorability for deforestation is indicated (Appendix A). When a microbasin included protected areas, the microbasin was sub-divided so that the part outside of the protected area was treated as an independent microbasin, subject to the rules described above. The part in the protected area was prohibited from being cleared and did not count among those microbasins serving as proxies for private properties.

Compliance

To assess degree of compliance with the Forest Code for each of the three "observed" dates, I calculated the area and quantified the percent of remaining native vegetation by biome for the entire headwaters region outside protected areas and compared this to the amount required under each the pre- and post-1996 versions of the Forest Code. I also calculated the area and percent of remaining native vegetation for each microbasin located outside protected areas. I present the mean and standard error for the population of microbasins in each biome at each date. For each year, I assessed the absolute amount and the percent of vegetation remaining in the 50-m riparian buffer zone at both the landscape and microbasin levels. I also present the number of microbasins complying with both the legal reserve and the riparian zone requirement under each version of the Forest Code. Finally, I assessed the transitions of forest microbasins among three categories of compliance: (a) compliance with neither legal reserve regulation (below 50% forest cover), (b) compliance with the 50% legal reserve requirement (between 50 and 80% forest cover), and (c) compliance with both regulations (greater than or equal to 80% forest cover). Since the microbasin boundaries do not coincide with those of actual properties in the region, the results must be interpreted with care. Because of the similarity in the range, mean size, size distribution, and percent clearing of the microbasins compared with the group of properties for which data are available, it is possible to assume that the results presented here represent the overall trends likely to be seen in a survey using property boundaries. Nevertheless, some differences are likely to be present and the results presented here should be seen as describing general trends rather than specific results about the location and amount of actual compliance.

To assess the extent to which the change in the Forest Code slowed illegal deforestation in the forest biome, I estimated the amount and rate of deforestation outside protected areas in the Xingu River headwaters region for two periods: 1996-1999 and 1999-2005. I identified each pixel that underwent deforestation during each of the two time periods. Pixels were then further classified by whether they were considered to be in or out of compliance with each of the legal reserve requirements of each of the two iterations of the Forest Code. The final classification allowed pixels to fall into one of 3 categories in each of the two periods, as follows:

- **Deforestation, legal**: forest clearing in microbasins that were in compliance with the regulations at the beginning of the period and remained in compliance at the end of the period;
- **Deforestation, illegal, new non-compliance**: forest clearing in microbasins that were in compliance with the regulations at the beginning of the period and became non-compliant by the end of the period as a result of clearing;
- **Deforestation, illegal, continued non-compliance**: forest clearing in microbasins that were out of compliance with the regulations at the beginning of the period and remained out of compliance at the end of the period.

Economic Aspects of Compliance

To evaluate the economic aspects related to compliance with the new Forest Code requirements in the forest biome for each of the three dates, I calculated the difference in the area of the watershed outside of federal and state protected areas (1) that would no longer be available for conversion to agricultural land, and (2) that would need to be restored to come into compliance under the new regulations (80% legal reserve in forest biome, 35% in cerrado) in comparison with the old regulations (50% in forest, 35% in cerrado). For each, I present the total area for the entire headwaters region as well as the mean per microbasin. Based on these figures, I estimated the economic costs of

complying with the policy change by estimating (1) the potential forgone profits to producers over the whole landscape and by microbasin using net present value as a proxy for the potential value of agricultural lands in the region, and (2) the cost of riparian forest restoration over the whole landscape and by microbasin, where necessary (methods are described in detail in Appendix C).

I estimated total maximum potential NPV for both forested and cleared lands—*if they were to be cleared or remain cleared*—within the forest biome as well as per microbasin. Furthermore, I estimated potential NPV for microbasins in and out of compliance with the pre- and post-1996 Forest Code requirements to estimate the potential for legal agricultural expansion under the 50 and 80% legal reserve requirement. In each case, I also calculated and compared the mean NPV ha^{-1} to test the hypothesis that mean potential earnings were associated with non-compliance. Restoration costs were estimated for riparian zone and legal reserve areas, at the aggregate and microbasin levels.

Ecological Consequences

I compared the final landscapes for each of the 5 alternative landscapes in terms of carbon stocks, river discharge, annual evapotranspiration, terrestrial habitat quality, and water quality (methods are described in Appendix C). Unlike all the preceding analyses, ecological consequences were assessed for the entire headwaters region, including all protected areas.

Results

Compliance

Legal reserve

At the scale of the entire Xingu River headwaters region, forest cover was greater than 80% (the new legal reserve requirement) in 1996, but fell below 80% in 1999 and further still in 2005. The region was in compliance with the 35% cerrado limit in all three periods (Table 2-1). In 1996, when the provisional measure was first adopted, 83% (79,910 km^2) of original forest cover and 64% (18,617 km^2) of original cerrado cover still remained outside protected areas across the Xingu headwaters region (Table 2-1, Figure 2-1), thus meeting both the old (50% legal forest reserve) and new (80% legal forest reserve) regulations. By the middle of 1999, 6 months before the policy was permanently adopted, original forest cover had fallen to 78% (75,139 km^2), and original cerrado cover had fallen to 54% (15,967 km^2). During the subsequent 6 years, to 2005, further deforestation reduced forest cover to 64% (61,934 km^2) and cerrado cover declined to 45% (13,065 km^2) (Table 2-1, Figure 2-1). Total forest loss on lands outside of protected areas from 1996 to 2005 equaled 17,976 km^2. The annual average area of forest loss increased from 1590 km^2 between 1996 and 1999 to 2201 km^2 between 1999 and 2005. In the cerrado biome, total native vegetation loss reached 5552 km^2 over the 10-year time period; mean annual clearing decreased by nearly half in the second period, from 883 km^2 to 484 km^2.

These aggregate statistics mask the degree of compliance by individual landholdings (represented here by microbasins, which I employ as proxies) with the regulations. Landholdings are the unit for which enforcement of the legislation takes place. In 1996, when the legislation changed, 8% of forest microbasins had cleared

more than 50% of forest vegetation and were therefore not in compliance with the old regulations (Table 2-2). However, with the increase in the legal reserve requirement from 50 to 80% of the property, an additional 20% of forest microbasins fell out of compliance, representing a sudden increase in illegality of 336%. The combined area of cleared land in excess of the new limit (20% clearing permitted) in these microbasins was 2404 km^2, 15% of all existing cleared area that year. Eight percent of microbasins located in the cerrado biome were also out of compliance with the 35% native vegetation cover legal reserve requirement which did not change (Table 2-2). In 1999, 36% of forest microbasins failed to meet the new requirements, representing an increase in illegality of 297% relative to compliance with the old requirements (Table 2-2, Figure 2-2). Eighteen percent of cerrado watersheds did not meet the legal requirements. By 2005, following the post-1995 spike in deforestation in the region, from 2002 to 2004, 54% of forest microbasins did not comply with the new requirements, whereas 21% would have been out of compliance under the old regulations. In contrast, more than three-quarters of cerrado microbasins met the requirements.

Changes in microbasin-level compliance take place through different pathways. I examined the transitions of forest microbasins among three categories of compliance: (a) compliance with neither legal reserve regulation (below 50% forest cover), (b) compliance with the 50% legal reserve requirement (between 50 and 80% forest cover), and (c) compliance with both regulations (greater than or equal to 80% forest cover) (Figures 2-2, 2-3). In 1996, the percentage of forest microbasins in each of these categories was 8, 20, and 72%, respectively (Figures 2-2, 2-3, Table 2-2). In 1999, the new allocation of microbasins among the three categories of compliance was 12, 24,

and 64%, respectively (Figures 2-2, 2-3, Table 2-2). The largest shift between categories (8%) occurred from compliance with both regulations (>=80% forest cover) to compliance with the old regulations (between 50 and 80% forest cover). Of the microbasins with less than 50% forest cover in 1996, 98% remained non-compliant in 1999, with 86% continuing to deforest (Figure2- 4). From 1999 to 2005, the new allocation of microbasins among the three categories of compliance was 21, 33, and 46%, respectively (Figure 2-2). Sixteen percent of forest microbasins shifted from compliance with both regulations to compliance only with the old regulation. Eight percent of forested watersheds were out of compliance with the old regulations in 1996 and continued to deforest through 2005 (Figure 2-4). These illegal microbasins are primarily concentrated on the eastern side of the Xingu Basin, where the profitability for soy cultivation was highest. Forty-six percent of the microbasins still retained 80% or more of their forest cover in 2005 (Figure 2-2), and are concentrated on the western side of the basin (Figure 2-3, 2-4), where selective logging constitutes an important part of the regional economy; this indicates, however, that remaining forests are likely to be degraded (Asner et al., 2005).

The microbasin-level analysis also permits quantification of the area of forest clearing that was in compliance with the old and new legal reserve requirements. Between 1996 and 1999, a total of 4867 km^2 (5% of the original forest area on private lands) were deforested by 68% of forest microbasins (Table 2-3). Fifty-seven percent of total microbasins (representing 39% of total deforestation in the period) deforested without falling below the 50% minimum forest cover requirement. As a result of the increase in the legal reserve requirement, a greater proportion of 1996-1999

deforestation (31%) was carried out on 430 (24%) microbasins already out of compliance with the law's requirements (Table 2-3). In the second period, from 1999-2005 (after the policy change was officially adopted as law), a total of 13,184 km^2 of forest were deforested by 86% of microbasins. Fifty-eight percent of this deforestation (8386 km^2) was in compliance with the old regulation and only 15% (2006 km^2) with the new regulation (Table 2-3).

Riparian buffer zone

A separate provision of the Forest Code mandates that the riparian zone be strictly protected and that vegetation be restored if any clearing has taken place in the buffer zone. In 1996, a total of 12% of the riparian zone in the headwaters region, outside of protected areas, had been cleared, with a greater proportion of cerrado riparian area cleared (18%, Table 2-1). Only 12% of cerrado microbasins were in compliance, one third the proportion of that encountered in the forest biome (Table 2-2). In 1999, the area of riparian zone cleared climbed to 16% (Table 2-1). Only 32% of microbasins were in compliance with the law at this time (Table 2-2). By 2005, 23% of the riparian vegetation had been cleared, and compliance fell to 17% of microbasins (Tables 2-1, 2-2). This decrease in riparian zone vegetation occurred mostly in forest microbasins, where compliance fell to 20% (Table 2-2).

Economic Aspects of Compliance

Opportunity cost

The increase in the legal forest reserve requirement from 50 to 80% imposed costs on landholders. The first cost that I consider is that associated with foregone profits from soy or cattle ranching incurred through compliance with the mandatory forest legal reserve requirements, which depends on the area of forest land landholders

must leave standing. At the level of the entire Xingu River headwaters region, in 1996, nearly 33,000 km^2 of forest could still be cleared by 1608 (92%) forest microbasins under the old regulations, whereas less than 10,000 km^2 could still be cleared by 1259 (72%) microbasins under the new regulations in 1996 (Tables 2-2, 2-4). The average landholding still in compliance with the old regulations would have been permitted to clear a further 20 km^2, whereas under the new regulations, only 7 km^2 of further clearing would have been allowed (Table 2-4). By 2005, 20,000 km^2 were potentially eligible for clearing under the old regulations, in contrast with less than 4000 km^2 under the new regulations. At the individual microbasin level, these numbers had fallen to less than 15 km^2 and 5 km^2, respectively (Table 2-4).

The total net present value (NPV, calculated over a 30-year time horizon) that could be achieved for lands outside of protected areas across the Xingu headwaters region, assuming that the maximum economic value were to be extracted from the landscape from soy or cattle ranching, reaches 32.9 billion USD (Table 2-5). The potential NPV of the entire Xingu River watershed area outside of protected areas associated with soy and cattle production differs by approximately 5 billion USD between a landscape reflecting the old (50%) and new (80%) regulations (difference in NPV of forest area in compliant microbasins, Table 2-5). This difference climbs to approximately USD 9 billion when the area of forest land “available” for conversion under the old vs. new regulation is compared for microbasins (Table 2-5). In 1996, producers in the aggregate still had the potential to realize an additional NPV of 11.6 billion USD under the old regulations, but only 2.9 billion USD under the new regulations (Table 2-5). By 1999, these values decreased to USD 9.6 vs. 0.9 billion, respectively,

then to USD 4.5 and -4.2 billion in 2005 (Table 2-5). Under the new regulations, land with a value of 4 billion had already been cleared illegally by 2005 (Table 2-5); this is income that producers out of compliance in the region would not have earned if they had complied with the new regulations.

At the level of the individual microbasin, the potential additional NPV that could be realized through legal clearing of land fell by an average of USD 4 million (Table 2-7). In other words, the change from a legal reserve of 50 to 80% represented a USD 4 million dollar decline in the potential NPV of the region's landholdings.

I tested the hypothesis that compliance with the legal reserve requirement is inversely proportional to the opportunity costs of compliance by comparing the mean NPV ha^{-1} of cleared land for compliant and non-compliant microbasins (Table 2-6). The hypothesis was supported for all comparisons of NPV ha^{-1}. The difference in NPV ha^{-1} between compliant and non-compliant microbasins became larger each successive year, and was greater for the 80% regulation than for the 50% regulation, where the per-hectare value in noncompliant basins remained 1.5 times greater than that of cleared lands in compliant microbasins (Table 2-5). Over time, this ratio increased, with the potential NPV ha^{-1} of cleared lands in non-compliant microbasins (3087 USD ha^{-1}) nearly double that of cleared lands in compliant microbasins (1772 USD ha^{-1}) in 2005 (Table 2-6).

Cost of restoration

As the forest area available for clearing declined through the change in the Forest Code, the aggregate area of legal reserve to be restored in 1996 rose from 1200 km^2 under the old regulations to 6500 km^2 under the new regulations (Table 2-4). For the average individual microbasin already in violation of the old requirements in 1996, the

legal reserve restoration requirement increased from 8 to 13 km^2, going from the old to the new regulations (Table 2-4). By 2005, the aggregate area to be restored increased to 3500 km^2 and 16,000 km^2, under the old and new regulations, respectively. By 2005, the average restoration requirement relative to the old and new requirements rose from 9 to 17 km^2. The total riparian area needing to be restored increased from approximately 1300 km^2 in 1996 to 2500 km^2 in 2005. For the average landholder, this meant an increase from 0.6 km^2 to over 1 km^2 of riparian forest restoration (Table 2-4).

In 1996, restoration costs to bring all microbasins into compliance with the old regulations (including riparian zone restoration) was 375 (±232) million USD (Table 2-8). To come into compliance with the new regulations, the cost more than doubled to 868 (±442) million USD. These costs increased in 1999 to 534 and 1226 million USD, respectively; by 2005, the costs climbed to 841 and 2018 million USD (Table 2-8). Whereas riparian restoration costs represented an average of 69% (under the old regulations) and 31% (under the new regulations) of the total restoration costs in 1996, by 1999 they had risen to 61% (old) and 27% (new) of the cost (Table 2-8). By 2005, these proportions had fallen to 51% (old) and 20% (new) of the cost. For the average individual landholder needing to come into compliance, the cost rose from 1.2 to 1.6 million USD in moving from the old to new regulations in 1996 (Table 2-8). This cost rose in 1999—to 1.4 and 2 million USD—largely because of a doubling in the area of riparian zone restoration. The overall average cost to landholders increased further in 2005 (1.6 million USD to comply with the old regulations, 2.3 million USD to comply with the new regulations) due to an increase in the amount of legal reserve restoration, despite the cost of riparian zone restoration remaining stable. At the level of the

individual landholder, riparian zone restoration costs represented a small proportion of the total restoration costs than for the landscape as a whole, never reaching more than 15% of the total costs (assuming a minimum of active restoration was carried out) (Table 2-8).

Ecological Consequences

Carbon stocks

Carbon stocks of the Xingu River headwaters region forests varied by 212 million tons from the highest level, in 1996 (661 MtC), to the lowest level, under the modeled 50% legal reserve regulation (449 MtC, Table 2-9). Carbon stocks between the modeled 50% and 80% landscapes differ by 27% (120 MtC, Table 2-9). Whereas implementing the new regulations would have meant a loss of only 50 MtC since 1996, retaining the old regulations would potentially have resulted in emissions of 150 MtC (640 $MtCO_2e$). Currently (2005), somewhat less than half that amount has been lost since 1996 (Table 2-9).

Hydrology and regional climate

Deforestation is known to reduce evapotranspiration and increase stream flow because of the reduced leaf area index, decreased root density and depth, and increased soil compaction (Bruijnzeel, 1990; Costa, 2005; Sahin and Hall, 1996; Scanlon et al., 2007; Nepstad et al., 1994). In the Xingu River headwaters, modeled streamflow was higher than the original landscape in all five scenarios. Among the 1996, 1999 and “80%” landscapes, mean annual stream discharge was similar ranging from 7 to 10% greater than that of the control landscape. In the 2005 and “50%” landscapes, stream discharge was higher, with 13 and 17% increases, respectively.

This implies lower potential for flooding, and lower risk of overland flow and associated soil erosion under the 1996, 1999, and “80%” landscapes.

Mean annual evapotranspiration decreases as forest cover decreases on the landscape, ranging from a 3 to 7% reduction from the control scenario. Evapotranspiration decreases more than twice as much in the theoretical 50% landscape over the 1996 and 1999 landscapes and nearly twice as much as the theoretical 80% landscape. More water vapor, equal to 60 mm y^{-1}, is therefore provided to the atmosphere under the high forest landscapes (1996, 1999, 80%) than under the low forest landscape.

Water quality

Whereas both the theoretical 50% and 80% landscapes had all riparian forests protected or restored , riparian forest cover was substantially lower (12 to 19%) in the historical landscapes. This suggests that streams in the historical landscapes are more likely to be affected by sedimentation from point-source than those in the theoretical landscapes.

Furthermore, by 2005, water temperatures are likely to be higher and dissolved oxygen levels lower in one-fifth of the landscape’s stream network (Neill et al., 2006), affecting species populations and assemblages. Some of these streams may also be subject to grass invasion (Neill et al., 2006). Riparian zone forests are also important in the food chains of aquatic communities (Neill et al., 2006; Burcham, 1988; Bojsen and Barriga, 2002; Lorion and Kennedy 2009).

Habitat

Overall, habitat quality and quantity are lowest in the theoretical 50% landscape, and highest in the 1996 landscape. The 1996 and 1999 landscapes are similar to one

another, having the highest amount of total forest and *cerrado* cover, the greatest mean fragment size (more than four times as large as the 50% landscape), as well as the highest amounts of interior (core) habitat area (more than twice as much as the 50% landscape) and lowest amounts of edge habitat (nearly 3 times less than the 50% landscape). Notably, the theoretical 80% landscape has substantially more edge habitat (1.5 times more) than the most fragmented historical landscape (2005), despite having fewer fragments and a greater amount of interior habitat than that landscape (Table 2-9). This is likely to be explained by the spatial stratification of legal reserves throughout the landscape in the theoretical landscape, in comparison to the current landscape where some areas of the landscape are not yet fragmented and other areas no longer have any native (or regenerated) vegetation remaining. The Forest Code, if imposed at the level of the microbasin, increases the amount of edge relative to the natural pattern of forest cover that arises through frontier expansion.

Discussion

The Brazilian Forest Code was designed to protect public interests in private-land forests, an intent that was fully canonized in the new Constitution of 1988. To successfully protect the “social function” of private land forests, the Code must change the behavior of landholders. They must comply with the restrictions on forest clearing that are defined by the code, and adjust this behavior when the Code changes. At the beginning of the study period, in 1996, most of the microbasins (92%) of the Xingu River headwaters region, outside of protected areas, were in compliance with the requirement that at least 50% of each property’s forests be maintained (Table 2-2). With the announcement during this same year of a temporary regulation to raise the legal forest reserve requirement to 80%, the number of microbasins in compliance with the new

requirement fell to 72%. This can be viewed as the beginning of the test of the effectiveness of the revised Forest Code. If the revised Code were viewed by landholders as carrying the full force of the law, then deforestation would have halted in microbasins that had already reached 20% forest clearing, continuing only on those microbasins with more than 80% forest cover remaining. In practice, however, the level of compliance with the 80% legal forest reserve requirement declined from 72% of microbasins in 1996 to 64% in 1999, then dropped precipitously to 46% in 2005, following the surge in clearing from 2002 to 2004 (Table 2-2). This illegal deforestation was manifested at the scale of the entire Xingu River headwaters region. Forest cover outside of protected areas declined from 83 to 64 % from 1996 to 2005 (Table 2-1). With the more restrictive Forest Code, annual deforestation in the region climbed from 1590 km^2 during the 1996-1999 period to 2201 km^2 during the 1999-2005 period, a further indication that the Code revision had little if any inhibitory influence on forest clearing by landholders.

It is not surprising that the level of compliance with the new legal reserve requirement was low. Compliance with environmental regulation is highest when (a) the process for achieving compliance is clear and practical, (b) the probability of non-compliant landholders being identified is high, (c) the probability of apprehended landholders paying fines or facing imprisonment is high, and (d) the costs of compliance are low. In sum, compliance is highest when non-compliance is very expensive. The change in the Code was not accompanied by an effective program for helping landholders bring their properties into compliance if they had already exceeded 20% clearing (28% of microbasins in 1996). It was only in 2000 that the requirement to bring

properties into compliance began to be enforced through creation of the Mato Grosso rural environmental licensing system, coincidentally with the sudden change in legislation. Properties could be brought into compliance through restoration of forests to bring the property's forest cover up to 80%, compensation of the legal reserve through the purchase of excess forests on other properties (Chomitz, 2004; Azevedo, 2009), and payment into a fund that would maintain or expand state protected areas were created (Azevedo, 2009). Even in 2005, only 3992 km^2 of forest could have been cleared legally and may have been set aside to compensate non-compliant properties (Table 2-3). However, the area of illegal clearing by this year had reached 16,000 km^2, more than four times the area available for compensation (3788 km^2). Hence, legal reserve compensation could have been used for, at most, one fourth of the illegally cleared land. Very few landholders brought their properties into compliance through compensation on other properties; between 1999 and 2007, only 5 such applications were processed by Mato Grosso's environmental agency (Azevedo, 2009). My data do not allow me to infer the degree to which landholders may have achieved compliance through payments into a state fund, although Azevedo (2009) indicates that it was also minimal.

Compliance with the new forest Code may also have been low because of landholder uncertainty about its longevity. I identify two major sources of this uncertainty. First, the state of Mato Grosso declared in 2000 that the legal reserve requirement for the "transition forests", which include approximately 76% of the forests in the Xingu River headwater region, was only 50%, despite the change to 80% for all forests in the Legal Amazon in the federal Forest Code. This state-level declaration

went unchallenged by the federal government for 5 years, and was accompanied by a vigorous debate about the definition of transition forest. As of 2005, the federal government over-ruled the state interpretation, and the entire forest biome returned to an 80% legal reserve requirement. Because these percentages are decided in the political arena, rather than through scientific assessment, some biomes (e.g., cerrado) and their ecological functions are undervalued relative to others. Conversely, the legal reserve for other biomes (e.g., forest) may be set higher than what is necessary to maintain ecological function and also result in low compliance as landholders attempt to gain parity with their peers in other biomes, ultimately to the detriment of the ecosystem.

A second source of uncertainty regarding the longevity of the 80% requirement was the frequent attacks on the forest Code within the Brazilian National Congress. The *bancada ruralista* (ruralist constituency, primarily the agricultural lobby) advanced proposals to reduce the legal reserve requirement of the Forest Code almost annually (Lima et al., 2005). The reduction of the legal reserve requirement was an important plank in the political platform of politicians, agro-industry's representative organizations, such as FAMATO (Mato Grosso Agriculture Foundation) and CNA (National Agriculture Confederation), and may have given landholders a sense of impunity regarding the Forest Code (Azevedo and Pasquis, 2006; Azevedo, 2009). Since 2005, Mato Grosso's agro-industrial sector continues to seek ways to reduce the legal reserve and/or to legalize properties that have cleared in excess of the permitted amount without requiring restoration (Camarga, 2008).

These weaknesses in implementation of the Forest Code may have been reinforced by the costs incurred by landholders through compliance with the Code. The

costs of registering with the state environmental licensing system (SLAPR) alone have been demonstrated to be prohibitive to many landholders, particularly if the landholder attempts to maintain a legal reserve of 80% (Guimaraes and Almeida, 2007; Azevedo, 2009). Opportunity cost may present an even greater obstacle to compliance. The forgone profits from deforestation-dependent economic activities, such as cattle ranching and soy cultivation, were particularly strong incentives for landholders to take the risk of getting caught and paying fines because of the high potential profitability of the Xingu region's soils for agriculture and ranching. Using spatially-explicit rent models for soy cultivation and cattle ranching, I estimate that the aggregate opportunity cost incurred by landholders through the increase in the legal reserve requirement was approximately USD 9 billion (Table 2-3). The mean opportunity cost for individual microbasins associated with the legal reserve requirement shift was USD 4 million (Table 2-7).

Further evidence of the contribution of opportunity costs to non-compliance is provided by the potential net present value of cleared land on a per-hectare basis for compliant and non-compliant properties. Mean NPV per hectare of cleared land was higher for non-compliant clearing than compliant clearing, and this difference was greater for the 80% restriction than for the 50% restriction, and it increased over time (Table 2-6). This suggests that landholders are more willing to break the law and run the risk of getting caught and punished if there is more profit to be made through illegal forest clearing. However, although it is unclear what proportion of violators were served with fines, of those who were, less than 2% of fines are estimated to have been

collected, even after Mato Grosso's environmental authority was reorganized (Azevedo, 2009).

In comparison to the forgone income from cleared lands, the costs to restore those lands are negligible, representing approximately 20% of the opportunity cost of the lands to be restored in 1996 and 1999 under both the new and the old regulations (Tables 2-5, 2-8). In 2005, restoration costs represented 13% and 18% of the opportunity cost of complying with the old and new regulations, respectively. This illustrates the point that the opportunity cost is likely to be far more prohibitive for landholders in bringing illegally cleared lands into compliance, but that adding restoration costs to opportunity cost increases the total cost by one-fifth, on average.

Although compliance with the change in the legal reserve requirement was low, simulation modeling allows us to understand what this policy decision would achieve if it were fully implemented. Under full compliance with the 80% forest cover, the level of "biotic regulation" (Bormann and Likens, 1979) of water flow through the Xingu River headwaters region increases. The higher level of evapotranspiration associated with the 80% legal reserve scenario relative to the 50% scenario (3%, Table 2-9), although a small difference, signifies a lower potential for surface runoff and associated soil erosion (Stickler *et al*. in press) and lower potential for stream and river flooding. Higher evapotranspiration also reduces the likelihood of deforestation-driven changes in the regional rainfall system, which some models suggest can take place when clearing exceeds 60 to 70% original forest cover (Sampaio et al., 2008; Werth and Avissar, 2002; Nobre et al., 1991). The Xingu headwaters region is likely to be severely impacted by rainfall reduction through climate change (Malhi et al., 2008) and the

maintenance of high levels of evapotranspiration could diminish the likelihood of these changes in rainfall (Nepstad et al., 2008). Both the 80% and 50% legal reserve scenario assume full restoration of riparian zone forests, potentially re-establishing the role of these forests in providing shade, fruits, and other forms of organic matter to the region's aquatic ecosystems. Riparian zone forests may prevent grass invasion in streams that can be associated with low levels of oxygen (Neill et al., 2006). Improvements in water quality under a fully implemented Forest Code could have direct positive impacts on the livelihoods of the indigenous peoples who reside in the Parque Indigena do Xingu (Xingu Indigenous Park) located at the core of the headwaters region (Figure 2-1). Tribes such as the Kisedje have observed declines in the quality of the fish and turtles that they catch for subsistence consumption and have also noted changes in the timing and strength of rains at the beginning of the rainy season (pers. comm., Chief Cuiuci Kisedje, 16 July, 2005). One of the biggest impacts of the 80% scenario fully implemented is in the increase in interior habitat of the region's forests (Table 2-9). Those species that depend upon forest interior habitats, such as ant birds (Bierregaard, 2001), would have more than twice the area of forest interior under the 80% scenario vs. the 50% scenario (Table 2-9).

The 80% legal reserve scenario also retains 120 million tons of carbon (450 million tons of CO_2 equivalent (tCO_2e), Table 2-9) more than the 50% legal reserve landscape. Carbon storage is the only ecosystem service in the Amazon region that is close to having a robust market mechanism to provide incentives for its maintenance. If the REDD (Reduced Emissions from Deforestation and Forest Degradation) regime under negotiation within the United Nations Framework Convention on Climate Change

(Gullison et al., 2007; Meridian Institute, 2009) results in the development of a market for carbon through reductions in deforestation, carbon could become a promising commodity for compensating land holders for the opportunity costs that they incurred when the legal reserve was changed from 50 to 80% of each property. This opportunity cost, that we estimate at approximately USD 9 billion (Table 2-5), could be fully compensated at a price of USD 20 tCO_2e^{-1} emission that is avoided; this price is within the range of current carbon prices (approximately USD 21 tCO_2e^{-1}; Point Carbon, 2009). In sum, the Xingu headwaters region provides insights into the limits of command-and-control regulations designed to defend public interests in private-land resources. The change of the legal reserve requirement from 50 to 80% of private properties imposed USD 9 billion in forgone potential present and future profits on the region's farmers and ranchers—averaging USD 3 to 4 million per microbasin—and was ineffective in creating processes and procedures through which landholders who wished to comply with the law could do so. The validity of the 80% legal reserve requirement was undermined by the state of Mato Grosso's decision that the transition forests of the region had a 50% legal reserve requirement, by the frequent attacks on the Forest Code within Brazilian legislature by the agricultural lobby, and by the low levels of enforcement of the 80% rule. Non-compliance was reinforced by the perceived risks associated with compliance, in which law enforcement officers punish those who are trying to comply with the law because of the threat that compliance poses to the culture of graft and corruption (Rosenthal, 2009).

The origins of these deficiencies in the Forest Code may be traced to the very system by which laws are created in Brazil and the extent to which this approach is

suited for handling complex natural resource governance issues. If the governance of common pool resources depends on designing a system of clear rules and incentives, the Forest Code fails on both accounts. Whereas the Forest Code certainly lays out a body of rules, the modifications made after 1996 combined with the process of decentralization to lead to a set of rules that are far from clear and that inspire uncertainty. Concurrently, powerful economic drivers created an even greater need for incentives to comply with the regulations. However, the moral idealism of the legal tradition in Brazil may have made the notion of incentives difficult to consider. With a focus on the "ideal" versus the "practical" solution to issues of the public good, and its defense, Brazil's civil law system typically produces new regulations and laws without the dialogue and debate among interested stakeholders that could build into the design of the new law mechanisms for increasing its practicality and chances of successful implementation. However, multi-stakeholder, participatory planning processes are already part of recent watershed management legislation (cite). More and more, civil society is initiating such processes for a variety of environmental governance issues, including infrastructure development (Campos and Nepstad, 2006; Soares-Filho et al., 2004), watershed management (ISA, 2005), and other regional planning processes (Perz et al., 2008), signaling an important trend in Brazilian environmental rule-making in the future.

Conclusion

This case study identifies a crucial challenge for lawmaking designed to defend public interests in privately controlled natural resources. At its core, it illustrates one legislative attempt to reconcile a trade-off that has repeated itself throughout human history. Are higher evapotranspiration, carbon stocks, greater rainfall security, reduced

soils erosion, and the maintenance of native habitats over a 178,000 km^2 watershed worth nine billion dollars? If the answer is affirmative, then creating incentives that facilitate private landholders' compliance with the law would seem to be strategic. The Brazilian Forest Code is an innovative legislation, and one of the first to recognize and attempt to protect the broader public interests in private land forests in the tropics. Its potential for fostering the reconciliation of conservation with agricultural development is high, but in its current state is not being realized. The Brazilian government might have achieved the objectives of defending public interests in private forests if the shift to 80% had been implemented in a different way. First, the change should have been accompanied by an effective set of options by which landholders could bring their properties into compliance with the new law. Second, the government should have developed a system of positive incentives for complying with the new regulation, potentially including compensation of at least part of the opportunity cost associated with forgone profits from soy cultivation or cattle ranching. The carbon market represents an important opportunity to achieve these economic incentives.

Table 2-1. The area and percent coverage of native forest and cerrado in the Xingu River headwaters region as required by old and new Brazilian Forest Code regulations, and as observed for 1996, 1999, and 2005. Data are presented both for the entire headwaters region and for the mean values of the region's microbasins. Results are for lands outside of protected areas.

	Required				Observed					
	Old Regulations		New Regulations		1996		1999		2005	
	Area (km^2)	%	Area (km^2)	%	Area (km^2)	%	Area (km^2)	%	Area (km^2)	%
Native Vegetation										
Whole watershed (Total)										
Legal Reserve										
Forest	48,255	50	77,208	80	79,910	83	75,139	78	61,934	64
Cerrado	10,245	35	10,245	35	18,617	64	15,967	54	13,065	45
Riparian Zone										
Forest	8487	100	8487	100	7584	89	7354	87	6814	80
Cerrado	2571	100	2571	100	2117	82	1964	76	1733	67
Total	11,058	100	11,058	100	9700	88	9318	84	8547	77
Microbasins (Mean (s.e.))										
Legal Reserve										
Forest	28 (0.6)	50	44 (0.9)	80	46 (1.0)	85 (0.5)	43 (1.0)	80 (0.6)	35 (0.8)	68 (0.6)
Cerrado	22 (1.1)	35	22 (1.1)	35	41 (2.1)	66 (1.1)	35 (1.8)	58 (1.2)	28 (1.5)	49 (1.2)
Riparian Zone										
Forest	4.8 (0.1)	100	4.8 (0.1)	100	4.3 (0.1)	90 (0.3)	4.0 (0.1)	88 (0.4)	3.9 (0.1)	80 (0.5)
Cerrado	5.7 (0.3)	100	5.7 (0.3)	100	4.7 (0.2)	84 (0.7)	4.3 (0.2)	79 (0.8)	3.8 (0.2)	70 (0.9)
Total	5.0 (0.1)	100	5.0 (0.1)	100	4.4 (0.1)	89 (0.3)	4.0 (0.1)	86 (0.4)	3.9 (0.1)	78 (0.5)

Table 2-2. The number and percentage of microbasins in the Xingu River headwaters region that were in compliance with Brazilian Forest Code forest cover requirements in 1996, 1999, and 2005. Data are for microbasins outside of protected areas.

		Required		Observed					
				1996		1999		2005	
		Number in Compliance	%	Number in Compliance	%	Number in Compliance	%	Number in Compliance	%
Microbasins									
Legal Reserve									
Forest	50%	1756	100	1608	92	1543	88	1383	79
	80%	1756	100	1259	72	1123	64	810	46
Cerrado	35%	460	100	423	92	377	82	359	78
Riparian Zone									
Forest	100%	1749	100	748	43	640	37	344	20
Cerrado	100%	454	100	65	14	59	13	36	8
Total	100%	2203	100	813	37	699	32	380	17

Table 2-3. Evolution of compliance with old and new Brazilian Forest Code in the microbasins of the Xingu River headwaters region, by category of change for 2 time periods. “Illegal” and “legal” deforestation refers to those microbasins that were above or below, respectively, the maximum percent of legal reserve clearing allowed by the Forest Code. “Continued non-compliance” refers to microbasins that began the time period out of compliance. “New non-compliance” refers to microbasins that moved from compliance to non-compliance during the time period.

	Period 1 (1996-1999)				Period 2 (1999-2005)			
	50% Legal Reserve		80% Legal Reserve		50% Legal Reserve		80% Legal Reserve	
	Area (km^2)	No. microbasins	Area (km^2)	No. microbasins	Area (km^2)	No. microbasins	Area (km^2)	No. microbasins
Deforestation								
Illegal								
Continued non-compliance	444 (9%)	125 (10%)	2129 (44%)	430 (36%)	854 (6%)	179 (12%)	5817 (44%)	570 (38%)
New non-compliance	756 (16%)	68 (6%)	1474 (30%)	145 (12%)	3953 (27%)	171 (11%)	5361 (41%)	334 (22%)
Legal								
In compliance	3667 (75%)	1008 (84%)	1263 (26%)	624 (52%)	8386 (58%)	1162 (77%)	2006 (15%)	603 (40%)

Table 2-4. Total area and mean area per microbasin of forest and cerrado within the Xingu River headwaters region that is "available" for conversion and/or needing to be restored under the old and new Brazilian Forest Code, measured at three dates. Data are for microbasins outside of protected areas. "Available for conversion" refers to forest and cerrado that could be cleared without exceeding the maximum percent clearing allowable under the Forest Code.

		1996		1999		2005	
		Total (km^2)	Mean (*s.e.*) per microbasin (km^2)	Total (km^2)	Mean (*s.e.*) per microbasin (km^2)	Total (km^2)	Mean (*s.e.*) per microbasin (km^2)
Area available for conversion							
Forest	50%	32,879	20.4 (0.5)	28,920	18.7 (0.5)	20,284	14.7 (0.4)
	80%	9221	7.3 (0.2)	7403	6.6 (0.2)	3992	4.9 (0.2)
Cerrado	35%	8499	20.1 (1.1)	6662	17.7 (1.0)	5362	14.9 (0.9)
TOTAL							
	50%	41,378	20.4 (0.4)	35,582	18.5 (0.4)	25,646	14.7 (0.4)
	80%	17,720	10.6 (0.3)	14,065	9.4 (0.3)	9354	8.0 (0.3)
Area to be restored							
Legal Reserve							
Forest	50%	1224	8.3 (0.7)	1929	9.1 (0.7)	3490	9.4 (0.5)
	80%	6520	13.1 (0.7)	9366	14.8 (0.7)	16,125	17.1 (0.6)
Cerrado	35%	104	3.0 (0.5)	318	3.9 (0.5)	487	4.9 (0.8)
TOTAL							
	50%	1328	7.3 (0.6)	2247	7.6 (0.5)	3977	8.4 (0.5)
	80%	7744	12.5 (0.7)	3684	13.6 (0.6)	16,612	16.0 (0.6)
Riparian Zone							
Forest	100%	894	0.5 (0.03)	1126	0.6 (0.03)	1673	1.0 (0.04)
Cerrado	100%	449	1.0 (0.1)	603	1.3 (0.1)	839	1.9 (0.1)
TOTAL		1344	0.6 (0.02)	1728	0.8 (0.03)	2512	1.1 (0.04)

Table 2-5. Potential net present value (NPV) associated with soybean cultivation or cattle production on forested and deforested land for the Xingu River headwaters region under alternative Brazilian Forest Code requirements at three dates. Data are for areas outside of protected areas in the forest biome region, for microbasins in compliance with 50 and 80% legal reserve requirements, and for microbasins out of compliance with 50 and 80% legal reserve requirements. Also, potential NPV that could be realized through additional legal clearing, summed across microbasins, for 50 and 80% legal reserve requirements.

		1996	1999	2005
		NPV (million USD)	NPV (million USD)	NPV (million USD)
Potential NPV				
All remaining forest (potential NPV)		26,215	24,266	19,138
All cleared land		6668	8617	13,746
Total		32,883	32,883	32,883
Compliant microbasins				
50%	Cleared	5070	5799	6905
	Forest	25,388	22,963	16,330
80%	Cleared	2673	2581	2541
	Forest	20,536	16,793	9388
Non-compliant microbasins				
50%	Cleared	1598	2818	6841
	Forest	827	1303	2809
80%	Cleared	3996	6036	11,204
	Forest	5680	7473	9750
Potential additional NPV through legal forest clearing				
Compliant microbasins				
	50%	11,595	9646	4518
	80%	2853	904	-4224

Table 2-6. Mean potential net present value of soy cultivation or cattle ranching for cleared and forested land of compliant and non-compliant microbasins of the Xingu River headwaters region at three dates. Results are for microbasins outside of protected areas.

		1996	1999	2005
Mean potential NPV per hectare (USD ha^{-1})				
Compliant microbasins				
50%	Cleared	2444	2467	2416
	Forest	2668	2673	2478
80%	Cleared	2009	1892	1772
	Forest	2588	2552	2211
Non-compliant microbasins				
50%	Cleared	2480	2513	3102
	Forest	2465	2510	3087
80%	Cleared	2877	2865	3087
	Forest	2961	2958	2997

Table 2-7. Mean potential net present value associated with soy cultivation or cattle ranching on cleared and forested lands in microbasins of the Xingu River headwaters region at three dates. Data are presented for all microbasins, for compliant and on-compliant microbasins, and for forest land that could be legally cleared.

			1996 Mean potential NPV per microbasin (s.e.) (million USD)	1999 Mean potential NPV per microbasin (s.e.) (million USD)	2005 Mean potential NPV per microbasin (s.e.) (million USD)
Potential NPV					
All microbasins					
	Forest (potential)		11.8 (0.4)	10.9 (0.4)	8.6 (0.3)
	Cleared		3.0 (0.2)	3.9 (0.2)	6.2 (0.3)
	Total		14.8 (0.5)	14.8 (0.5)	14.8 (0.5)
Compliant microbasins					
	50%	Cleared	2.5 (0.2)	3.0 (0.2)	4.0 (0.2)
		Forest	12.5 (0.4)	12.0 (0.4)	9.4 (0.3)
	80%	Cleared	1.6 (0.1)	1.7 (0.2)	2.2 (0.2)
		Forest	12.2 (0.4)	11.2 (0.4)	8.0 (0.4)
Non-compliant microbasins					
	50%	Cleared	8.8 (0.8)	9.6 (0.7)	14.5 (1.0)
		Forest	4.5 (0.5)	4.4 (0.4)	6.0 (0.4)
	80%	Cleared	7.5 (0.4)	8.4 (0.4)	10.7 (0.5)
		Forest	10.7 (0.7)	10.5 (0.6)	9.3 (0.4)
Potential additional NPV on legally cleared lands					
	50%	All	5.2 (0.2)	4.4 (0.2)	2.0 (0.1)
	80%	All	1.3 (0.1)	0.4 (0.1)	-1.9 (0.2)

Table 2-8. The average and range of estimated costs of restoration (million USD) of the legal reserve and riparian zone under 2 iterations of the Forest Code in relation to three points in time, for the entire basin in the aggregate and for the average microbasin out of compliance. Ranges refer to upper and lower estimates of restoration costs per hectare.

		1996 *Whole Basin* (million USD)	1999 *Whole Basin* (million USD)	2005 *Whole Basin* (million USD)
Legal Reserve				
Forest	50%	114 (±48)	180 (±76)	325 (±138)
	80%	607 (±258)	872 (+/371)	1502 (±638)
Cerrado	35%	9 (±4)	30 (±12)	45 (±19)
Riparian Zone				
	100%	252 (±180)	324 (+/232)	471 (±337)
Total				
	50%	375 (±232)	534 (±322)	841 (±494)
	80%	868 (±442)	1226 (–±615)	2018 (±994)
		Mean per microbasin (million USD)	*Mean per microbasin* (million USD)	*Mean per microbasin* (million USD)
Legal Reserve				
Forest	50%	0.8 (±0.3)	0.8 (±0.3)	0.9 (±0.3)
	80%	1.2 (+/0.5)	1.4 (±0.6)	1.6 (±0.7)
Cerrado	35%	0.3 (±0.1)	0.4 (±0.2)	0.5 (±0.2)
Riparian Zone				
	100%	0.1 (±0.1)	0.2 (±0.1)	0.2 (±0.2)
Total				
	50%	1.2 (±0.6)	1.4 (±0.5)	1.6 (±0.8)
	80%	1.6 (±0.7)	2.0 (±0.8)	2.3 (±1.1)

Table 2-9. Ecological attributes of the Xingu River headwaters region (including protected areas) at three 3 dates (1996, 1999, 2005) and of modeled landscapes representing the region with application of 80% legal reserve or 50% legal reserve on private lands. These attributes include: carbon stocks, surface hydrology and regional climate, indicators related to water quality, and terrestrial habitat quantity and quality.

Indicator	*Scenarios*				
	1996	1999	2005	50%	80%
Carbon Stocks					
(MtC)	623	606	546	448	623
($MtCO_2e$)	2280	2217	1998	1638	2082
Surface Hydrology and Regional Climate					
Mean Annual Discharge ($m^3 s^{-1}$)	2958	3007	3128	3235	3032
(% change from potential)	(7%)	(9%)	(13%)	(17%)	(10%)
Mean Annual Evapotranspiration ($m^3 s^{-1}$)	6754	6705	6583	6477	6679
(% change from potential)	(-3%)	(-3%)	(-5%)	(-7%)	(-4%)
Water Quality					
Riparian forest cover, 50-m buffer (km^2)	14,862	14,369	12,884	15,500	15,500
Mean % vegetation cover per microbasin (s.e.)	86 (0.4)	80 (0.5)	73 (0.5)	61 (0.5)	78 (0.4)
% of microbasins with greater than 60% vegetation cover	88	79	69	31	84
% of microbasins with less than 40% vegetation cover	4	9	14	15	14
Terrestrial Habitat					
Vegetation cover (km^2)					
Forest	86,944	82,130	71,336	51,561	80,005
Cerrado	16,496	14,326	13,238	9083	9083
Number of fragments					
Forest	9810	9609	12,295	23,673	11,439
Cerrado	7223	10,291	12,632	18,422	18,422
Mean distance to nearest neighbor fragment (m)					
Forest	383	372	367	376	390
Cerrado	404	368	385	354	354
Mean fragment size (ha)					
Forest	886	855	580	218	699
Cerrado	228	139	105	50	50
Total interior habitat area (km^2)					
Forest	82,248	76,752	63,847	36,975	68,598
Cerrado	13,373	11,082	9338	4001	4001
Total edge habitat area (km^2)					
Forest	4700	5378	7489	14,586	11,407
Cerrado	3123	3245	3900	5082	5082

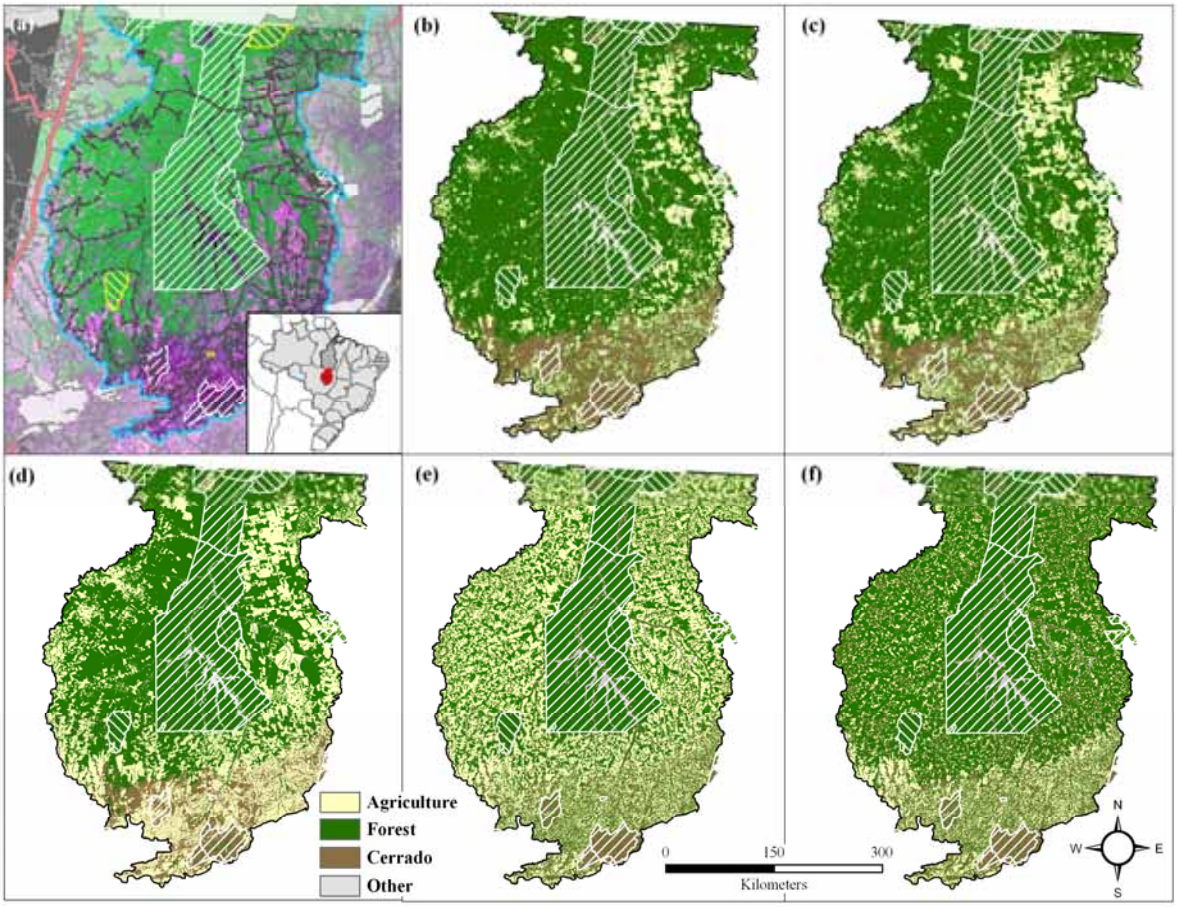

Figure 2-1. Maps showing the Xingu River headwaters region under different scenarios. (a) The Xingu River headwaters region (outlined in blue), showing federal and state protected areas (yellow), indigenous territories (white), paved roads (red), and other major unpaved roads (black). Land cover is shown for a Landsat 5 TM mosaic from 2005; greener areas indicate presence of more native vegetation or higher biomass regeneration, pinker areas indicate cleared areas or areas of low native biomass. (b-f) Comparison of land-cover in the basin representing 3 time points relevant to changes in the Brazilian Forest Code—(b) 1996, (c) 1999, and (d) 2005—and 2 alternative versions of the Forest Code, with a requirement of (e) 50% legal reserve, and (f) 80% legal reserve on private properties in the Amazon forest biome.

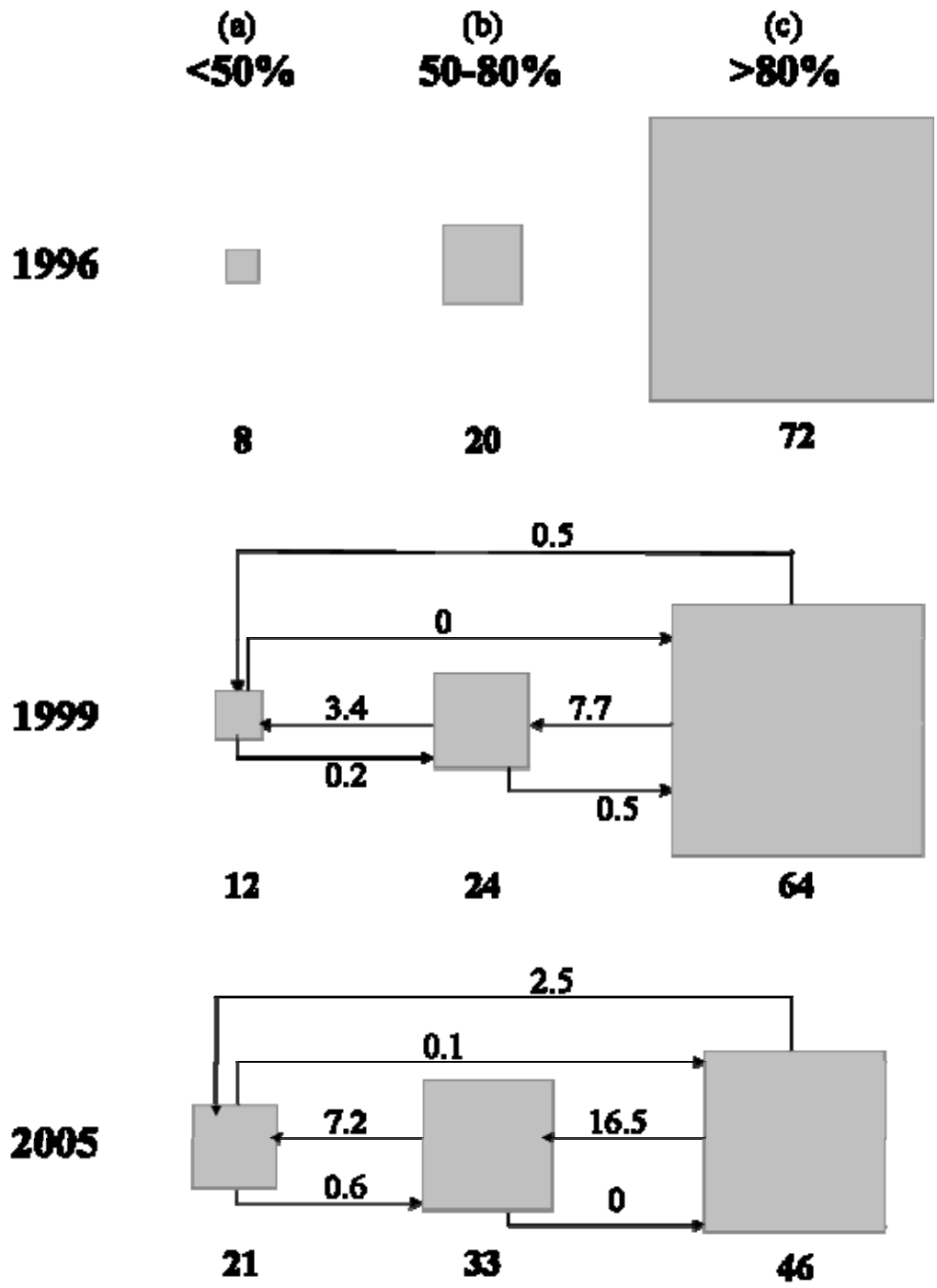

Figure 2-2. Box diagram showing the fraction of microbasins (which serve as proxies for private properties) in the forest biome in each of three categories of legality and the change in that fraction from the previous date. Categories are: (a) below 50% legal reserve, (b) between 50 and 80% native vegetation, and (c) greater than or equal to 80% native vegetation cover. The size of each box is proportional to the fraction of total forest microbasins in that category.

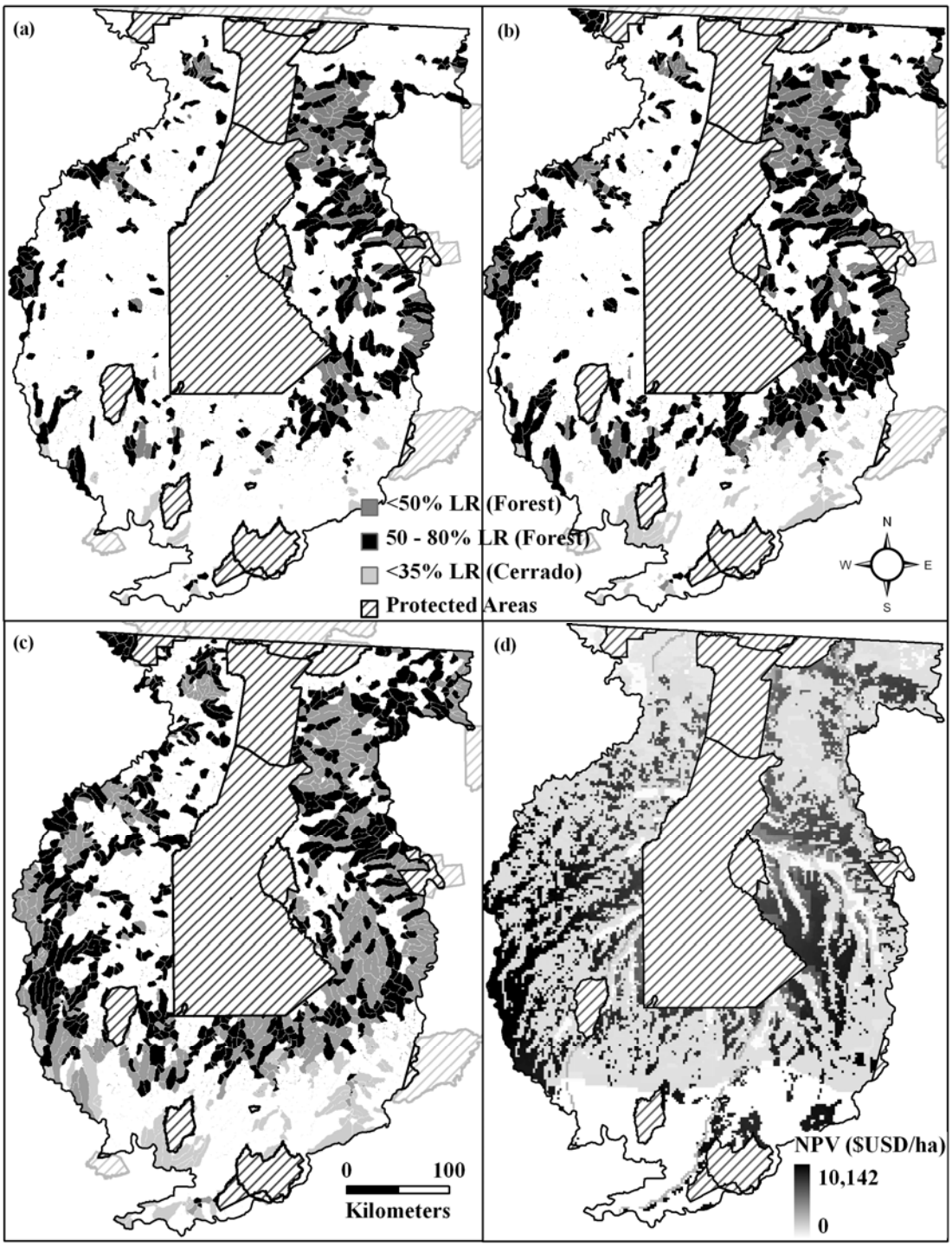

Figure 2-3. Map showing microbasins (which serve as proxies for private properties) out of compliance with two iterations (50% and 80% legal reserve requirement) of the Forest Code at three time points: (a) 1996, (b) 1999, and (c) 2005. Microbasins with <50% forest cover (forest biome), 50 to 80% forest cover (forest biome), and <35% native vegetation (*cerrado* biome) are highlighted. Map of net present value (NPV) (d) is provided for interpretation.

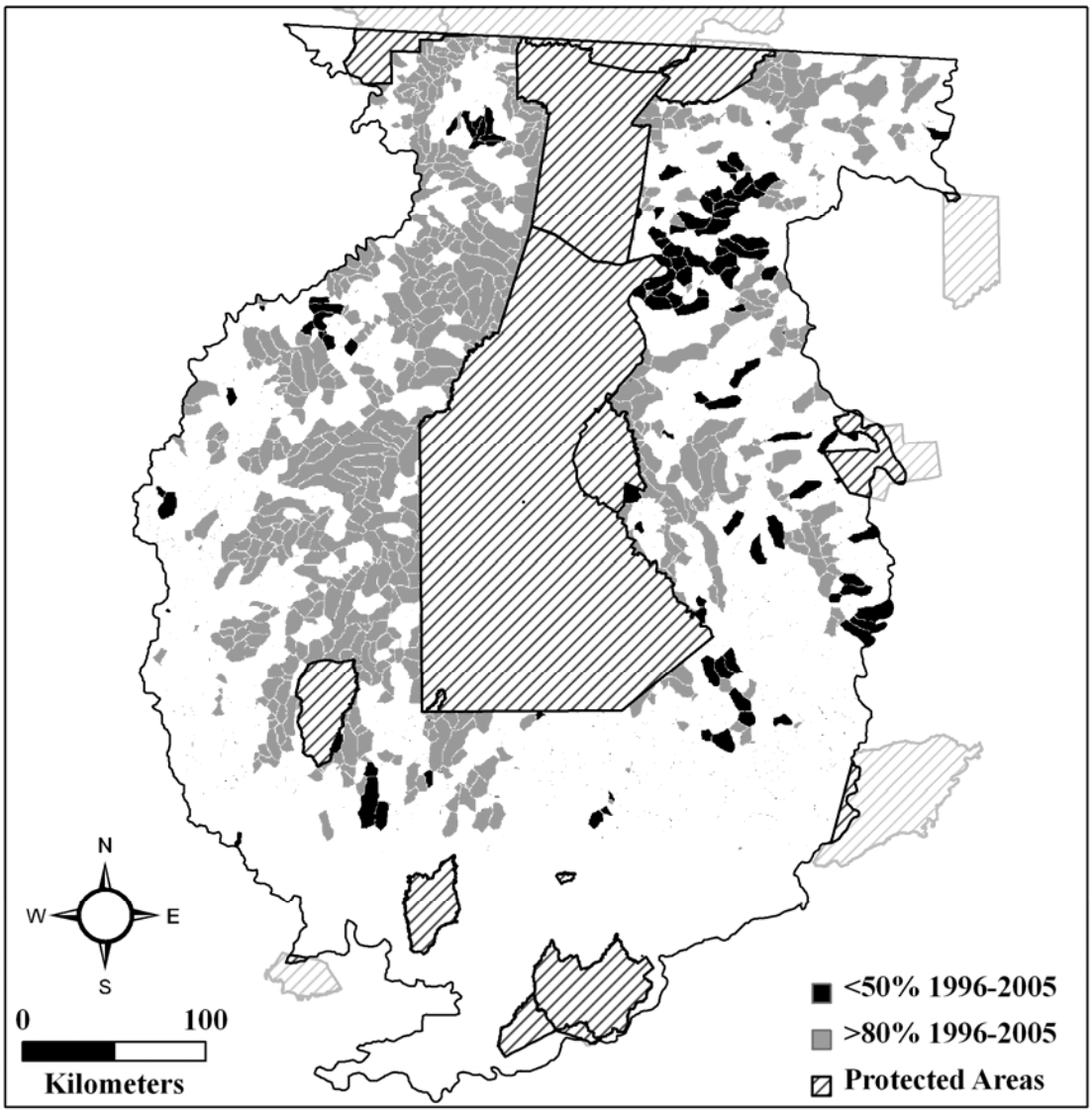

Figure 2-4. Microbasins within two extreme categories of legality in relation to the modified Forest Code: (a) microbasins with less than 50% forest cover remaining 1996 through 2005; (b) microbasins with more than 80% forest cover remaining since 1996.

CHAPTER 3
HYBRID REGULATORY-ECONOMIC POLICY INSTRUMENTS IN PRIVATE FOREST GOVERNANCE

Introduction

The main instrument for protecting forests on private lands in Brazil is the federal Forest Code, which mandates that landholders maintain a certain percentage of native vegetation in a private reserve. In 1996, the Brazilian federal government responded to record high deforestation in the previous year by raising the proportion of private properties in the Amazon to be maintained in a forest reserve from 50% to 80%. This measure provoked an intense reaction from the agro-industrial lobby, particularly in the state of Mato Grosso, Brazil's only major agricultural producing and exporting state located in the Legal Amazon.

Although it is unclear what the extent of non-compliance with the new Forest Code regulations is for the whole state of Mato Grosso, in the Xingu headwaters (a region representing 20% of the state's area and more than one quarter of the state's forest area), an estimated 28% of landholdings were non-compliant in 1996, increasing to 55% in 2005 (Chapter 2). A partial explanation of this imperfect level of compliance is offered by the large potential earnings that can be made from soy cultivation or cattle ranching in comparison with the earnings potential of selective logging. In the Xingu headwaters, the change in the legal reserve requirement imposed USD 9 billion in potentially foregone profits on the region's farmers and ranchers (an average of USD 3 to 4 million per microbasin) (Chapter 2). Powerful economic drivers to clear forest were not met with any counter-incentive from the government to facilitate landholder compliance. Moreover, the change in the Forest Code was not accompanied by effective processes and procedures through which landholders who wished to comply with the law could do

so. Finally, a number of other legal and administrative factors—including lack of financial and human resources for monitoring and conflicting messages from the state and federal governments about the exact regulations—conspired to leave landholders in doubt as to what rules they ought to adhere (Chapter 2).

In recognition of the inefficiency and ineffectiveness of the Forest Code, lobbyists and legislators have made several proposals for modifications or alternate policy instruments designed to reduce the cost of compliance both to individual landholders and to the State's economy as a whole. Among these are two "hybrid" instruments that combine a regulatory framework with an economic instrument with the goal of increasing the efficiency and effectiveness of the Forest Code. This first is a provision (already incorporated into the current Forest Code) to permit landholders having less than the required area of forest reserve to buy another landholder's rights to clear forest on the second property (MP 2611). This mechanism is a type of economic policy instrument, akin to tradable permits that have been successfully used for reducing some pollutants and gas emissions. The second policy proposal focuses on designing a land-use zoning plan that incorporates measures of agricultural suitability and ecological value and vulnerability to determine where and in what measure the requirements of the current Forest Code will be applied (SEPLAN-MT, 2008). This policy incorporates both economic and regulatory measures.

In this chapter, I present the results of a study focusing on the provision to allow tradable development rights in the Forest Code and the Mato Grosso state zoning plan these other existing legal instruments that could potentially lower the opportunity cost of complying with the new Forest Code regulations without forfeiting the environmental

gains. First, I describe each of these instruments and compare them with the Forest Code, analyzing the extent to which they represent economic instruments or exclusively regulatory instruments. Next, I carry out a quantitative analysis of the 3 instruments, focusing on the ability of each to balance agricultural production with ecological conservation. I compare modeled landscapes representing the land-cover consequences of implementing each of the three policy alternatives in a landscape on the southeastern Amazon frontier in terms of area available for agricultural activities, the costs of compliance with each theoretical landscape, and the extent to which ecosystem services are maintained.

Forest Governance on Private Lands

Brazilian environmental legislation is considered to be among the most sophisticated in the world. The early part of the 20th century saw enactment of laws that recognized and protected forest resources as a common good (Chapter 2). Successive legislation, including the 1988 Brazilian Constitution, created requirements for environmental impact assessments, environmental quality standards, and licensing and monitoring of a range of activities with environmental impacts, as well as environmental crimes legislation. However, natural resource and environmental policies have been notoriously difficult to implement in practice. Primarily, this is due to poor coordination among responsible authorities, lack of human and financial resources, and a pervasive perception of environmental policy as generally obtrusive to economic development and, thus, of significantly lower priority than many other policies (Ascher, 1999). Many key policy decisions that greatly affect natural resources are still primarily the responsibility of ministries other than the Ministry of Environment, such as the Ministry of Agriculture, the Ministry of Transportation, and the Ministry of National Integration

and Planning, which often leads to direct inconsistencies with environmental legislation. Moreover, the responsibility for developing and implementing specific environmental regulations is divided among federal, state, and municipal authorities, often without great clarity about which agency is responsible for which issues and with inadequate communication and coordination between these authorities (Ascher, 1999; May, 1999). Even where responsibility is clearer, lack of personnel or funding for enforcement can hamper policy implementation. Ascher (1999) also observes that government policies frequently are not designed to encourage landholders or land-users to invest in good land management and often do just the opposite, providing incentives to invest in inappropriate technologies or practices. Furthermore, the judicial system is ill-equipped to help to enforce the laws, largely because of its slowness and weakness in carrying out the rule of law (Ames and Keck, 1998; Brito and Barreto, 2006), and because of rampant corruption (Brito and Barreto, 2006). In this chapter, I examine examples of Brazilian policy instruments that attempt to overcome some of these obstacles to protecting native ecosystems, and the services that they provide, on privately owned properties.

Policy Instruments for Natural Resource Management

Environmental policy instruments are typically divided into three broad categories: regulation, economic instruments, and voluntary action. Whereas regulatory and occasionally economic instruments are considered to be standard tools for government policy-making world-wide, market-based mechanisms and voluntary approaches are increasingly employed to complement or substitute traditional regulatory environmental policy approaches. Regulatory (often referred to as “command-and-control”) instruments typically define performance-based or technology-based standards with which

producers or landowners are required to comply by law. Regulation can take a variety of forms, including the outright prohibition of an activity or substance, setting limits for the amount of a pollutant that may be produced, or defining the terms by which natural resources can be extracted (Sterner, 2003). Failure to comply with regulations generally involves fines or other penalties. Thus, monitoring and enforcement is a critical component of direct regulation. In contrast with regulatory instruments, economic instruments are generally considered to be more economically efficient (Carter, 2001). This is true because these instruments influence behavior, in effect, by increasing the profitability of a more desirable form of resource user behavior, hence harnessing the user's rent-seeking behavior. They include ecological taxes, user charges, deposit refunds, tradable permits, and subsidies. Finally, voluntary approaches involve substitution of state-defined regulations with self-defined implementation standards for regulations, self-financed certification systems for enforcement, and elective public reporting for public disclosure requirements (Ten Brink, 2001; Carter, 2001). In the private sector, voluntary approaches typically involve adoption of social responsibility standards and environmental management systems (EMS), which include standards such as ISO 14000[5], eco-labeling, and certification. Recently, payment-for-environmental-services (PES) schemes have become a prevalent type of voluntary approach based on the logic of an economic instrument; prominent among these are schemes focusing on carbon sequestration and avoided carbon emissions. In this chapter, I focus primarily on extant regulatory and economic instruments, returning to

[5] The ISO (International Standards Organization) 14000 environmental management standards provide a set of guidelines to help organizations develop environmental management system whose primary focus is on increasing management efficiency and reducing negative environmental impacts as a by-product, and on compliance with applicable environmental laws and regulations.

voluntary approaches in the final discussion of the results and their implications. In the following sections, I discuss 3 policy instruments to regulate land-use on private lands. The three instruments include a standard regulatory instrument and 2 hybrid economic-regulatory instruments.

The Legal Reserve of the Federal Forest Code

In Brazil, use of regulatory instruments has been the dominant policy approach to environmental problems. The Forest Code is an example of such an instrument in that it requires private landholders to maintain forest reserves on their lands under penalty of fines. Since 1965, the Brazilian Forest Code requires landowners to maintain 2 types of native vegetation reserves on their properties: (1) a “legal reserve”, and (2) a riparian buffer zone. The legal reserve requires landholders to maintain a minimum percentage—determined by the phytogeographic location of the property—of native vegetation on their properties, which may be managed for sustainable timber harvests. In 1996, the legal reserve in the Amazon forest biome was increased from 50% to 80%; the legal reserve in the *cerrado* continued to be set at 35% (Chapter 2). In practice, this requirement has been difficult to enforce, for a variety of reasons. On the one hand, monitoring and enforcement of the policy is human and financial resource intensive. Several studies carried out in Mato Grosso—the state with the highest deforestation rates for much of the past decade—demonstrate that staff to conduct monitoring expeditions is in short supply, corruption in the legal and regulatory system is rampant (Lima et al., 2005), and the costs of registering properties and maintaining the legal reserve are prohibitive (Guimaraes and Almeida, 2007). The effectiveness of the Forest Code in the Brazilian Amazon has been restricted by frequent changes in regulations, the lack of an effective mechanism for enabling landholders to comply with these

changing regulations, and the absence of economic incentives for compliance, as described in Chapter 2. On the other hand, the potential income from lands cleared for ranching or soy farming (the principal agricultural activities in the region) is 2 to 20 times greater than the potential income from reduced-impact selective timber harvest (Nepstad et al. 2009). Thus, many landowners consider the risk of fines merely a cost of doing business. Both the absence of enforcement and the profitability of agricultural production in the region (and thus, willingness to face sizeable fines when they are applied) help to explain this result (Chapter 2).

In general, the legal reserve requirement has been found to be both an economically and ecologically inefficient way to meet a quantitative target for forest area to be preserved (Soulié de Amaral, 1997; Nogueiro-Neto, 1998). According to Chomitz (2004), the reserve requirement is “economically inefficient because it requires the same amount of preservation (or restoration) of forest cover regardless of the property’s agricultural potential or market access.” Moreover, it is environmentally inefficient because (1) it does not prioritize forest cover according to ecological or biodiversity value, and (2) it tends to result in fragmented forests (Chomitz, 2004). In addition, compliance with the Forest Code does not reach 100%. In the Xingu River headwaters region (in northeastern Mato Grosso), compliance with the increased legal reserve requirement fell from an estimated 72% of properties in 1996 to 46% in 2005 (Chapter 2). Simultaneously, the area requiring restoration of native vegetation under the law increased more than 2-fold and the opportunity cost (resulting from lost potential net present value) increased by nine billion dollars. In an effort to increase compliance with the regulations by reducing the cost to producers, lawmakers included 2 provisions in

the modified version of the Forest Code that was finally adopted into law in 2001: (1) the tradable deforestation rights scheme required that an RL be “compensated” in another area of equal extent and ecological function and character within the same micro-basin (Chomitz, 2004); (2) the legal reserve could be reduced to 50% from 80% in the Amazon forest biome within the context of state zoning plans. I consider both of these mechanisms to be a hybrid regulatory-economic instrument in that each essentially maintains the requirements of the Forest Code but permits individual landholders to determine the way in which he/she will meet the requirements. I describe these instruments in further detail below.

Tradable Deforestation Rights

Tradable permits represent a rights-based mechanism for internalizing environmental externalities. The tradable permits (or ‘cap-and-trade’) system represents a relatively new institutional approach to manage the problem of rationing access to the commons (Tietenberg, 2002). It has been applied to a number of different resources or resource systems, including air pollution control, fisheries, water resource management, water pollution control, and land-use control (Tietenberg, 2002; Chomitz 2004). Tradable permits address the commons problem by rationing access to the resource and privatizing the resulting access rights. The first step involves setting a limit (or “cap”) on user access to the resource. In the case of fisheries this would involve the total allowable catch, for example. For water supply, it would involve the amount of water that could be extracted. For pollution control it typically specifies the aggregate amount of access to the resource that is authorized or the total amount of pollution that is permitted. These access rights are then distributed (typically by direct allocation or by a public auction) to potential individual users. Depending on the specific system, these

rights may be transferable to other users (or “tradable”) and/or bankable for future use. Users who exceed limits imposed by the rights they hold face penalties up to and including the loss of the right to participate (Tietenberg, 2002).

The tradable permits system essentially creates a market for a common pool resource, if there is force of law accompanying it. Generally it will not work on a voluntary basis or under conditions of low enforcement, since those individuals or firms that comply accrue higher costs of production than those who do not (Tietenberg, 2002). The key difference between cap-and-trade systems and command-and-control is that the latter seeks to control emissions at the individual level, whereas cap-and-trade seeks to control emissions at the aggregate level. Such a system encourages innovation and incentivizes the least-costly means to meet the targets. This flexibility is likely to produce a higher level of compliance and thereby lead to meeting the overall environmental target more readily. Furthermore, as Carter (2001) explains, “In short, regulations provide no incentive for polluters to reduce their pollution any further than that required by law. [Market-based instruments] are intended to provide that incentive.” Like command-and-control systems, tradable permits systems require adequate monitoring and enforcement.

When the legal reserve requirement in the Amazon forest biome was increased to 80% in 1996, approximately one-fifth of landholders who had been in compliance with the law up to that date found themselves suddenly in violation of the new Forest Code regulations (Chapter 2). Under the new law, they were required to reforest their land to meet the new legal reserve requirement. However, this represented both the collective loss of nine billion USD in potential net present value and a collective cost of 1 to 3

billion USD in restoration costs (Chapter 2). As a means of bringing these landholders into compliance, a 1998 re-edition (*MP 1736/98*) of the provisionary measure MP 1.511 modifying the legal reserve included a provision to allow landowners whose property does not meet the legal reserve requirements to compensate the 'missing' reserve area elsewhere within the same watershed, provided it has equal or greater ecological value. More specifically, landowners who had deforested more than 20% of their land, but not more than 50% before December 31, 1999, were eligible to participate in the scheme. Instead of handing over title to the land, the owner of land having an amount of forest over and above the minimum requirement for the legal reserve would sell the rights to develop that land to another owner with inadequate forest to meet the requirements.

To date, however, the option to trade development rights within the Forest Code has been little exercised within the Legal Amazon. The major barrier is the lack of sufficient current information about individual properties' forest reserves (due to inadequate monitoring and enforcement). Even in the state of Mato Grosso, where the innovative digital licensing system should facilitate the legal reserve compensation system (but see Azevedo, 2009), only 5 transfers have been documented as having taken place since 2000. This is due in large part to disorganization in data base management within the licensing system (Lima et al., 2005; Azevedo, 2009). In addition, however, the legislation does not make clear at what scale development rights trading may be carried out. Although it specifies that it may be carried out within the same watershed, it is unclear about what size of watershed. As Chomitz (2004) notes, if trading is carried out on a smaller scale (at the level of lower-order watersheds), the feasibility of monitoring and enforcement is likely to increase, thus decreasing the

transaction costs (from society's point of view, if not the landowners). Furthermore, trading on a smaller scale increases the homogeneity or substitutability of the forest areas involved. However, reducing the scale may also severely restrict the potential of the system to lower the economic costs of environmental management by reducing available options (Chomitz, 2004).

State Socioeconomic-Ecological Zoning Plans

Since the enactment of the 1988 Constitution, the primary instrument for regional land-use planning in Brazil within the National Environment Policy (*Politica Nacional de Meio-Ambiente, PNMA*) is intended to be the *Zoneamento Socio-Economico Ecologico* (ZSEE; Socioeconomic Ecological Zoning Plan). The state zoning plans specify the types or intensities of land-uses which may occur throughout a given state, based on analyses incorporating biophysical and economic factors. It was not until 2002 that a federal decree (*Decreto Federal 4.297/2002*) established criteria that all state zoning plans would be required to meet to be recognized by the national government, including being approved by the respective state's legislative assembly and ZSEE Commission. Several federal technical agencies—including the National Institute for Space Research (INPE), the Brazilian Enterprise for Agricultural Development (EMBRAPA), and the Brazilian Institute of Geography and Statistics (IBGE)—are charged with assisting states in developing their zoning plans. One of the most important features of the state zoning plans is that they may supersede the Forest Code, relaxing or raising the size of the legal reserve in some zones, for example.

Although the federal Ministry of Environment set a deadline of July 2009 for states to approve their ZSEE plans, only 2 of Brazil's 9 Legal Amazon states—Rondônia and Acre—met this deadline. In Mato Grosso, the ZSEE has been under development and

discussion for 18 years (Folha de São Paulo, 2009). The state Planning Secretariat (SEPLAN) presented what it considered to be a final plan together with the state environmental agency (FEMA, at the time) to Governor Blairo Maggi in 2004. However, the plan was heavily criticized by environmentalists and following reorganization of the state's environmental regulatory and management structure by the federal government in 2005, a new proposal began to be developed. A new draft of the plan was presented to the governor in April 2008, but was strongly criticized by the agro-industrial sector as it increased the land area in protected areas to 27%, up from 20%. In part, the state Planning Secretariat increased the proportion of protected areas based on recommendations from INPE, which had predicted increased deforestation rates in the second half of 2007, while the ZSEE proposal was still being modified. In contrast, the proportion of the state dedicated to agricultural production (with more limited environmental requirements) amounted to only 11% of the total area. Since the proposal was unveiled, the ZSEE has undergone a lengthy consultation process (originally slated to be completed before the end of 2008, it continues into the second half of 2009) throughout the state with local stakeholders and the involvement of national civil society and government agencies. There is considerable concern about the validity of the plan, since the majority of the data on which the plan is based were collected between 1995 and 1997, before the major expansion of agro-industry into the forest zones of the state had begun.

The Mato Grosso ZSEE plan is a hybrid regulatory-economic instrument in that it requires landholders to comply with the Forest Code, but redefines the proportion of a property that must be protected using a combination of economic and ecological criteria

to reduce some of the inefficiencies previously identified in the Forest Code. Furthermore, in the version of the zoning plan currently being promoted by environmentalists in Mato Grosso (A. Lima, pers. comm., April 15, 2009), application of the tradable deforestation rights scheme described above will be permitted. The plan identifies 4 major zones (Figure 3-1), each having specific rules and restrictions. The zone classification determines the fiscal incentives and public expenditures that will be made available, as well as the environmental licensing requirements for an area. Among the major issues under debate—aside from the proportion of land devoted to protected areas—is whether the legal reserve in Zone 3 (see below) should remain at 80% and whether landholders having less than 80% forest on their land must restore the full 80%, as currently stipulated under the ZSEE. The zones are defined as follows:

- **Zone 1**. Areas of historical long-term dedicated agricultural use or areas to be dedicated to agricultural production in the future

  In this zone, the Forest Code will be altered, if approved. Private properties may be deforested up to 50%; if more than 50% of the area has already been deforested, this excess must only be restored to 50% of the property area. Properties in the *cerrado* biome may not clear more than 65% of the total area and must restore up to 35% of native cerrado vegetation, if more has been cleared.

- **Zone 2**. Areas requiring correction of management systems: In this zone, the Forest Code may be altered

  Private properties may not deforest more than 20% of the total area; if more than 20% of the area has already been deforested, this excess must only be restored to 50% of the property area. Properties in the *cerrado* biome may not clear more than 65% of the total area and must restore up to 35% of native cerrado vegetation, if more has been cleared.

- **Zone 3**. Areas requiring special management: In this zone, the Forest Code may not be altered

  Thus, in the forest biome, private properties may not deforest more than 20% of the total area; if more than 20% of the area has been deforested previously, this excess must be restored. However, eligible properties may participate in trading their deforestation rights, as explained above. Properties in the *cerrado* biome may

not clear more than 65% of the total area and must restore up to 35% of native cerrado vegetation, if more has been cleared.

- **Zone 4**. Protected areas: this category includes all current and proposed protected areas

  No deforestation is allowed in this zone. Theoretically, reforestation will take place in proposed protected areas, as they currently encompass privately-owned areas.

Materials and Methods

Study Area

The 177,780 km^2 Xingu River headwaters region is located in the northeastern corner of Mato Grosso state, in central Brazil (Figure 3-2). The region's soils, topography (100-300 m altitude, with flat interfluvial expanses) and climate are well-suited for soybean production and cattle ranching. Native vegetation types in the region are comprised of forests (tall evergreen, transitional semi-deciduous, and riparian) and savannas (*cerrado* woodland, mosaics of grassland, thickets, gallery forests) which comprise approximately two-thirds and one-third of region, respectively. Ten indigenous territories are completely contained within the boundaries of the Xingu watershed in Mato Grosso (Figure 3-2). Indigenous territories cover approximately 42,200 km^2 within the basin, representing 24% of the total area of the headwaters region. Private landholdings comprise nearly 70% of the total area, and smallholder settlements comprise less than 5% of the region. The streams and rivers of the major protected forest area that lies at the center of the region—the PIX complex, which alone comprises nearly 20% of the headwaters area—are under growing threats from sedimentation, agrochemical run-off, and associated fish die-off from the unprotected headwaters regions outside of the park boundaries (Sanches, 2002). The Xingu region is representative of many areas along the Amazon's agricultural frontier, but faces a

more acute and immediate threat because it lies between two major federal highways (BR-158, BR-163) that are partially paved, and lies in the pathway of the northward expansion of Brazil's grain belt.

Land Cover Maps

I developed maps of the observed landscape in 2005 and 3 theoretical landscapes reflecting land-cover corresponding to alternatives for regulating land-use on private lands in Mato Grosso. I used these maps to estimate and compare the amount of vegetation—remnant native and restored or regenerating—that would be maintained under each scenario. Furthermore, I used these maps to calculate the area available for agricultural production under each policy alternative. I also combined them with maps of net present value for the region to estimate and compare the potential net present value of cleared lands in the region under each scenario in the aggregate and on average per microbasin. Finally, the maps were also used as inputs for calculating several ecological indicators.

Observed landscapes

I developed maps of land-cover for 2 dates of significance regarding private land-use legislation in Mato Grosso: 1999 and 2005. 2005 is the year in which the federal government dissolved the Mato Grosso state environmental "foundation" (subsequently reorganized as the state environmental secretariat) and mandated that landholders in Mato Grosso follow the requirements of the Forest Code as modified at the federal level in 2001, rather than as interpreted by the state government since 2000. Here, 2005 also represents the current landscape in the Xingu River headwaters, as very little new deforestation took place since that year. The map for 1999 was used to identify those landholders eligible to participate in the tradable deforestation rights (or compensation)

scheme. Maps with 4 classes (forest, cerrado, agricultural lands, other) were classified as described in Appendix A.

Modeled landscapes

Three theoretical landscapes corresponding to the requirements set forth by (a) the current Forest Code (after 1996) without the compensation option (that I refer to here as “CFC-NC”; Chapter 2), (b) the Forest Code with the compensation option (“CFC-C”), and (c) the Mato Grosso State socio-economic, ecological zoning plan (referred to here as “ZSEE”, SEPLAN-MT, 2009) were developed using a spatially explicit dynamic landscape simulation model. The basic architecture and function of the simulation model is described in Appendix A; details regarding the modeled landscapes follow here. The assumptions underlying each of the three scenarios differed only in the percent and location of native vegetation that was to be maintained or restored on each private land-holding in the headwaters region in accordance with the rules of each alternative mechanism, as described below. Since a complete map of property boundaries for the region is not available, I used micro-basins representing individual stream reaches (1:1,000,000 scale) as proxies for individual properties. The mean, range, and distribution of sizes of the 2881 microbasins microbasin ($\bar{x}$ = 5981 ha, 4-70,766 ha) are comparable to that of private properties in the region (Jepson, 2006; Fearnside, 2005; Appendix B). Furthermore, the mean percent clearing in the current (2005) landscape is comparable among microbasins and properties for which property limit data are available, indicating that microbasins are suitable substitutes for individual properties in terms of sampling the population of properties in the region. It was necessary to use a spatial unit with complete coverage of the study region to simulate

the distribution of vegetation across the landscape corresponding with the requirements of the pre- and post-1996 versions of the Forest Code. The tradable deforestation rights provision in the Forest Code stipulates that property owners who have exceeded the legally permitted clearing of their land can compensate this forest deficit within the same microbasin, as described below.

For all three scenarios, full compliance with the law was assumed. All indigenous reserves and state and federal protected areas were strictly protected. Furthermore, a 50-m riparian buffer zone surrounding each stream and river visible in a map derived from a thematic stream layer obtained from the Mato Grosso State Regional Planning Secretariat (SEPLAN-MT) was strictly protected. The Forest Code stipulates that a riparian buffer zone of at least 50 m be protected around every natural body of water, and that the size of the buffer zone (up to 500 m) is dependent on the width of the stream. However, as stream width is difficult to assess and no official map of riparian buffers exist, the estimation of variable-width riparian buffers was not possible and I assumed, conservatively, that all water bodies were surrounded by a 50-m wide buffer. Where necessary, vegetation was restored to this riparian buffer zone so that each theoretical landscape had 100% native vegetation cover within the boundaries of the riparian zone.

To calculate the amount of deforestation or restoration that could or should take place, respectively, outside the riparian zone to meet the legal reserve requirements under each of the three scenarios—subject to further restrictions or allowances, as described for each scenario below—I classified each micro-basin according to biome (cerrado or forest). The cerrado-forest biome map was obtained by merging a map of

forest/non-forest derived from INPE Prodes maps with a map of biomes from the IBGE RADAM vegetation thematic map. Of 2881 micro-watersheds in the Xingu River headwaters region, 34 straddled both biomes. To facilitate model design and processing, each of these microbasins was assigned to the biome representing more than 50% of that micro-watershed's area. At time step 0, the model calculates how much of each watershed's area consists of cleared area. If this area is greater than the allowed amount (as determined by each iteration scenario's conditions) after the area of the riparian zone is subtracted from the total watershed area, the model reforests the area up to the allowable amount of cleared area. If the cleared area is less than the allowable area, the model begins to deforest the watershed based on where the highest favorability for deforestation is indicated (Appendix A).

In this study, I modeled 3 alternative scenarios over a 60-year time period (to ensure that the maximum allowable clearing and regenerating under each scenario's assumptions would be achieved) in annual steps, using 2005 as a starting point for: (1) Current Forest Code, No Compensation (CFC-NC), (2) Current Forest Code, Compensation (CFC-C); and (3) State Zoning Plan (ZSEE). The assumptions and conditions for each scenario are described below.

Current Forest Code, no compensation The CFC-NC scenario assumes full compliance with the current Forest Code, such that properties (microbasins) located in the forest biome maintain a legal reserve of 80% and those in the *cerrado* maintain a reserve of 35%. In addition, all properties must maintain 100% forest cover in riparian zones.

To calculate the amount of deforestation or restoration that could or should take place, respectively, I classified each micro-basin according to biome (cerrado or forest), as micro-basins serve as proxies for individual properties in the model. The cerrado-forest biome map was obtained by merging a map of forest-non-forest derived from INPE Prodes maps with a map of biomes from the IBGE RADAM vegetation thematic map. Of 2881 micro-watersheds, 34 straddled both biomes. To facilitate model design, each of these micro-watersheds was assigned to the biome representing more than 50% of that micro-watershed's area. At time step 0, the model calculates how much of each watershed's area consists of cleared area. If this area is greater than the allowed amount (see amounts as defined by zone below) after the area of the riparian zone is subtracted from the total watershed area, the model reforests the area up to the allowable amount of cleared area. If the cleared area is less than the allowable area, the model begins to deforest the watershed based on where the highest favorability for deforestation is indicated (see Simulation Model description).

I generated a riparian buffer zone map by applying a 50-m buffer to either side of streams, as this width represents the minimum required under the Brazilian Forest Code. According to the legislation, the buffer width should vary with stream width. However, as I did not have access to data on stream width for the entire region, I assume that the minimum riparian zone width will be maintained, at a minimum. At time step 0 (the initial 2007 landscape, in this case), the model determines how much, if any, of the area within the limits of the riparian buffer within each micro-watershed is cleared, then proceeds to reforest the cleared area such that 100% of the riparian zone is

forested or in regeneration. Both currently forested and regenerating areas were prohibited from being cleared in the future.

Current Forest Code, compensation Under the *CFC-C* scenario, landholders are required to comply with the requirements of the Forest Code as described above. However, properties with less than the minimum required forest cover and that meet certain other conditions, explained below, are permitted to buy the rights to the forested area of another property to meet the legal reserve requirement. The concept is essentially a version of tradable development rights schemes, such as “cap-and-trade” programs, for deforestation rights. Only properties which had already deforested more than 20% of the property’s area by January 1, 2000, are eligible to buy the deforestation rights of a property that currently still maintains more than 80% forest cover, up to the amount that must be restored on the first property. In this case, the second property gives up the right to deforest that portion for which the rights have been sold.

To include the option of tradable deforestation rights, the model employs a mask that identifies properties eligible for participation in the scheme to calculate, within each of the 6 sub-basins, the amount of remaining forest (from properties having more than 80% forest cover) that may not be deforested because it will be set aside in compensation for one of the micro-basins that is eligible to reconstitute its own legally inadequate reserve by this means. Furthermore, the model determines how much reforestation is required using these parameters. If more than 20% of a micro-basin’s area was already cleared prior to January 1, 2000, then the model only requires it to reforest up to 50% of the total area of the microbasin outside the riparian zone, or in the case where a trade is made, to set aside the equivalent of this area within the sub-

basin. However, if less than 20% was cleared by 1999, but more than 20% was cleared by 2008, then the model reforests (or sets aside) up to 80% of the total area of the microbasin outside the riparian zone. These conditions conform to those set forth in the MP 2611 (the 2001 version of the Forest Code).

State zoning plan The *ZSEE* scenario represents land-use/land-cover in the watershed if the Mato Grosso state zoning plan were to be approved and implemented (SEPLAN-MT, 2009). Although the plan requires that landowners observed the regulations of the Forest Code as described above, it allows restrictions imposed by the current Forest Code to be relaxed in specific zones, as dictated by the zoning plan. In addition, for eligible properties, as described above, compensation is applied under the zoning plan. Finally, the state zoning plan highlights a number of critical biophysical restrictions that limit agricultural development and encourage forest conservation or restoration.

To develop the ZSEE scenario, I produced several maps delineating critical restrictions and zones in which the scenario conditions would be applied (Figure 3-1). The map of zones was derived from the official state zoning plan map, corresponding to the 4 major categories identified by the plan, as described above (in State Ecological-Economic Zoning Plans).

In addition, I identified three critical restrictions among the list of directives of the ZSEE that are explicitly spatial in nature and that I determined to be limiting such that areas described by this set of restrictions would not be eligible for relaxation of the current Forest Code, regardless of which zone they fall in. Moreover, the areas described by this set of restrictions would be subject to full protection of the native

vegetation, and most likely, require restoration of the native vegetation should the vegetation no longer be intact. The restrictions (defined in the ZSEE; SEPLAN-MT, 2009) and methods I used for delineating spatial layers are as follows:

- **Directive 7.** Strictly protect floodplains, prohibiting any type of vegetation clearing, with the exception of clearing carried out for purposes of subsistence agriculture or for clearing of native pasture without the use of fire.

  I derived a map of floodplains in the basin from SRTM data (90-m resolution oversampled to 30-m; USGS 2007; Farr et al., 2007) for the region by dividing the region into 3704[6] micro-watersheds and calculating the negative exponential decay (-0.01) of distance upstream from each watershed's pourpoint to calculate the distance between the elevation at each point along each stream and the elevation at the pourpoint of the stream reach. I used this scale to estimate the reasonable flood/non-flood elevation at each point upstream from the pourpoint.

- **Directive 42.a.** Permit agro-pastoral uses only under suitable soil morphological conditions, prohibiting agro-pastoral uses in sensitive environments without appropriate slope and soil conditions, particularly on sandy and alluvial soils that are not well drained, because of the importance of these areas to the stability of the local and regional hydrological regime.

  I obtained a thematic map of soil types for the region (SEPLAN/SEMA-MT) and identified all soil types that can be categorized as sandy or alluvial. In addition, I used the SRTM data to derive a map of slope and then identified all areas with a slope greater than 10% as being restricted, as this is the maximum value defined by the Brazilian Forest Code, above which permanent vegetation cover is required to be maintained. Finally, I combined both restricted soils and slopes into one layer to represent the spatial restrictions represented by this directive.

- **Directive 81.** Strictly protect aquifer recharge zones under "campos umidos" and "murunduns" which are sensitive habitats essential for the maintenance of water resources, eliminating any interference or establishment of structures that alter the hydrological regime and accelerate erosion processes; these areas should be assigned to legal reserves.

  I identified wetland areas by merging all areas identified as wetlands in 3 Landsat-based classifications for the years 1996, 2005, and 2007. Classification accuracy for wetlands in these classifications separately ranged from 0.88 to 0.91 (mean 0.90).

[6] This set of microbasins includes all microbasins in the region, not just those outside of protected areas which comprise the set used in the remainder of the analysis.

I merged the layers generated for each of these restrictions to obtain a single restriction map that was used in the development of the *ZSEE* scenarios. I adapted the *CFC-NC* scenario described above by assigning each micro-watershed to a zone and altering the percentage of allowable deforestation and required reforestation in each watershed according to zone as well as to biome. Furthermore, the model reforested both the riparian zone and the area indicated by the composite map of restrictions; both areas were subtracted from the total area of each watershed before the model calculated the amount of deforestation or reforestation that should take place in each zone.

Analyses

Vegetation cover

For each scenario, I calculated the amount of remnant native forest and cerrado vegetation remaining. I also calculated the amount of restored vegetation of each type that would be necessary to meet the requirements of each scenario. “Restored” vegetation also includes areas that may be left to regenerate naturally; however, throughout the chapter I will refer to restored vegetation to identify this category of vegetation.

Economic aspects

To evaluate the economic aspects related to compliance with each of three policy alternatives in the forest biome in comparison with the current date, I calculated the difference in the area of the watershed outside of federal and state protected areas (1) available for agricultural production, and (2) that would need to be restored to come into compliance under each scenario. For each, I present the total area for the entire headwaters region as well as the mean per microbasin. Based on these figures, I

estimated (1) the net present value (NPV) of agricultural activities carried out on legally cleared lands, and (2) the cost of riparian forest restoration over the whole landscape and by microbasin, where necessary (both methods are described in detail in Appendix C).

Ecological consequences

I compared the final landscapes for each of the 4 alternative landscapes in terms of carbon stocks, river discharge, annual evapotranspiration, terrestrial habitat quality, and water quality (methods described in Appendix C). Unlike all the preceding analyses, ecological consequences were aggregated for the entire headwaters region, including all protected areas.

Results

Remnant Forest and Cerrado

The impacts of the policy scenarios analyzed here are best understood in light of the influence of each scenario on mandatory forest cover at the level of the microbasin. The areal coverage of each ecosystem (forest *versus* cerrado, remnant vegetation *versus* restored/regenerated vegetation), the potential area and associated economic value of land cleared for agriculture, and the ecological integrity of the Xingu River headwaters region, all varied as a function of the microbasin-level legal reserve requirement. The most restrictive scenario is the current forest code without compensation of the legal forest reserve (CFC-NC), which requires that every microbasin outside of the protected areas maintain 80% forest cover or 35% cerrado cover. This policy scenario allows the clearing of approximately 6,500 km^2 of extant forest and a similar area of cerrado vegetation present in the 2005 landscape that exceeded the microbasin-level legal reserve requirement representing a 17% reduction

below the 2005 landscape (Table 3-1, Figure 3-1). The CFC-C scenario, which imposes the Forest Code but allows microbasins to achieve the legal reserve requirement through compensation in other microbasins, is virtually the same as the CFC-NC scenario since there was very little forest or cerrado vegetation available for compensation. This is somewhat contrary to the expectation that remaining native vegetation would be substantially greater in the CFC-C scenario than the CFC-NC scenario. The ZSEE (ecological-economic zoning) scenario lowered the mandatory forest legal reserve to 50% in some regions, but only decreased the area of remnant vegetation remaining by 1,400 km^2 relative to the CFC-NC and CFC-C scenarios. In other words, the ZSEE lowered the microbasin-level legal reserve requirement to 50% in some regions, but only 1,400 km^2 of forest were in excess of 50% forest cover in these zones (Table 3-1, Figure 3-1).

Restored Forest and Cerrado

All three policy scenarios allowed some of the remnant vegetation present in 2005 to be cleared, but they also required the regeneration/restoration of a greater area. Seventeen thousand square kilometers of restored/regenerated forest are required under these scenarios compared with 11,000 km^2 under the Ecological-Economic Zoning scenario (Table 3-2). It is largely because of this requirement for regenerated/restored forest and cerrado that the total cover of natural forests and cerrado was greatest in the CFC-C and CFC-NC scenarios with approximately 95,000 km^2 (70% of the total area outside of protected areas in the Xingu River headwaters). The ZSEE scenario, which permits some microbasins to fall to 50% forest cover, had only 3500 km^2 (3%) less vegetation cover (Figure 3-2, Table 3-1). Hence, mean native

vegetation cover per microbasin was greatest in the CFC scenarios3 (6scenarios (8 km^2) and only half that in the ZSEE scenario (4 km^2; Table 3-2).

Native Vegetation Cover

Native vegetation cover outside of protected areas and outside of the riparian zone was highest in the current (2005) landscape, in both the forest and *cerrado biomes,* owing to the fact that some microbasins could still legally clear vegetation under all three of the alternative policy scenarios. Among those, however, the ZSEE scenario had the highest amount of native forest remaining, 8498 km^2 (11%) less than in 2005 (Table 3-2). In contrast, both the CFC-NC and CFC-C scenarios had 17% less native vegetation than the current landscape. At the individual microbasin level, native vegetation cover ranged between 28 km^2 and 34 per microbasin across the 4 landscapes (Table 3-1). Of the alternative land-use policy scenarios, the ZSEE scenario again had the highest mean vegetation cover per microbasin, after 2005, with 30 km^2.

Potential Agricultural Area

The potential agricultural area associated with each scenario depends upon the microbasin-level requirements for natural vegetation cover and the modeled clearing of native vegetation across the region. Land that is legally cleared, or that could be legally cleared for potential agricultural production was highest in 2005. The ZSEE scenario had only 1000 km^2 less potential agricultural area than the 2005 landscape despite restoring nearly 10,000 km^2 of vegetation throughout the landscape (Table 3-3, 3-5). The ZSEE scenario had 4000 km^2 less area available within the forest biome than in 2005, but nearly 3000 km^2 more in the cerrado biome. But lowering the legal reserve requirement in many microbasins of the forest biome, the ZSEE permits approximately 7,000 km^2 more forestland to go to agriculture than the CFC scenarios, but allows 3,000

km^2 less cerrado vegetation to be converted to agriculture (Table 3-4). This biome-specific influence of the ZSEE can be explained on the basis of the reduced demand for forest restoration in the ZSEE scenario. Microbasins in Zone 2 that had cleared more than 20% of their forests are not required to reforest back up to 80% coverage in the ZSEE as they are in the CFC scenarios. There was less cerrado conversion because the 3 biophysical restrictions included in the ZSEE protected areas in the cerrado biome that were not protected in the CFC scenarios. In addition, requiring less reforestation in Zone 2 prevented clearing from being displaced to the cerrado biome as extensively as in the CFC scenarios. The CFC-NC and CFC-C scenarios had the least amount of area legally available for agricultural production, approximately 41,000 km^2 (Table 3-3). These rankings were maintained at the microbasin level, with the ZSEE scenario allowing an average of 20 km^2 per microbasin of legally cleared lands, compared with 21 km^2 in the current landscape and 18.5 km^2 in the CFC-NC and CFC-C landscapes.

Potential Net Present Value

The difference in area that could be legally cleared for agricultural activities between the ZSEE and 2005 landscapes represented a decrease of 1.6 billion dollars over the entire region (Table 3-4). In contrast, the CFC-NC and CFC-C scenarios would reduce potential NPV in the region by 4 billion USD from the current landscape. Hence, the ZSEE reduces the opportunity cost of forest conservation by 2.4 billion dollars. For the individual landholder, the average potential NPV per microbasin under the ZSEE scenario would be 0.7 million USD lower than in the 2005 landscape and 1.2 million USD more than the CFC-NC and CFC-C landscapes (Table 3-4).

Costs of Restoring Forest and Cerrado

Restoration costs were highest in the CFC-NC and CFC-C scenarios, totaling just over 2000 million USD for the entire region, 1.6 times more than for the ZSEE scenario (Table 3-6). For the average individual landholder, the restoration burden was lowest both in terms of area and cost in the ZSEE scenario, as well. The mean area of legal reserve to be restored in the ZSEE scenario (4.4 km^2) is approximately half that of the other two policy scenarios (approximately 8 km^2), with the bulk of restoration needing to occur in the forest biome. The mean cost of restoration (for legal reserve and riparian zone restoration combined) was approximately 0.8 million USD per microbasin in the ZSEE scenario, approximately one-third less than in the CFC-C and CFC-NC scenarios (Table 3-6).

Overall, when combined with the loss in potential NPV, restoration costs doubled the cost of adopting the ZSEE scenario compared to the current (2005) scenario, reaching 1.5 million USD. Under the CFC-NC and CFC-C scenarios, restoration increased the average cost to the landholder to 3.2 and 3.1 million USD. The total cost of legal compliance under the CFC-C AND CFC-NC scenarios, combining the opportunity costs (Table 3-5) and the increase in restoration costs relative to the current (2005) landscape (Table 3-6) was approximately USD 6620 million for the entire region and averaged USD 3.1 million per microbasin.

Ecological Consequences

Carbon stocks

Contrary to expectations, the ZSEE scenario achieved higher overall emissions reductions than either the CFC-NC or CFC-C scenarios when each is compared to the 2005 (current) landscape (only 33 MtC emitted vs. 47 MtC) (Table 3-7). This can

primarily be attributed to the greater area of forest and *cerrado* woodland savanna that is maintained or restored under the former scenario in areas defined as restricted by the zoning plan, despite also relaxing the requirements of the Forest Code in 60% of the area outside protected areas. Although existing protected areas contain the highest biomass stands in the region, since both these and the riparian zones are strictly protected under all three alternative policy scenarios, the additional carbon stocks in the *CFC-NC* and *CFC-C* scenarios are entirely attributable to protection and restoration of forest and woodlands on private lands. Restored vegetation contributed an estimated 23 MtC to the total carbon stock in both the CFC-C and CFC-NC landscapes over a 30-year time period, about 1.6 times more than in the ZSEE landscape (Table 3-7).

Hydrology and regional climate

All three scenarios show decreases in stream discharge in comparison with the 2005 landscape, ranging from 2 to 3% less than that of the current landscape (Table 3-7). The greatest decrease in discharge occurs in the CFC-C and CFC-NC scenarios. Mean annual evapotranspiration decreases as forest cover decreases in the 3 alternative policy scenarios, although all exhibit a 4% reduction from the control scenario (Table 3-7). However, all three have increased evapotranspiration by 1% over the current landscape.

Water quality

All three of the alternative policy scenarios had all of the riparian forests protected or restored, thus riparian forest cover increased by approximately 2600 km^2 (20%) over the 2005 landscape (Table 3-7). Thus, in all cases, 20% of streams would be likely to have lower temperatures and higher dissolved oxygen levels (Neill et al. 2006) than the current landscape, affecting species populations and assemblages. The proportion of

microbasins having less than 40% vegetation cover was lowest in the ZSEE scenario (Table 3-7). This indicates that although this scenario did not have the highest proportion of microbasins with 60% or more vegetation cover (both CFC scenarios had more), more microbasins would be likely to maintain basic hydrological functions (Coe et al., 2009).

Habitat

Overall, habitat quality and quantity differed between scenarios by biome. For the forest biome, habitat quality and quantity are lowest in the 2005 landscape, and highest in the *CFC-NC and CFC-C* scenarios (Table 3-7). The CFC-NC and CFC-C landscapes have the highest amount of total forest cover, the smallest number of forest fragments, the greatest mean forest fragment size, as well as the greatest amount of core or interior area (representing 90% of the total forest area in the landscape) (Table 3-7). In contrast, the 2005 landscape compared most favorably for the cerrado biome, with the ZSEE scenario indicating the next best outcome for the biome. Total vegetation, mean fragment size and interior habitat area were greatest in the 2005 landscape. Total edge area was lowest by a factor of 1.5 (in the forest biome) to 1.3 (in the cerrado biome) in the 2005 in comparison with the three alternative policy scenarios. Among the modeled scenarios, the CFC-C had the least amount of edge forest habitat.

Discussion

The hybrid regulatory-economic policy instruments examined in this study differ in their degree of flexibility and responsiveness to frontier dynamics, with important implications for ecological and economic processes in the Xingu River headwaters region. The Brazilian Forest Code provision that allows compensation of the legal reserve among property holders (CFC-C scenario) is intended to reduce the economic

costs of the Forest Code by allowing more agricultural expansion on lands with higher returns from deforestation-dependent agricultural activities, such as soy production and cattle ranching, in exchange for reduced deforestation on lands with lower potential returns from soy and cattle. In the case of the Xingu River headwaters region, however, this provision had virtually no effect on native ecosystem cover when compared with the Forest Code with no compensation (CFC-NC scenario), in part because of the strict eligibility criteria, the small amount of excess forest or cerrado vegetation in microbasins having more than 80% or 35% cover, respectively, and possibly because of restrictions in the model that limited trading to a 4th-order watershed level (Table 3-4). This form of flexibility in land use regulations must be functional early in the evolution of an agricultural frontier to significantly reduce the economic costs of compliance. In the case of the Xingu River headwaters region, there may have also been a strong reluctance on the part of landholders to use the compensation provision after it was enacted, in 2000[7], because of the great uncertainty surrounding the legal reserve requirement itself. To use the compensation provision, landholders must alter their land titles to forgo development rights on all remaining forests or cerrado on their property as they enter into a financial compensation contract with another property holder. These restrictions become unpalatable for many when there is a perceived likelihood that the legal forest reserve requirement could be reduced permanently, especially when severe forms of punishment for non-compliance can be avoided through bribes (Rosenthal, 2009).

The Brazilian socio-economic and ecological zoning policy (ZSEE) is a far more sophisticated instrument than the legal reserve compensation provision in that it

[7] The provision was included in the revised Forest Code in late 1998, but landholders were identified as eligible depending on their vegetation cover status at the end of 1999.

responds to biophysical and infra-structural (transportation, urban centers, electrical supply) features of landscapes to restrict clearing for agriculture and landscape in areas of marginal suitability and liberate clearing in areas of agricultural consolidation. The modeled ZSEE scenario, using economic rent models that are responsive to the same factors that impinge upon the profitability of agriculture that the ZSEE is designed to represent, demonstrates the potential for this instrument to considerably reduce the economic costs of legal compliance while protecting public interests in private land native ecosystems and their services. The ZSEE reduces the opportunity costs associated with full compliance with the Brazilian Forest Code (with no compensation) from 4.2 to 1.6 billion dollars (Table 3-5), primarily by allowing a greater share of cleared lands that are non-compliant with the 80% or 35% legal reserve requirement to stay in production. More specifically, in Zone 2 of the ZSEE scenario, microbasins that have cleared more than 80% of forest cover or 35% of cerrado cover are not required to reforest (or compensate) the difference. For the average landholder, this means that the net present value of a landholding is worth 1.2 million dollars more than under the CFC scenarios, and only 0.7 million dollars less than in the current (2005) landscape (Table 3-5).

The ZSEE also has features that improve the ecological performance of the landscape in ways that are not captured by parameters such as the total cover of native vegetation. For example, agriculturally marginal lands, with steep slopes and high erodibility or that are located in wetland or floodplain areas, are protected under the ZSEE. In the Xingu River headwaters region, this feature of ZSEE is manifested as a greater coverage of cerrado vegetation. The ZSEE protects 3,100 km^2 more remnant

cerrado vegetation than the CFC scenarios, largely because of their marginal suitability for agriculture (Table 3-1). By most ecological parameters, the ZSEE scenario is nearly indistinguishable from the CFC scenarios, despite its far lower opportunity costs. The ZSEE scenario has the same river discharge, evapotranspiration, and nearly the same number of microbasins with at least 60% native ecosystem coverage (75% vs. 80% and 81% for the CFC scenarios). The ZSEE scenario has smaller carbon stocks (1924 vs. 1909 and 1907) than the CFC scenario landscapes, but less interior habitat (99,000 km^2 vs. 105,000 km^2) than the CFC scenario landscapes.

Conclusion

Hybrid regulatory-economic policy instruments may be particularly appropriate for achieving land-based economic and ecological outcomes. Unlike pollution regulation, where environmental impacts are lowered by reducing emissions of a pollutant regardless of the exact location of this emission, ecosystem services are best protected through instruments that guarantee that these services will be spatially distributed. This is a positive feature of the Forest Code, that requires that all riparian zone forests be protected and a portion of the native ecosystems of every microbasin. The economic component of one of the hybrid instruments that I evaluated, the ZSEE, lowered the cost to landholders and to secondary and tertiary industries that depend upon agricultural production, with only a small sacrifice of ecological performance. Albeit not a perfect solution to balancing agricultural productivity with environmental sustainability, in this landscape it may be among the optimal policy frameworks.

Table 3-1. Total area and mean area per microbasin of remnant forest and cerrado woodland within the Xingu River headwaters region in 2005 and under 3 land-use policy scenarios. Data are for microbasins outside of protected areas. “CFC-NC” is the abbreviation for “Current Forest Code, No Compensation”. “CFC-C” is the abbreviation for “Current Forest Code with Compensation”. “ZSEE” is the abbreviation for “Zoneamento Socio-Economico Ecologico” (Socioeconomic Ecological Zoning).

	2005		CFC-NC		CFC-C		ZSEE	
	Total (km^2)	Mean (*s.e.*) per microbasin (km^2)	Total (km^2)	Mean (*s.e.*) per microbasin (km^2)	Total (km^2)	Mean (*s.e.*) per microbasin (km^2)	Total (km^2)	Mean (*s.e.*) per microbasin (km^2)
Remnant native vegetation								
Forest	61,934	35.3 (0.8)	55,416	31.6 (0.7)	55,410	31.6 (0.7)	56,882	32.4 (0.8)
Cerrado	13,065	28.5 (1.5)	6547	14.3 (0.8)	6558	14.3 (0.8)	9619	21.0 (1.1)
TOTAL	74,999	33.9 (0.7)	61,963	28.0 (0.6)	61,968	28.0 (0.6)	66,501	30.0 (0.6)

Table 3-2. Total area and mean area per microbasin of forest and cerrado within the Xingu River headwaters region that will consist of restored vegetation in 2005 and under 3 land-use policy scenarios. Data are for microbasins outside of protected areas. Scenario abbreviations are as in Table 3-1.

	CFC-NC		CFC-C		ZSEE	
Restored Vegetation	Total (km^2)	Mean (*s.e.*) per microbasin (km^2)	Total (km^2)	Mean (*s.e.*) per microbasin (km^2)	Total (km^2)	Mean (*s.e.*) per microbasin (km^2)
Legal Reserve						
Forest	17,101	9.7 (0.4)	17,235	9.8 (0.4)	9090	5.2 (0.3)
Cerrado	722	1.6 (0.2)	687	1.5 (0.3)	565	1.2 (0.1)
TOTAL	17,823	8.0 (0.3)	17,922	8.1 (0.3)	9654	4.4 (0.2)
Riparian Zone						
Forest	1666	1.0 (0.04)	1666	1.0 (0.04)	1666	1.0 (0.04)
Cerrado	835	1.9 (0.1)	835	1.8 (0.1)	835	1.8 (0.1)
TOTAL	2501	0.9 (0.03)	2501	0.9 (0.03)	2501	0.9 (0.03)

Table 3-3. The area and percent coverage of forest and cerrado vegetation (remnant and restored combined) in the Xingu River headwaters region as observed for 2005 and modeled under three land-use policy scenarios. Data are presented both for the entire headwaters region and for the mean values of the region's microbasins. Results are for lands outside of protected areas. Scenario abbreviations are as in Table 3-1.

	2005		CFC-NC		CFC-C		ZSEE	
	Area (km^2)	%	Area (km^2)	%	Area (km^2)	%	Area (km^2)	%
Total Natural Vegetation (remnant and restored)								
Whole watershed (Total)								
Legal Reserve								
Forest	61,934	64	75,590	80	75,718	80	69,044	71
Cerrado	13,065	45	9265	35	9241	35	12,194	42
Total	74,999	60	84,855	68	84,959	67	81,138	64
Riparian Zone								
Forest	6052	78	7736	100	7736	100	7736	100
Cerrado	1695	67	2530	100	2530	100	2530	100
Total	7747	76	10,266	100	10,266	100	10,266	100
Legal Reserve and Riparian Zone	82,746	60	95,121	69	95,225	70	91,404	67
Microbasins (Mean (s.e.))								
Legal Reserve								
Forest	35 (0.8)	68 (0.6)	43 (0.9)	80 (0.3)	43 (0.9)	78 (0.3)	39 (0.8)	74 (0.4)
Cerrado	28 (1.5)	49 (1.2)	20 (1.1)	35 (0.8)	20 (1.1)	33 (0.8)	27 (1.3)	45 (1.0)
Riparian Zone								
Forest	3.5 (0.1)	79 (0.5)	4.4 (0.1)	100 (0)	4.4 (0.1)	100 (0)	4.4 (0.1)	100 (0)
Cerrado	3.7 (0.2)	69 (0.9)	5.6 (0.3)	100 (0)	5.6 (0.3)	100 (0)	5.6 (0.3)	100 (0)
Total	3.5 (0.1)	77 (0.5)	4.7 (0.1)	100 (0)	4.7 (0.1)	100 (0)	4.6 (0.1)	100 (0)

Table 3-4. Total area and mean area per microbasin of forest and cerrado within the Xingu River headwaters region that is "available" for agricultural production in 2005 and in 3 land-use policy scenarios. Data are for microbasins outside of protected areas. "Available for agricultural production" refers to areas that already are or could be legally cleared without exceeding the maximum percent clearing allowable under each scenario.

	2005		CFC-NC		CFC-C		ZSEE	
Area available for agriculture	Total (km^2)	Mean (*s.e.*) per microbasin (km^2)	Total (km^2)	Mean (*s.e.*) per microbasin (km^2)	Total (km^2)	Mean (*s.e.*) per microbasin (km^2)	Total (km^2)	Mean (*s.e.*) per microbasin (km^2)
Forest	31,419	17.9 (0.6)	20,812	11.8 (0.3)	20,881	11.9 (0.3)	27,429	15.6 (0.5)
Cerrado	14,110	30.8 (2.0)	19,931	43.5 (2.2)	20,082	43.8 (2.2)	17,003	37.1 (2.2)
TOTAL	45,529	20.6 (0.6)	40,743	18.4 (0.6)	40,963	18.5 (0.6)	44,432	20.0 (0.6)

Table 3-5. Potential net present value (NPV) associated with soybean cultivation or cattle production on cleared land for the Xingu River headwaters region under 3 alternative policy scenarios and in 2005. The opportunity cost of adopting each scenario (relative to the 2005 landscape) is also presented. Data are presented for the entire region and per microbasin, for areas outside of protected areas in the forest biome region.

	2005 NPV (million USD)	CFC-NC NPV (million USD)	CFC-C NPV (million USD)	ZSEE NPV (million USD)
Potential NPV				
Whole basin	13,755	9522	9529	12,104
Mean per microbasin (s.e.)	6.2 (0.3)	4.3 (0.2)	4.3 (0.2)	5.5 (0.2)
Opportunity Cost				
Whole basin	n/a	4233	4228	1651
Mean per microbasin (s.e.)	n/a	1.9 (0.2)	1.9 (0.2)	0.7 (0.1)

Table 3-6. The estimated costs of restoration of the legal reserve and riparian zone under 3 alternative policies, for the entire basin in the aggregate and for the average microbasin out of compliance. Ranges refer to upper and lower estimates of restoration costs per hectare, as defined in Table C-1.

	CFC-NC *Whole Basin* (million USD)	CFC-C *Whole Basin* (million USD)	ZSEE *Whole Basin* (million USD)
Legal Reserve			
Forest	1593 (+/676)	1605 (±682)	847 (±359)
Cerrado	67 (±29)	64 (±27)	53 (±22)
Riparian Zone	366 (±261)	366(±261)	366(±261)
Total	2026 (±966)	2035 (±970)	1266 (±642)
	Mean per microbasin (million USD)	*Mean per microbasin* (million USD)	*Mean per microbasin* (million USD)
Legal Reserve			
Forest	0.9 (±0.4)	0.9 (±0.4)	0.5 (±0.2)
Cerrado	0.2 (±0.1)	0.1 (±0.1)	0.1 (±0.1)
Riparian Zone	0.2 (±0.1)	0.2 (±0.1)	0.2 (±0.1)
Total	1.3 (0.6)	1.2 (±0.6)	0.8 (±0.4)

Table 3-7. Ecological attributes of the Xingu River headwaters region (including protected areas) in 2005 and of modeled landscapes representing the region under 3 policy alternatives on private lands. These attributes include: carbon stocks, surface hydrology and regional climate, indicators related to water quality, and terrestrial habitat quantity and quality.

Indicator	*Scenarios*			
	2005	CFC-NC	CFC-C	ZSEE
Carbon Stocks				
Stored in native vegetation				
(MtC)	544	497	497	511
($MtCO_2e$)	1995	1824	1822	1873
Stored in restored vegetation (30 years)				
(MtC)	5	23	23	14
($MtCO_2e$)	17	85	85	51
Total				
(MtC)	549	520	520	525
($MtCO_2e$)	2012	1909	1907	1924
Emissions since initial year				
(MtC)	n/a	47	47	33
($MtCO_2e$)	n/a	171	173	122
Surface Hydrology and Regional Climate				
Mean Annual Discharge ($m^3 s^{-1}$)	3128	3032	3033	3070
(% change from potential)	(13%)	(10%)	(10%)	(11%)
Mean Annual Evapotranspiration ($m^3 s^{-1}$)	6583	6679	6678	6641
(% change from potential)	(-5%)	(-4%)	(-4%)	(-4%)
Water Quality				
Riparian forest cover, 50-m buffer (km^2)	12,931	15,509	15,509	15,509
Mean % native vegetation cover per microbasin (s.e.)	69 (0.5)	73 (0.4)	73 (0.4)	73 (0.4)
% of microbasins with greater than 60% vegetation cover	65	80	81	75
% of microbasins with less than 40% vegetation cover	18	16	16	11
Terrestrial Habitat				
Vegetation cover (km^2)				
Forest	107,789	116,487	116,395	110,902
Cerrado	17,037	13,297	13,149	15,209
Number of fragments				
Forest	13,427	12,666	12,958	13,777
Cerrado	13,285	18,238	18,138	15,571
Mean distance to nearest neighbor fragment (m)				
Forest	361	381	379	363
Cerrado	406	371	371	378
Mean fragment size (ha)				
Forest	803	920	898	805
Cerrado	128	73	73	98
Total interior habitat area (km^2)				
Forest	99,978	104,754	105,287	98,886
Cerrado	12,737	7652	7433	9762
Total edge habitat area (km^2)				
Forest	7810	11,732	11,108	11,969
Cerrado	4300	5645	5716	5472

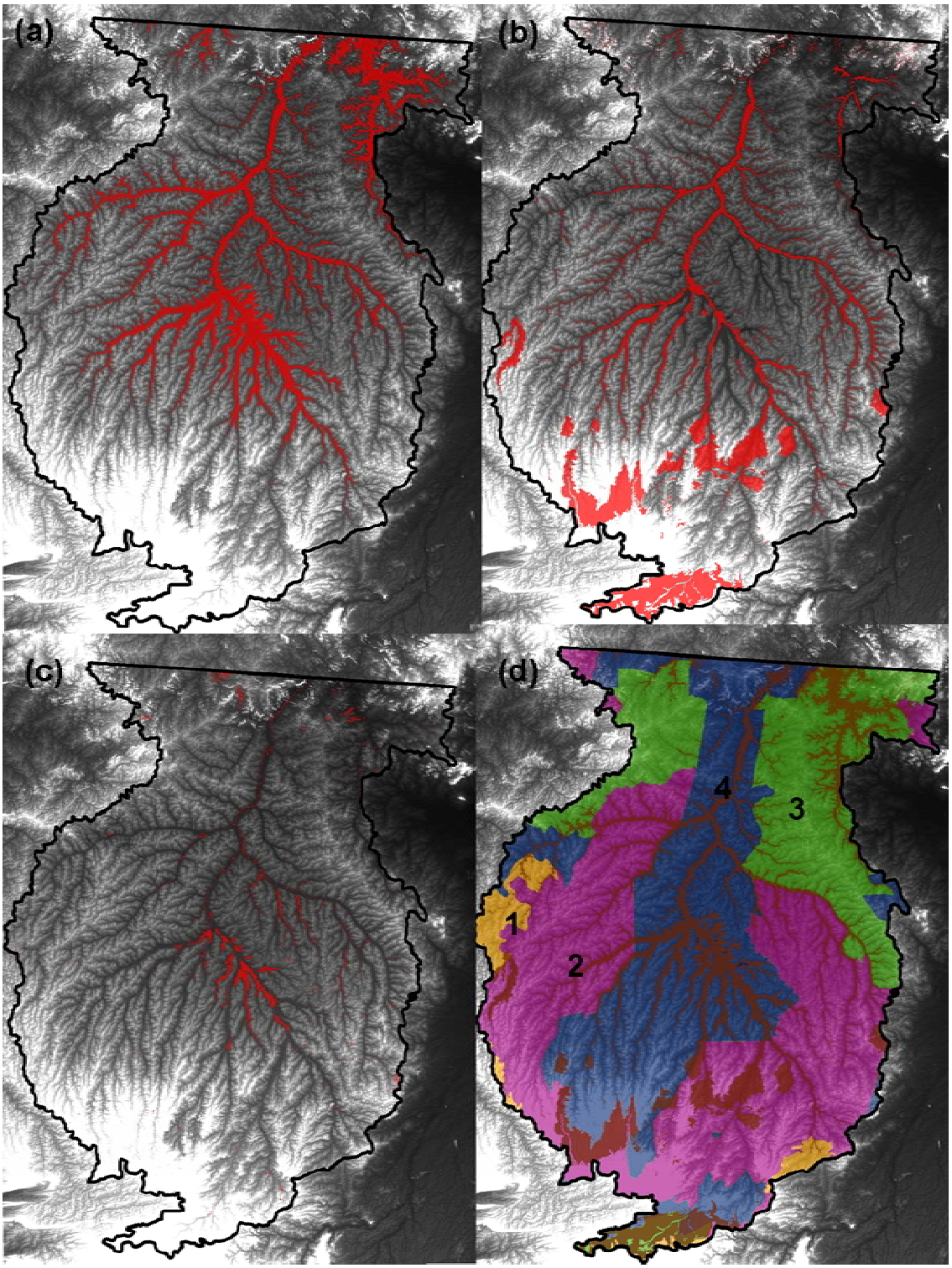

Figure 3-1. Location and distribution of the major biophysical restrictions (shown in red) and the 4 major zones of the Mato Grosso State Zoning Plan (ZSEE) within the Xingu River headwaters: (a) floodplain areas; (b) soil and slope restrictions; (c) wetland areas; (d) the 4 major zones (noted by number) combined with the 3 biophysical restrictions.

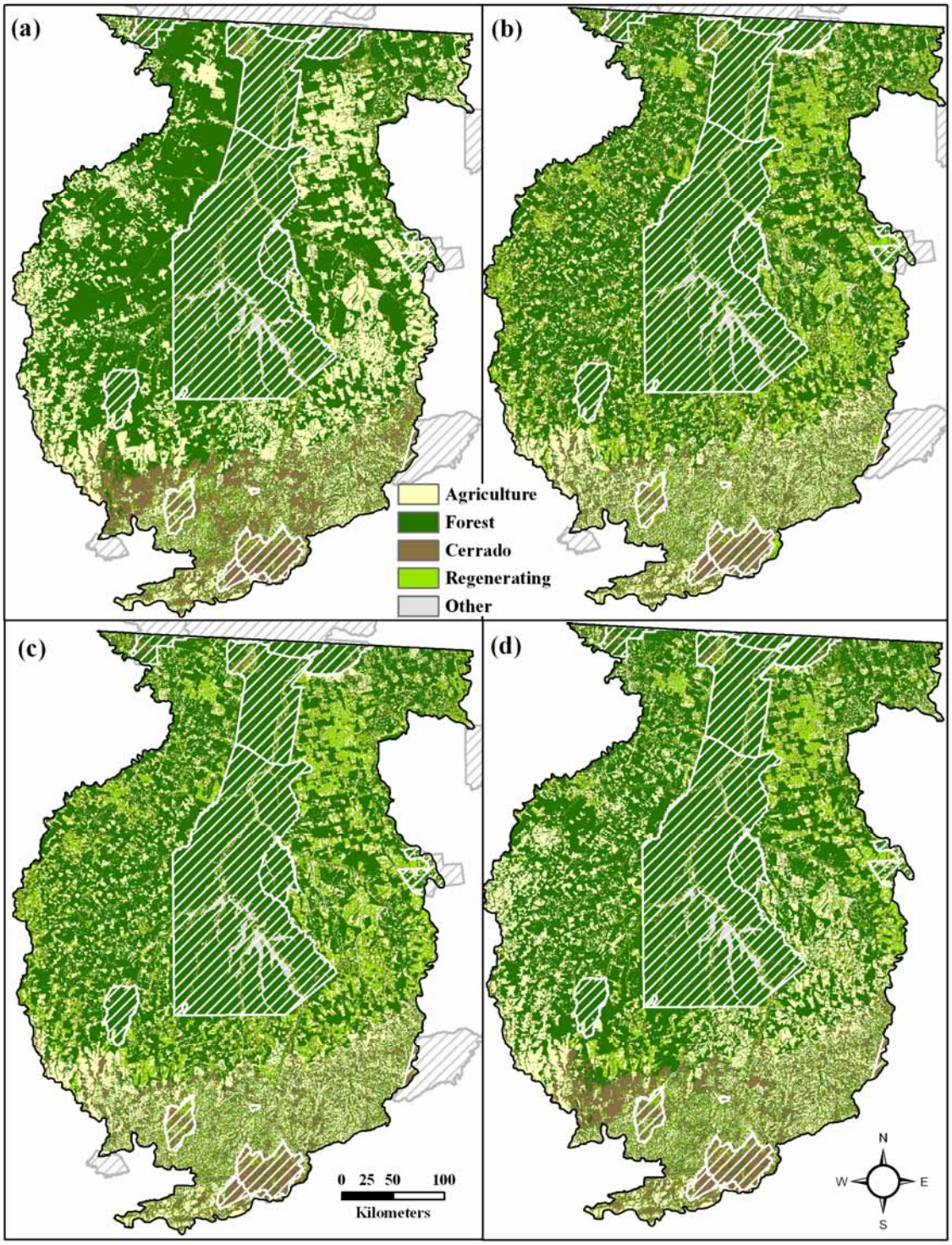

Figure 3-2. Comparison of land-cover in the basin for (a) 2005 (observed), (b) full compliance with 80% legal reserve in forest biome, no compensation (CFC-NC) (modeled), (c) full compliance with 80% legal reserve in the forest biome with compensation (CFC-C) (modeled), and (d) Mato Grosso state socio-economic ecological zoning plan (ZSEE) (modeled), with compensation.

CHAPTER 4
THE COST OF CARBON TO OFFSET OPPORTUNITY COSTS ON PRIVATE LANDS UNDER ALTERNATIVE POLICIES IN THE UPPER XINGU RIVER BASIN

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 tropical nations that succeed in reducing carbon emissions from deforestation and forest degradation, known by the acronym “REDD” (from Reducing Emissions from Deforestation and Degradation; Gullison et al., 2007). REDD has the potential to both increase the scale of tropical forest conservation and significantly reduce greenhouse gas emissions. REDD could also become the vehicle for protecting and restoring the role of tropical forests in the provision and regulation of pure freshwater, biodiversity conservation, soil conservation, and the protection of regional climate systems (Stickler et al., 2009), channeling financial resources from either a new forest carbon market or from expanded carbon-related donations from developed countries.

As negotiations of the REDD mechanism within the UNFCCC framework move toward fruition, federal and state governments of tropical nations are developing programs to achieve reductions in deforestation and forest degradation. Brazil has been developing a program to reduce emissions from deforestation in the Amazon—the country’s largest source of GHG emissions (76%, 1.2 billion tons CO_2e; MCT, 2009). Under the Brazilian National Climate Policy, the government established a target of 80% reduction in deforestation in the Amazon region by 2020 in 2009 (GOB 2008; Nepstad et al., 2009). With a USD 1 billion commitment from the Norwegian government, Brazil

has already established a fund to finance projects and programs for achieving these emissions reductions (BNDES, 2009). The Amazon Fund, as negotiated with Norway, would be replenished as long as Brazil continues to reduce deforestation in the Amazon region below its historical baseline. The initial payment of USD 110 million from Norway was in recognition of deforestation reductions already achieved since 2005. Though not officially considered a REDD program by the Brazilian government, for all intents and purposes, it serves as the first regional- or national-scale REDD experiment in the world. However, the Amazon Fund does not have a well-developed plan for achieving reductions in deforestation and forest degradation in the Amazon region.

REDD land use policy design is advancing within the Amazon states of Brazil. Under the Brazilian National Climate Policy, Amazon state governments are expected to establish their own deforestation reduction targets that, combined, provide the requisite reductions for the entire region (Tollefson, 2009). Mato Grosso state has historically been responsible for the greatest amount of deforestation (39% of annual regional average for 1996-2005 time period; INPE, 2009), but from 2006-2009, it has been responsible for 59% of deforestation reduction already achieved under the BNCP goals over the last 4 years (INPE, 2009; GOMT, 2009). Furthermore, in its own plan, Mato Grosso is committing to 89% reduction in deforestation by 2020, 9% more than the national plan. This target represents more than 60% of the national target for deforestation reduction in the Amazon, and approximately 40% of Brazil's total goal of GHG emission reduction by 2020.

According to the state's draft REDD plan, the implementation of REDD is considered vital to achieve Mato Grosso's deforestation reduction targets to finance

credits for emissions reductions financed by the Amazon Fund and other voluntary and potential official market mechanisms. The emissions reductions effectively achieved during a given period of time are intended to provide financing for future deforestation reductions. “REDD Certificates” are proposed to be allocated between six programs for (1) indigenous peoples; (2) protected areas; (3) private forests; (4) smallholder settlements; (5) state governance ; and (6) an insurance fund. During the first phase of implementation, indigenous lands, protected areas, and smallholder settlements will participate in the mechanism through priority pilot projects that will help to develop adequate approaches and methodologies for state-wide programs. The private forests program, linked to the State environmental licensing system of rural properties (Azevedo, 2009), will provide payments for forest reserves located in rural properties that fully comply with the federal and state environmental laws.

In this paper, I attempt to provide input to the policy framework design of the Mato Grosso REDD program, specifically to the design of the program to reduce deforestation on private lands. I examine the extent to which existing forest and land-use policies might be used as a basis for effectively applying REDD or other carbon credits in the upper Xingu River basin in northeastern Mato Grosso—one of 3 pilot sites for designing and testing the state level program (GOMT, 2009). I estimate the emissions reductions below (a) a historical baseline and (b) 3 BAU scenarios that could be achieved on private lands in the region under 3 alternative policy frameworks. I also estimate the price of carbon that would be needed to offset the opportunity costs incurred by private landholders of complying with each alternative framework used to achieve target (portion coming from private lands). Finally, I compare the alternative

scenarios with respect to a series of ecological indicators to evaluate the performance of each in providing co-benefits beyond carbon stocks.

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) (Figure 2-1a). 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 (Figure 2-1a). The Basin supports *cerrado* (savanna woodland) in the south and dense humid forest in the north (Figure 2-1a).

The Xingu region is also an advanced laboratory for exploring the potential ecological co-benefits of alternative approaches to REDD plans. 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 multi-stakeholder 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

I compared 3 alternative land-use policies that represent possible approaches to the implementation of emissions reductions targets over 30 years in the Xingu headwaters region. I developed land-cover maps representing each of the alternative

scenarios, as well as 3 business-as-usual (BAU) scenarios. I used these maps to estimate and compare the volume of carbon stocks that would be maintained under each scenario, in both remaining native and restored or regenerating forest or *cerrado*. I then used the maps to calculate the cost of (1) reducing carbon emissions, and (2) restoring or enhancing carbon stocks under compliance with each of the 3 alternative policy scenarios relative to each of three business-as-usual (BAU) scenarios. Finally, the maps were also used as inputs for calculating several ecological indicators as a basis for assessing the potential ecological co-benefits of REDD.

Model Development

Six land cover maps corresponding to three alternative policy scenarios and three business-as-usual (BAU) scenarios were developed using a spatially explicit dynamic landscape simulation model described in Appendix A. This spatial-statistical model of land-use change 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 4 focal land-use transitions: (1) forest→agriculture (pasture or annual crops); (2) *cerrado*→agriculture; (3) agriculture→regenerating forest; and (4) agriculture→regenerating *cerrado*. For the BAU scenarios, the model simulates land-cover change over 30 time steps, beginning in 2005 and ending in 2035, using land-cover conversion probabilities and rates calculated from the 1996-2005 reference period (Appendix A). “High” and “low” BAU scenarios were developed in an attempt to bracket the range of likely future deforestation by applying the highest and lowest historical observed deforestation rates, respectively, as described below. For the policy scenarios, maximum required reforestation and maximum allowed clearing were imposed upon the Xingu River headwaters region

landscape, independent of the amount of time required to reach these levels. The basic architecture and function of the simulation model is described in further detail in Appendix A.

Scenarios

Each of the scenarios is based on recent, existing and/or proposed legislation (Chapter 2, Chapter 3) and was compared with a range of BAU simulations that assume no REDD interventions. The basic assumptions underlying each scenario, including the reference scenarios, are as follows:

Business-as-usual scenarios

Business as Usual (BAU) assumes that the historical rate and pattern of deforestation continues into the future, and thus serves as a baseline model against which to compare other options. Here, I develop 3 BAU scenarios that represent alternative development trajectories with no additional REDD or other governance interventions (e.g., creating or enforcing protected areas). The average BAU scenario uses the average deforestation rate calculated for this region over the 1996-2005 period and applies it over 30 years. The reference period corresponds to that set for determining crediting levels for the Amazon Fund (GOB 2009). Under the Amazon Fund, the reference scenario is estimated by extending the average rate for the 10-year period from 1996-2005 as an absolute (gross) rate into the future. In the analysis presented here, the BAU scenario is somewhat more conservative, applying the same annual rate of clearing as a percentage (net rate) of the remaining forest; thus the absolute amount cleared each year decreases proportionally with the decrease in total forest cover. This is a more realistic reference scenario for a region such as the Xingu River headwaters, as it has historically high levels of deforestation which are unlikely to

be sustained at the same absolute level in the future. Under the Amazon Fund, the reference scenario is extended only through 2020; here, however, extend the simulation through 2035 to provide an assessment of how policies aided by carbon offset to create incentives might protect ecological resources in the longer term.

I also model two further BAU scenarios—*BAU High* and *BAU Low*—using a higher and lower annual deforestation rate, respectively, than the original BAU uses. The scenarios use deforestation rates observed in the region for the 2005-2007 and the 2001-2003 periods, respectively, to provide a range of reference values that reflect business-as-usual under alternative economic conditions. These periods represent the periods of highest and lowest observed deforestation rates in the region to date and thus represent realistic endmembers for attempting to bracket historic land-cover change patterns into the future.

Policy scenarios

I modeled three scenarios representing the Xingu River headwaters landscape under alternative extant or proposed federal and state policies (Chapter 2, Chapter 3).

The *Current Forest Code (CFC)* scenario represents the landscape under the assumption that the current Federal Forest Code was perfectly implemented and enforced. Since 1996, the Forest Code requires that properties located in the forest biome in the Legal Amazon maintain 80% of the native vegetation in a permanent legal reserve; properties in the *cerrado* biome are required to maintain 35% of the native vegetation in a legal reserve (Chapter 2). Where less than this amount is present, the vegetation must be restored. In addition, vegetation within 50-m of each stream on private properties must be strictly protected or restored if it is absent.

The *Reduced Legal Reserve (RLR)* scenario models the Forest Code under the law before 1996 and as was required by the Mato Grosso state government until as recently as 2005, enforcing a legal reserve of only 50% on properties located in the Amazon forest biome (Chapter 2). The legal reserve on *cerrado* properties remains 35% and the riparian zone is also strictly protected.

The *Socio-Economic Ecological Zoning Plan (ZSEE)* scenario assumes that the proposed Mato Grosso state zoning plan is implemented, as described in Chapter 3. The zoning plan has 4 major zones which determine the percent of legal reserve that is required for properties falling within each zone. The scenario modeled here also assumes that the "tradable deforestation rights" option included in the federal Forest Code in 1998 will be exercised (Chapter 3). Furthermore, the scenario assumes strict protection of areas falling in any one of 3 areas described as requiring special attention and protection under the ZSEE (Chapter 3). Finally, as in the other two scenarios, all riparian areas within 50-m of streams are strictly protected and reforested.

Analyses

Emissions reductions and carbon enhancement

For the initial landscape (2005) and each alternative scenario, I calculated the carbon (CO_2e) stocks stored in remaining native vegetation using a map of forest biomass developed for the entire Amazon basin (Saatchi et al., 2007, adapted in Nepstad et al., 2007a and Nepstad et al., 2009).

To estimate the gain in carbon stocks resulting from reforestation under each alternative policy scenario, I assigned a value of 1.5 tC $ha^{-1}y^{-1}$ and 0.5 tC $ha^{-1}y^{-1}$ (Houghton et al., 2000; Zarin et al., 2001) per pixel, respectively, and multiplied by the

number of years that regeneration in each pixel had taken place (ranging from 1 to 29 years).

I present the distribution of carbon stocks among riparian and non-riparian zones separately. Furthermore, I present regenerating and maintained carbon stocks aggregated over the entire Xingu River headwaters region outside of protected areas.

Thus, I obtained estimates of total carbon emissions, of carbon emissions due to clearing of native vegetation, the potential emissions reductions that could be achieved, and the total carbon enhancement represented by each policy alternative.

Price of carbon

To calculate the minimum price of a ton of carbon (CO_2e, in this case) needed to offset the opportunity cost represented by each of the 3 policy scenarios for private lands in the basin in the aggregate, I divided the difference in total opportunity cost of a given scenario and each of the BAU scenarios by the corresponding difference in carbon stocks in remnant vegetation. The opportunity cost was calculated by estimating the potential net present value (NPV) of forested (native or regenerating) lands maintained under each scenario associated with soy or cattle production (the higher of the two) (Appendix C), and then calculating the difference in potential NPV between each policy scenario and each of the BAU scenarios. I obtained a range of prices per ton of CO_2e to offset the aggregate opportunity cost faced by private landowners under each policy scenario.

In addition, I calculated the price per ton of carbon necessary to compensate private landholders for the costs of restoring lands where restoration is necessary to comply with each alternative policy scenario. The costs of restoration are two-fold: (1) the foregone profits (opportunity cost) of the land taken out of production, and (2) the

direct expenditures for restoration. The direct costs of restoration are likely to be negligible outside riparian areas, as landholders will simply allow the land to regenerate naturally (Chapter 3). However, here, I present the range of potential direct costs associated with active restoration for illustrative purposes. The opportunity cost is calculated as above and the range of direct costs is calculated as described in Appendix C. The per unit price of carbon is calculated by dividing the sum of opportunity costs and direct restoration costs for each scenario by the corresponding volume of carbon accumulated on regenerating lands over a 30 year time horizon. We assume no significant amount of natural regeneration or active restoration occurs in any of the BAU scenarios; thus, the amount of accumulated carbon for each scenario represents the difference between the total observed regeneration for that scenario and the initial volume of regenerating vegetation in 2005.

Eligible credits

Finally, for microbasins located in the forest biome only, where the legal reserve was 50% of private landholdings in the Amazon until 1996 (Chapter 2), I also present the total volume of credits and the necessary price for a restricted portion of carbon stocks which represent the carbon stocks held in forests in excess of 50% of total potential forest in each microbasin. This restriction represents a scenario under which governments limit compensation for the costs of forest maintenance and restoration to those landowners who contribute more to the maintenance of carbon stocks and provision of ecosystem services than the minimum required under the law until 1996 (and until 2005, in Mato Grosso) (Chapter 2). In this case, I assume that the law requires landholders to set aside at least 50% of their land in a forest reserve, for reasons explained in Chapter 2. This restriction addresses concerns both domestically

and internationally that individuals should not be paid to comply with the law, and furthermore, that individuals who violate the law not be compensated for whatever amount of forest they do maintain. The cost of reducing emissions and the price of carbon for only this subset of the total carbon stock in the region were calculated as described above. The per unit price to compensate restoration costs was not included. I present the results for this subset of microbasins in the region at the aggregate (landscape) level.

Assessment of ecological co-benefits

I compared the final landscapes for each of the 3 alternative scenarios and for each BAU scenario (for a total of nine comparisons) in terms of river discharge, annual evapotranspiration, habitat quality, and water quality (methods described in Appendix C).

Results

Emissions Reductions and Carbon Enhancement

The simulated policy scenarios affected land cover in the Xingu River headwater region by imposing restrictions on private-land forest and cerrado clearing and by defining forest/*cerrado* restoration requirements for each privately-held microbasin. The reduced legal reserve (RLR) scenario was the most permissive, allowing forest clearing to proceed to 50% of each microbasin and requiring forest restoration only up to 50% of each microbasin in cases where there was less than 50% forest cover during the base year, 2005. It is therefore not surprising that the RLR scenario resulted in the lowest amount of carbon in remnant native vegetation and restored vegetation of the three policy scenarios for private lands in the region (Table 4-1, Figure 4-1).

The current Forest Code (CFC) scenario was the least permissive policy among those simulated. It restricts forest clearing to 20% of individual microbasins and requires landholders to restore forest cover to 80% if clearing had already exceeded 20% during the base year, 2005. The socio-economic, ecological zoning plan (ZSEE) scenario was the same as the CFC scenario except for one difference: landholders were required to restore forest cover only up to 50%, instead of 80%. As a result of this difference, the CFC scenario had five times more restored forest carbon than the ZSEE and six times more than the RLR (Table 4-1). But the CFC had slightly less remnant native vegetation (1113 $MtCO_2e$) than the ZSEE (1241 $MtCO_2e$) (Table 4-1), because of the additional restricted areas created under the plan, in addition to the regulations defined for each zone (Chapter 3).

The policy scenarios exerted a large impact on forest/*cerrado* carbon stocks in the Xingu River headwaters region. The ZSEE and CFC scenarios maintain the greatest amount of carbon in remnant native vegetation relative to the other modeled scenarios (Table 4-1, Figure 4-1). They maintain approximately 1.5 times as much carbon as the BAU Low and RLR scenario and 2 to 5 times more than the BAU and BAU High scenarios. Notably, more native carbon stocks are maintained (not emitted) under the BAU Low scenario than the RLR scenario. Under the worst case BAU scenario, BAU High, the least amount of carbon stocks were maintained, approximately 5 times less than either the ZSEE or CFC scenarios. In contrast, only a 2 to 3-fold difference in carbon stocks was observed between the average BAU scenario and the CFC and ZSEE scenarios.

Price of Carbon

All carbon

The price per ton of CO_2e necessary to compensate all private property holders in the region for maintaining or restoring carbon stocks over and above the amount that would be maintained under the BAU scenario was greatest for the RLR scenario (Table 4-2). The price ranged from 25 to 54 USD tCO_2e^{-1} (average 32 USD tCO_2e^{-1}), whereas under the CFC and ZSEE scenarios, the range of prices was both lower and narrower (22-26 and 22-28 USD tCO_2e^{-1}, respectively). However, the total cost of maintaining and restoring forests to reach compliance with each of the latter scenarios was 2 to nearly 3 times higher, respectively, than for the RLR scenario (Table 4-2). Removing compensation for the cost (both opportunity and direct costs) of restoration reduces the per ton price substantially, in the case of the RLR scenario, with prices ranging from 18 to 21 USD tCO_2e^{-1}. Again, the range of prices for the other two scenarios is narrower and slightly lower, ranging around 18 and 17 USD tCO_2e^{-1} for the CFC and ZSEE scenarios, respectively. Under the RLR and ZSEE scenarios, compensating for regeneration is expensive (the per unit price ranges around 102 and 103 USD tCO_2e^{-1}, respectively), as the amount of carbon accumulated is relatively low and the costs are high (including both foregone potential income and the direct cost of restoration). In contrast, under the CFC scenario, where far more restoration is required, something of an economy of scale is achieved, and the price falls to a far more reasonable 36 USD tCO_2e^{-1}. This surprising result can be explained as the result of higher carbon accumulation in restored forests under this scenario and lower opportunity costs. In other words, this scenario forces the restoration of a large area of land that once

supported forests with high carbon density and that has low potential profitability for soy and cattle ranching.

Eligible carbon

When compensation for carbon maintenance (reduced emissions) is restricted only to that portion of forest that exceeds 50% of the total possible native forest in each microbasin, a 3-fold reduction in per unit prices for CO_2e is observed (Table 4-3). Prices for compensating landholders for their "excess" carbon maintenance under the RLR scenario range from 11 to 12 USD tCO_2e^{-1}, compared with 15-16 USD tCO_2e^{-1} and 14-15 USD tCO_2e^{-1} under the CFC and ZSEE scenarios, respectively.

Ecological co-benefits

Hydrology and regional climate. Different changes in forest cover caused by each of the three policy scenarios resulted in corresponding changes in key ecological parameters. The CFC and ZSEE scenarios induced twice as much forest cover as the average BAU scenario and therefore had far more evapotranspiration (approximately 300 mm yr^{-1}, Table 4-4). With more evapotranspiration, the two high-forest-cover scenarios allowed less river discharge than any of the other scenarios that I modeled (Table 4-4). The RLR scenario, which induced forest cover intermediate to the BAU-high and BAU-average scenarios, had correspondingly little effect on evapotranspiration and discharge.

Water quality. All three of the alternative policy scenarios forced full protection or restoration of riparian forests, thus riparian forest cover increased by approximately 2600 km^2 (20%) over the 2005 landscape (Table 4-4). Thus, in all cases, 20% of streams would be likely to have lower temperatures and higher dissolved oxygen levels (Neill et al., 2006) than the current landscape, affecting species populations and

assemblages. In contrast, riparian forest cover decreased by an average of 3200 km^2 (ranging from 2500 km^2 to 6400 km^2) under the BAU scenarios. Thus, more streams would be likely to have higher temperatures and lower dissolved oxygen levels than the current landscape. More importantly, an average across BAU scenarios of nearly 40% of all streams would be subject to forest removal and potential oxygen depletion relative to any of the three alternative policy scenarios.

Among the policy scenarios, the proportion of microbasins having less than 40% vegetation cover was lowest in the ZSEE scenario (Table 4-4) because of differences in model predictions for cerrado microbasins. The RLR and CFC scenarios were similar to the current landscape, whereas under the BAU scenarios, the proportion of microbasins with this level of native vegetation loss from 41 to 72%. Only the CFC and ZSEE scenarios had a greater percentage of microbasins (10 to 16%) with more than 60% forest cover than the 2005 landscape. The RLR was equivalent to the BAU scenario, with 37% fewer microbasins having over 60% vegetation cover. These results indicate that any of the policy scenarios would be likely to maintain basic hydrological functions in more microbasins (Coe et al., 2009) in the region than the BAU scenarios, although the RLR scenario would be the least favorable option of the three policy scenarios.

Habitat. Overall, habitat quality and quantity differed between scenarios by biome, but was highest in the CFC and ZSEE scenarios. Perhaps most surprising, the RLR scenario had lower habitat quality than the BAU Low scenario. Although the BAU scenarios maintained nearly as much or more forest cover than the RLR scenario (Table 4-4), cerrado cover was generally much lower. This can be explained by the model's representation of the most likely pattern and trajectory of forest and cerrado

deforestation, which is more likely close to roads and previously cleared areas, if no controls on land-use that regulate the spatial distribution of native vegetation are put in place.

Forest fragmentation was consistently 2 to 3 times higher in the BAU scenarios than in the alternative policy scenarios; cerrado fragmentation was 1.4 to 1.7 times greater under BAU scenarios than in the alternative policy scenarios. Mean fragment sizes were 4- to 9-fold lower in the forest biome under BAU scenarios than under CFC or ZSEE scenarios. Differences were far less pronounced between fragment sizes under the RLR and BAU scenarios.

A strikingly different pattern emerges in comparing edge and interior habitat area among the scenarios. The RLR scenario has the highest amount of forest edge habitat and even less forest interior habitat than the BAU high scenario. All of the alternative policy scenarios have more total edge habitat than the BAU scenarios in both the cerrado and forest biomes. Additionally, the BAU Low scenario maintains more total interior habitat in both the forest and cerrado biomes than the RLR scenario.

Discussion

On balance, the results of this analysis indicate that even in an active agroindustrial frontier, with very high opportunity costs associated with forest conservation (Chapter 2, Chapter 3), compensation of opportunity costs can be achieved through carbon prices that are at the low end of estimates of future carbon markets. Larsen and Heilmayr (2009) estimate a substantial amount (300 $MtCO_2e$) of greenhouse gas emissions reductions in the United States could be achieved for less than 50 USD tCO_2e^{-1} by 2030. Currently, CO_2e is trading for approximately 20 USD tCO_2e^{-1} under the European Union's Emissions Trading Scheme (ETS; Point Carbon,

2009). In the CFC and ZSEE scenarios, I estimate the price of CO_2e at approximately 20 to 30 USD per ton.

Overall, the ZSEE scenario performed best in balancing ecosystem service protection with the costs of providing incentives to private landholders in the region to comply with the zoning plan, although its performance in terms of price was surprisingly similar to that of the CFC scenario. Emissions reductions are maximized under the ZSEE scenario, although overall carbon stocks are greatest under the CFC scenario by approximately 100 $MtCO_2e$ due to forest restoration. The per unit price of CO_2e to achieve the desired emissions reductions is lower under the CFC and ZSEE scenarios than the RLR scenario, although the total cost to achieve the reductions is more than 2-fold greater under the former scenarios. Notably, the RLR scenario provides only modest gains against BAU scenarios in terms of forest carbon stocks compared to the CFC and ZSEE scenarios. Although this scenario permits the greatest amount of agricultural development and the least opportunity cost to landholders (Chapter 2), its relatively poor performance (in comparison with the CFC and ZSEE scenarios) with respect to carbon stocks and other ecosystem services strengthens the argument for some type of compensation or incentive to encourage increased forest protection in the region.

Enhancing carbon stocks through active restoration doubles the total cost of complying with the RLR and CFC scenarios, but only increases the total cost of complying with the ZSEE scenario by 50%. In general, compensating restoration costs is expensive when the amount of accumulated carbon is relatively low given both the foregone potential income and the direct cost of restoration. In contrast, when a large

volume of carbon is accumulated, as under compliance with the CFC scenario, the per unit price of CO_2e falls markedly. This large volume of carbon in restored forests implies a much larger absolute cash outlay will be necessary to make the CFC scenario feasible. Nevertheless, even compensation for restoration at the same level as for the opportunity cost alone could potentially be important in encouraging riparian zone restoration, which is critical for overall stream health.

Critics of the Amazon Fund—and REDD more broadly—cite concerns that the proposal intends to reward landholders for complying with legislation to protect forests, arguing that individuals or companies should not be paid to obey the law (FOEI 2008). However, land-use policies which set out ambitious goals for forest and ecosystem service protection in the Amazon impose high costs on landholders and therefore achieve only low compliance levels (Chapter 2). Such policies could become more feasible under a compensation mechanism. However, it may be important to restrict eligibility for compensation so as not to inadvertently punish landholders that have maintained a minimum forest reserve of 50% by rewarding those who blatantly disregarded regulations (Chapter 2). When the subset of carbon for which landholders may be compensated is restricted to that portion that exceeds the stocks contained in legally required forest reserves, the ZSEE scenario is demonstrated to be the most efficient although the per unit price is not the lowest. This is because it retains 24 times the carbon stocks on the land than the lowest priced policy alternative, the RLR scenario, at only 1.3 times the per unit price. This pattern holds true at all three examined reference levels. The CFC scenario also retains high carbon stocks (although not as high as the ZSEE scenario), but the per unit price to achieve them is even higher

than for the ZSEE. Overall, compensating landholders in good standing for only part of their forest reserve reduces the cost of reducing emissions by nearly 4 USD tCO_2e^{-1} when compared to a plan that compensates all landholders in the region for protecting all remnant native vegetation. This price would put compensation well in the range of current carbon prices (Point Carbon, 2009).

REDD is also criticized for its narrow focus on carbon and the concern that non-carbon 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 are important concerns, especially given the poor performance of previous global initiatives to protect tropical forests (Winterbottom, 1990). The potential ecological costs and co-benefits 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 United Nations Framework Convention on Climate Change (UNFCCC).

An integrated zoning plan that takes into account agricultural suitability as well as environmental sensitivity may be one of the best ways to carry out REDD-related policies. As demonstrated in the Xingu River headwaters case study, the ZSEE scenario performs best with respect to a number of ecological indicators, including surface hydrology, several measures of water quality, and several measures of terrestrial habitat quality (e.g., increased forest connectivity and reduced edge effects). Furthermore, it optimizes agricultural production (Chapter 3) and carbon stocks relative to other existing policy alternatives. The case study demonstrates that the ecological co-

benefits 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 co-benefits could be maximized in the context of an integrated regional plan. In addition, a zoning plan, if extended over an entire region or nation (in this case, first to Mato Grosso, then the Brazilian Amazon), could reduce potential "leakage" effects, in which forest protection in one location leads to increased clearing or degradation in other location.

In this analysis, I have only explored the possibility of providing incentives and compensation to private landholders, largely because in the study region altering the behavior of landholders with respect to forest protection will be critical to an overall deforestation reduction strategy and thus is one of the major components of the Mato Grosso draft REDD plan (GOMT, 2009). In Brazil, the agro-industrial lobby is very powerful due to its large and growing share in generating national income. Currently, representatives of the sector are successfully lobbying to significantly weaken or overturn the bulk of Brazil's environmental legislation, arguably in response to the environmental lobby's largely uncompromising stance in recent years (Folha de São Paulo, 2009). Thus, including agricultural producers and ranchers in a scheme to provide incentives for good forest stewardship will be crucial for achieving deforestation reduction targets. However, for REDD or any other carbon-based compensation mechanism to be successful in Brazil and elsewhere, an integrated approach that

addresses a broad range of stakeholders and resource use types will be necessary. In the case study presented here, the ZSEE scenario addresses indigenous territories, smallholder settlements, and state and federal protected areas, as well as private landholdings. Furthermore, the extended plan includes a large number of non-spatial considerations related to optimizing ecological integrity and agricultural production for the state of Mato Grosso.

Conclusion

The regulatory framework created by the frameworks that I analyzed could greatly facilitate the implementation of markets for ecosystem services. Carbon storage and sequestration by forests is the ecosystem service with the most advanced market, and may soon expand precipitously for tropical forests through the UNFCCC (Gullison et al., 2007). It is very likely that REDD funding will flow to nations and regions that have mechanisms in place for rewarding communities and private landholders who are keeping their forests standing, and for identifying those rural land users who are out of compliance with land use legislation. The ZSEE under negotiation in Mato Grosso, and the property-level land registry developed in this state (Fearnside, 2003; Azevedo, 2009), provide an excellent framework for developing and implementing deforestation reduction targets that are tied to the emerging REDD carbon market.

Furthermore, if well executed, the potential social and ecological benefits of REDD are numerous. 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 non-local 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.

Table 4-1. Forest carbon stocks on private lands of the Xingu River headwaters region in 2005 and simulated for 2035 under six scenarios: Business-as-usual (BAU) with low, high, and average rates of deforestation; reduced legal reserve (RLR, legal reserve of 50% in the forest biome and 35% in the cerrado biome), current Forest Code (legal reserve of 80% in the forest biome and 35% in the Cerrado biome), and the socio-economic, ecological zoning plan of Mato Grosso state (ZSEE).

		Scenario (2035)					
	2005	BAU Low	BAU High	BAU	RLR	CFC	ZSEE
Non-riparian zone							
Stored in native vegetation ($MtCO_2e$)	1161	770	215	501	684	993	1041
Stored in restored vegetation (30 years) ($MtCO_2e$)	15	n/a	n/a	n/a	38	241	47
Total ($MtCO_2e$)	1176	770	215	501	722	1234	1088
Riparian zone							
Stored in native vegetation ($MtCO_2e$)	120	90	39	80	120	120	120
Stored in restored vegetation (30 years) ($MtCO_2e$)	2	n/a	n/a	n/a	7	7	7
Total ($MtCO_2e$)	122	90	39	80	127	127	127
Grand total							
Stored in native vegetation ($MtCO_2e$)	1281	860	254	581	804	1113	1241
Stored in restored vegetation (30 years) ($MtCO_2e$)	17	n/a	n/a	n/a	45	248	54
Total ($MtCO_2e$)	1298	860	254	581	849	1361	1295

Table 4-2. The effect of policy scenarios on the costs, carbon stocks, and costs per ton of carbon associated with additional forest cover in the privately-owned land of the Xingu River headwater region. Calculations are made relative to each of three business-as-usual (BAU) scenarios of future forest cover. Costs include the total economic costs (opportunity costs of foregone profits from agriculture or ranching plus forest restoration costs). Policy scenarios (RLR, CFC, ZSEE) are described in Table 4-1.

	BAU (average)			BAU (high rate)			BAU (low rate)		
	Reduced Emissions	Enhancement	Total	Reduced Emissions	Enhancement	Total	Reduced Emissions	Enhancement	Total
Cost (million USD)									
RLR	4872	4316	9188	8898	4316	13,214	1477	4316	5793
		(±899)	(±899)		(±899)	(±899)		(±899)	(±899)
CFC	9562	8709	18,271	13,818	8709	22,527	4938	8709	13,647
		(±905)	(±905)		(±905)	(±905)		(±905)	(±905)
ZSEE	9940	4917	14,857	14,256	4917	19,173	5274	4917	10,191
		(±583)	(±583)		(±583)	(±583)		(±583)	(±583)
Carbon stocks ($MtCO_2e$) (relative to BAU stocks)									
RLR	247	38	285	486	38	524	70	38	108
CFC	521	241	762	789	241	1030	276	241	517
ZSEE	559	54	613	834	54	888	304	54	358
Price of carbon (USD tCO_2e^{-1})									
RLR	20.1	102.3	32.2	18.3	102.3	25.2	21.2	102.3	53.6
		(±11.9)	(±3.2)		(±11.9)	(±1.7)		(±11.9)	(±8.4)
CFC	18.3	36.1	24.0	17.5	36.1	21.9	17.9	36.1	26.4
		(±3.7)	(±1.2)		(±3.7)	(±0.9)		(±3.7)	(±1.7)
ZSEE	17.8	103.9	24.2	17.1	103.9	21.6	17.4	103.9	28.5
		(±12.3)	(±1.0)		(±12.3)	(±0.8)		(±12.3)	(±1.6)

Table 4-3. The effect of policy scenarios on the basin-wide costs, carbon stocks, and costs per ton of carbon associated with additional forest cover in excess of 50% cover of privately held lands in the forest biome of the Xingu River headwater region, represented by individual microbasins. Calculations are made relative to each of three business-as-usual (BAU) scenarios of future forest cover. Policy scenarios (RLR, CFC, ZSEE) are described in Table 4-1. Costs include the total economic costs (opportunity costs of foregone profits from agriculture or ranching plus forest restoration costs).

	BAU (average) Reduced Emissions	BAU (high rate) Reduced Emissions	BAU (low rate) Reduced Emissions
Cost (million USD)			
RLR	84	92	59
CFC	2456	2556	2103
ZSEE	2758	2873	2396
Carbon ($MtCO_2e$)			
RLR	8	8	5
CFC	162	171	134
ZSEE	192	204	162
Price of carbon ($USD\ tCO_2e^{-1}$)			
RLR	11.2	11.0	12.2
CFC	15.2	14.9	15.7
ZSEE	14.4	14.1	14.8

Table 4-4. Comparison of ecological features of the Xingu River headwaters regions with 2005 forest cover, under three simulations of future forest cover under business-as-usual assumptions, and under three simulated policy scenarios. The policy scenarios include RLR, CFC, and ZSEE, as described in Table 4-1. Ecological features include parameters for surface hydrology and regional climate, indicators related to water quality, and terrestrial habitat quantity and quality.

				Scenarios			
	2005	BAU Low	BAU High	BAU	RLR	CFC	ZSEE
Surface Hydrology and Regional Climate							
Mean Annual Discharge ($m^3 s^{-1}$)	3113	3314	3579	3410	3235	3055	3018
(% change from potential)	(13%)	(21%)	(31%)	(25%)	(17%)	(11%)	(10%)
Mean Annual Evapotranspiration ($m^3 s^{-1}$)	6570	6368	6103	6272	6477	6665	6628
(% change from potential)	(-5%)	(-8%)	(-12%)	(-10%)	(-7%)	(-4%)	(-4%)
Water Quality							
Riparian forest cover (km^2)	12,931	10,431	6527	9697	15,509	15,509	15,509
Mean % vegetation cover per microbasin	69 (0.5)	53	30	43	57 (0.4)	73 (0.4)	73 (0.4)
% of microbasins with greater than 60% vegetation cover	65	34	23	28	28	81	75
% of microbasins with less than 40% vegetation cover	18	41	72	61	18	16	11
Terrestrial Habitat							
Vegetation cover (km^2)							
Forest	107,789	81,483	45,246	64,602	51,561	116,395	110,902
Cerrado	17,037	8567	3988	6305	9083	13,149	15,209
Number of fragments							
Forest	13,427	40,568	49,780	46,510	23,673	12,958	13,770
Cerrado	13,285	24,881	25,467	26,702	18,422	18,138	15,573
Mean distance to nearest neighbor fragment (m)							
Forest	361	403	564	465	376	379	363
Cerrado	406	427	490	442	354	371	378
Mean fragment size (ha)							
Forest	803	201	91	139	218	898	805
Cerrado	128	34	16	24	50	73	98
Total interior habitat area (km^2)							
Forest	99,978	72,744	40,498	56,977	36,975	105,287	98,973
Cerrado	12,737	5758	2030	3569	4001	7433	9753
Total edge habitat area (km^2)							
Forest	7810	8739	4748	7624	14,586	11,108	11,930
Cerrado	4300	2809	1958	2736	5082	5716	5456

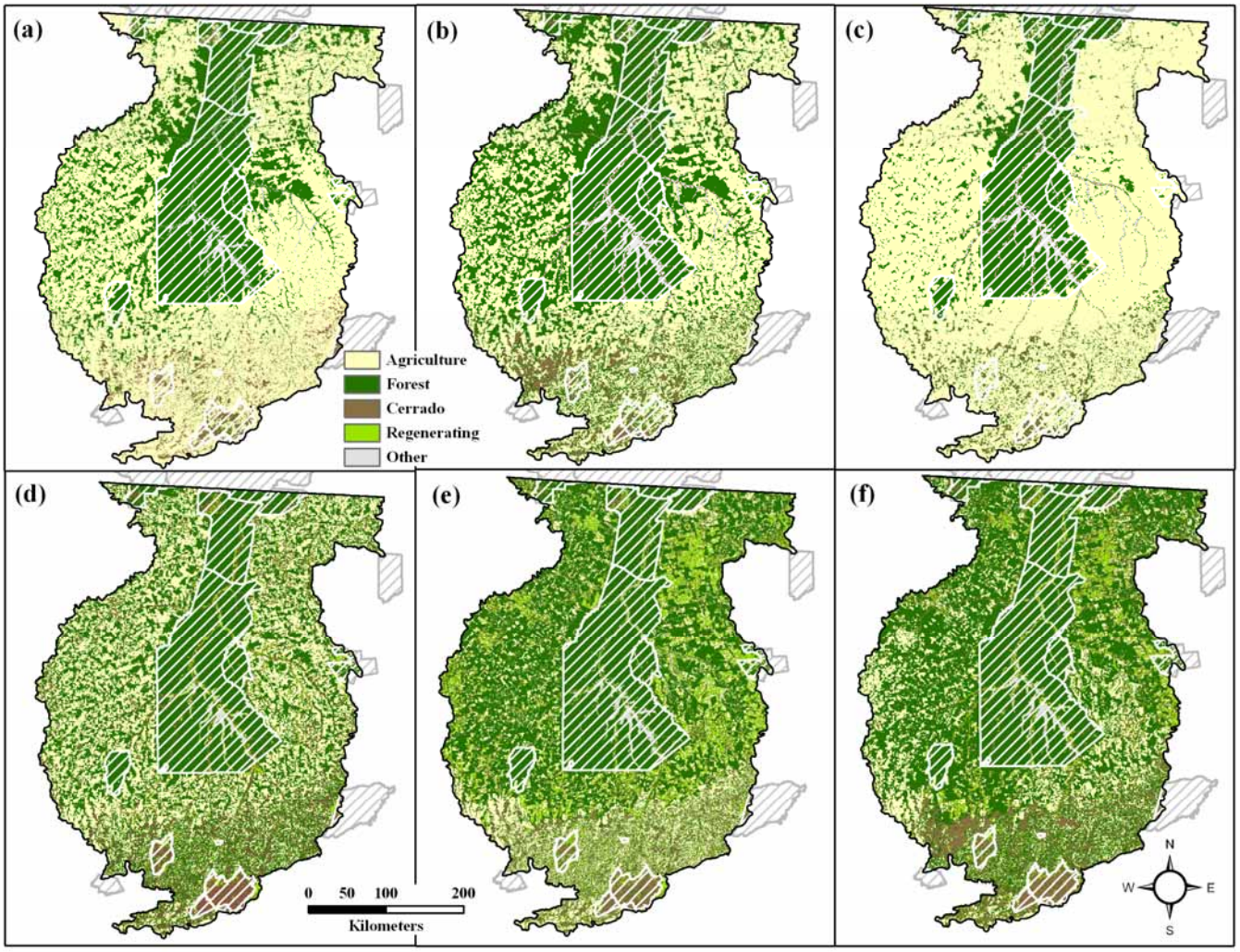

Figure 4-1. Maps showing simulated future (2035) land cover of the Xingu River headwaters region under six different scenarios. These scenarios are described in Table 4-1 and include: (a) Business as Usual (BAU), average rate; (b) BAU, low rate; (c) BAU, high rate; (d) RLR (reduced legal reserve); (e) CFC (current Forest Code); and (f) ZSEE (socio-economic, ecological zoning plan). Indigenous territories and protected areas are indicated with white hatching.

CHAPTER 5
CONCLUSION

By suddenly increasing the “legal forest reserve” of the Brazilian Forest Code from 50 to 80% of private landholdings in the Amazon region, in 1996, the Brazilian Government imposed substantial costs on farmers and cattle ranchers, reaching nine billion dollars in the Xingu River headwaters region alone (three to four million dollars, on average, for each of the region’s landholders). By maintaining the new measure in a state of perpetual modification and revision for the next five years, the government may have further undermined the perception, by landholders, of the new rules as binding and permanent. Finally, by failing to provide programs that facilitate landholder compliance with the new regulation, the potential role of the law to protect private land forests may have been further diminished. Compliance with the new Forest Code fell to only 46% of landholders in 2005 (Chapter 2). The potential ecological benefits of a fully implemented 80% legal reserve requirement are substantial. Compared to the fully implemented 50% requirement, it would provide an additional 120 million tons of carbon (450 tons of CO_2 equivalent (tCO_2e)) stored in forests, higher evapotranspiration, lower stream and river discharge (with reduced risks of flooding), and nearly twice the amount of interior forest habitat (Table 2-9).

Problems with the modified Forest Code were foreseen. The 1988 Constitution established a new land use policy instrument, known as the socio-economic, ecological zoning plan (ZSEE). These state-level plans, upon approval by the state legislative assembly, permit the definition of specific geographical zones in which the legal forest reserve requirement of the Forest Code can be reduced to 50%. A second instrument for addressing the economic costs of implementing the 80% modification of the Forest

Code is the provision for legal reserve compensation. In the case of the Xingu River headwaters region, the Mato Grosso state ZSEE has not been finalized. If it were implemented, it could greatly increase the feasibility of moving agricultural regions into compliance with the law while protecting ecosystem services. The economic opportunity cost of compliance with land-use regulations incurred by landholders in the region as a whole would decline by USD 2.5 billion; for the average landholder, the cost of compliance would decline from 4.2 million USD to 1.6 million USD through implementation of the ZSEE. In contrast, the tradable development rights provision of the modified Forest Code would have virtually no effect on the cost of legal compliance in the Xingu River headwaters region, since very few of the region's microbasins (the proxy for landholdings in this study) have more native vegetation than is mandated by the law. This sharp reduction in opportunity costs achieved through the ZSEE does not imply deep sacrifices in the provision of ecological services. Evapotranspiration and stream discharge remain virtually the same in ZSEE compared with full compliance with 80% legal reserve, although forest carbon stocks and forest interior habitat decline. Furthermore, the ZSEE protects more native *cerrado* vegetation than the post-1996 Forest Code because the reduction in restoration requirements in the forest biome reduce leakage of clearing for agricultural lands into the *cerrado* biome.

The major policy alternatives under discussion for the Xingu River headwaters region, including a reduced legal reserve (RLR, returning to 50% from the current 80% requirement), the current legal code (CFC, 80%), and the ZSEE, would result in different amounts and patterns of coverage of remaining and restored forests and cerrado with important implications for the amount of carbon contained on the

landscape, the costs to producers, and the associated ecosystem services. The gain in forest carbon would be much smaller under the RLR scenario (285 to 584 $MtCO_2e$), than under the CFC (762 to 1030) and ZSEE (613 to 888 $MtCO_2e$) relative to the range of BAU scenarios. The associated cost per ton of CO_2e would be somewhat higher for the RLR scenario (18-21 USD tCO_2e^{-1}), than the CFC (18 USD tCO_2e^{-1}) and ZSEE (17-18 USD tCO_2e^{-1}) scenarios. The current price of CO_2 in the carbon market is approximately USD 21 per ton (Point Carbon, 2009), suggesting that the potential for carbon offsets to provide one source of financial incentives for landholders in the region to comply with regulations is high.

There is concern, however, that emerging carbon markets, such as REDD, could generate negative ecological consequences as it fosters greater carbon storage on the land. REDD could displace agricultural expansion from a region of high carbon density to a region of low carbon density, potentially eradicating native ecosystems that are rare and contain species threatened with extirpation or extinction (Stickler et al., 2009). In the case of the Xingu River headwaters region, under a zoning plan, incentives tied to carbon stock maintenance and enhancement could also foster improved watershed function, and increase habitat quality and quantity (e.g., through increased forest connectivity, reduced edge effects) while allowing for reasonable areas of land to be made available for agricultural production. The Xingu headwaters case study demonstrates that the ecological co-benefits of REDD are sensitive not only to the quantity of forests and woodlands remaining on the landscape, but also to their spatial distribution.

This case study of the Xingu River headwaters region indicates that the prospects for protecting public interests in private forests in regions such as the Brazilian Amazon are high. Although no policy framework is likely to present a perfect solution to balancing two public goods—agricultural production and the provision of ecosystem services—this assessment does indicate that a happy medium may be achieved by identifying areas likely to be more suitable for agriculture as well as those which are more environmentally sensitive. With 83% of the forest remaining, there is still time to correct the mistakes made in modifying and implementing the Forest Code. However, the moral idealism of the legal tradition in Brazil may make the notion of positive economic incentives for compliance with the law difficult to consider. With a focus on the "ideal" versus the "practical" solution to issues of the public good, and its defense, Brazil's civil law system typically produces new regulations and laws without the dialogue and debate among interested stakeholders that could build into the design of the new law mechanisms for increasing its practicality and chances of successful implementation. However, multi-stakeholder, participatory planning processes are already part of recent watershed management legislation. More and more, civil society is initiating such processes for a variety of environmental governance issues, including infrastructure development (Campos and Nepstad, 2006; Soares-Filho et al., 2004), watershed management (ISA, 2005), and other regional planning processes (Perz et al., 2008), signaling an important trend in Brazilian environmental rule-making in the future.

This study presents the results of an integrated assessment that attempts to evaluate a more complete set of ecological and economic trade-offs associated with alternative policies for managing a large-scale agro-industrial landscape by projecting

the likely land-cover outcomes of each of those policies into the future. This approach is important for policy makers and stakeholders in the region because it allows them to make informed decisions regarding land-use planning based on a quantitative analysis of the effects of each policy option on a series of indicators. For the first time, this research brings such a quantitative analysis into a debate that has been historically ruled by ideological perspectives. Thus, the results presented here are already being used to improve policy design in the region.

APPENDIX A
LAND-COVER MAPS

I developed land-cover classification maps for the years 1996, 1999, 2005, and 2007 using image segmentation and object-oriented classification techniques of 30-m resolution Landsat TM image mosaics corresponding to each year. I generated maps with 5 land-cover classes for each year: forest, cerrado, agriculture in the forest biome, agriculture in the cerrado biome, and a miscellaneous category.

Sensor Data

Twelve Landsat-5 TM (L1G) scenes (bands 1-7) were acquired for each of 4 years (1996, 1999, 2005, 2007). All images were acquired during the dry season to minimize cloud cover (Table A-1). Pre-processing of the L1G scenes included radiometric calibration, atmospheric correction, georeferencing, and mosaicing. The radiometric data were converted from radiance to ground reflectance (Jensen, 2000) to eliminate variability and noise due to differences in satellite instrumentation, Earth-Sun distance, solar elevation angle, solar curve, and atmospheric effects on different acquisition dates and times using the calibration technique of Green et al. (1999). Geometric rectification was performed using nearest neighbor resampling to co-register each scene to its orthorectified GeoCover analogue obtained from the Global Landcover Facility (GLCF). Using a combination of manual and automatic registration methods, all scenes were transformed to the UTM coordinate system and WGS 84 datum. A root mean square (RMS) error of <0.5 pixels was achieved for all scenes. Automatic mosaicing was conducted after manual exclusion of areas covered by clouds, shadows, and/or smoke, using histogram matching and seam-line generation. The 1996, 1999, and 2005

mosaics were co-registered to the 2007 mosaic to increase consistency between the images and subsequent maps.

Reference Data

A total of 989 reference points were used for classification model calibration and validation. Of these, 535 were ground reference plots sampled across 15 land-cover/land-use types during an eight-week, dry-season field campaign between 26-June and 23-August 2006 (Walker et al., in prep.). At each location, a 900-m^2 plot was established, and the land-cover/land-use class, land-use history, mean vegetation height, and geographic coordinates were recorded. All plots were located along or within the Xingu basin watershed boundary, and for accessibility reasons, tended to be concentrated along primary and secondary transportation routes outside of major protected areas in the region. Landcover classes for all years were assigned using land-cover/land-use class in 2006, land-use history information, and cross-referencing between years.

The remaining 454 points were collected via on-screen identification using the 2007 Landsat 5 mosaic as a base map and in conjunction with the ground reference dataset. Point locations were determined through the random superimposition of a uniform 35 km^2 point grid on the Landsat image mosaic with the overall goal of obtaining supplemental reference data points in regions previously unsampled, most notably areas that were inaccessible for field sampling. Land-cover classes for the two other years were assigned by cross-referencing between years.

Classification and Mapping Approach

Segmentation and Attribute Extraction

Segmentation refers to the automated and optimal grouping of image pixels, i.e., the partitioning of a landscape, into spectrally homogenous and spatially distinct regions or “image objects” based on some predefined, knowledge-based criteria (Baatz et al., 2004; Navulur, 2007). Unlike individual image pixels, image objects can be smartly characterized by hundreds of object-level attributes, including measures of shape, size, texture, and morphology, in addition to standard spectral descriptions, and numerous investigators have demonstrated the superiority of object-based strategies to empirical modeling and mapping over traditional pixel-based approaches (Baatz et al., 2004; Walker et al., 2007).

The eCognition software package provided the computational framework for image segmentation and subsequent object-level attribute extraction. For the purposes of producing a second segmentation product suitable for classification and mapping, fourteen individual image data layers were used as inputs to this segmentation including raw bands 3-7, principal components 1-3, Tasseled Cap brightness, greenness, and wetness, NDVI, the 3-arcsecond SRTM DEM, and the SRTM-derived slope.

The segmentation process is guided primarily by three criteria: 1) scale, 2) shape, and 3) smoothness/compactness. Whereas the scale parameter controls the average size of the image objects to be generated, the latter two criteria determine the homogeneity of image objects. Hierarchical (top-down), bi-level segmentation procedures were parameterized using scale parameters of 40 and 20, shape factors of 0.1, and compactness/smoothness factors of 0.5. Parameters were determined empirically using an iterative post-segmentation inspection/refinement approach.

Further information on eCognition parameterization can be found in Baatz et al. (2004). The final segmentation generated an average of 474,265 image objects with an average object size of 0.7 km^2 (range: 0.001-36.0 km^2) for each image. Fifty putative predictor variables were computed for each of the segments. Of the 50 variables, 26 spectral, 13 textural, 7 ancillary-spatial, and 4 ancillary-topographic variables were included.

Spatial Database Joining

To compile the tabular databases required for use in the development of the classification models, spatial joins were established between the reference point dataset and each of the segmentation products (i.e., for each year). The joining procedure resulted in a database file of 989 rows by 53 columns (52 predictor and 1 response variables).

Classification and Mapping

Empirical-statistical machine learning techniques such as classification and regression trees (CART) have become increasingly popular within the remote sensing research and applications community, particularly when the objectives involve broad-scale mapping (e.g., Baccini et al., 2009; Blackard et al., 2008; Goetz et al., 2005; Hansen et al., 2008; Walker et al., 2007). These approaches tend to be user-friendly, computationally less demanding, and can achieve high predictive accuracies when well calibrated for the region of interest. For the purposes of this classification effort, the randomForest (RF) algorithm, implemented in the open source R statistical programming environment, was used. First proposed by Breiman (2001), the RF algorithm falls into the category of ensemble learning methods where the goal is to construct a “forest” (i.e., ensemble) of individual classifiers that are later combined to improve predictive accuracy. In the case of RF, independent trees are constructed using

a bootstrap sample of the data set in a process called *bagging*, a term derived from *b*ootstrap *agg*regation (Bauer and Kohavi, 1999; Dietterich, 2000; Liaw and Wiener, 2002). In each bootstrap sample, approximately 1/3rd of the reference cases are left out. These “out-of-bag” samples are predicted during each bootstrap iteration (i.e., generation of each tree), and later aggregated to produce an out-of-bag (OOB) estimate of error. Direct comparisons between OOB error rates and error rates computed under independent validation scenarios have found the OOB estimates to be quite robust (Liaw and Wiener, 2002; Walker et al., 2007).

Three separate randomForest classifications (corresponding to each year in the study) with 6 classes (agriculture, forest, *cerrado*, open water, wetlands, sandbars in areas of open water) were generated and evaluated based on overall classification accuracy and Cohen’s Kappa statistic. Subsequently, some classes were merged (water, wetlands, and sandbars to a miscellaneous class) and post-classification sorting (Janssen et al., 1990; Vogelmann et al., 1998) split some of the automatically generated classes to achieve a final classification with 7 classes for each year (forest, *cerrado*, regenerating forest, regenerating *cerrado*, agriculture in the forest biome, agriculture in the *cerrado* biome, other). Specifically, cleared areas were recoded by biome (forest and *cerrado*) to differentiate between dynamics of clearing in the forest and *cerrado* biomes. The *cerrado*-forest biome map was obtained by merging a map of forest/non-forest derived from INPE Prodes maps (INPE, 2009) with a map of biomes from the IBGE RADAM vegetation thematic map (IBGE, 1981). Regeneration and/or abandonment were also recoded as necessary by cross-referencing between years.

Table A-1. Image dates for Landsat-based mosaics for 4 years (1996, 1999, 2005, 2007).

PATH/ROW (WRS-2)	*YEAR* (MM/DD)			
	1996	1999	2005	2007
P224/R67	07/03	10/08	07/28	06/16
P224/R68	07/03	08/21	07/12	06/16
P224/R69	06/17	08/21	08/13	08/19
P224/R70	07/03	08/21	06/10	08/19
P225/R67	07/10	08/12	07/03	07/09
P225/R68	07/10	08/12	08/04	07/09
P225/R69	07/10	08/12	08/20	07/09
P225/R70	06/24	08/12	04/14	09/09
P226/R67	07/01	08/19	07/10	08/17
P226/R68	07/01	07/02	07/26	08/17
P226/R69	06/15	07/02	05/07	08/17
P226/R70	07/17	08/03	05/07	08/17

APPENDIX B
DYNAMIC SPATIAL SIMULATION MODEL DEVELOPMENT

Overview

I developed a dynamic landscape simulation model to model future landscape trajectories corresponding to a set of alternative policy proposals. The model is based on a spatial-statistical model of land-use change, derived from a land-use/land-cover change analysis and a GIS consisting of data related to the location and neighborhood context of 4 focal land-cover 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 rates calculated from the 1996-2005 reference period. The model integrates coupled components developed within two spatial structures: (1) subregions defined by hydrographic sub-basins, and (2) raster cells (4510x5963) at 1-ha resolution.

The model uses a nested, sub-regional approach based on hydrographic basins, to better reflect the ecological and legal reality of land-use policy in the region, as well as to facilitate model processing. The model is calibrated and run separately for 6 3rd-order sub-basins (explained in more detail below), the results of which are merged at every time step. This step decreases processing memory requirements and better simulates actual land-cover change processes by regionalizing rates, relationships to proximal drivers, and patterns.

Each of the 6 sub-basins is further sub-divided into micro-watersheds representing individual stream reaches (1:1,000,000 scale), which interact such that the proximity of a deforestation front in one micro-basin influences deforestation in a neighboring micro-

basin. Within a sub-basin, all microbasins are subject to the total annual land-cover change rate for the whole basin. For the baseline (Business as Usual) scenario simulation, the sub-division into microbasins does not affect the location of the focal land-cover change events. However, this becomes important in modeling alternative land-use policy scenarios in which regulations are established for the property level. As a complete map of property boundaries for the region is not yet available, I employ the micro-basins as proxies for individual properties to better simulate policy outcomes on private lands. The mean size and range of sizes of the 2881 micro-basins (x = 5981 ha, 4-70,766 ha) is comparable to that of private properties in the region (Jepson, 2006; Fearnside, 2005). Furthermore, Brazilian water law requires management plans at the watershed level and the Brazilian Forest Code stipulates that deforestation rights may be traded by property owners within watersheds in certain cases (MP 2166-67, 2001; Chomitz, 2004; Stickler, 2009).

The model has 4 basic steps: First, annual deforestation rates are calculated for each 3rd-order watershed based on conversion rates calculated for the period 1996 to 2005. Next, annual deforestation probability in relation to a set of spatial variables is obtained using “weights of evidence” analysis (Soares-Filho et al., 2004) for each of the 6 sub-basins. Third, for each sub-basin, I developed a unique spatial simulation model. Finally, I validated the model by comparing the simulated and observed landscapes for the year 2007. I developed all modeling phases using the Dinamica EGO graphical interface platform (http://www.csr.ufmg.br/dinamica/) that has the capacity to process multiple large map sets and has special features for advanced spatial modeling and simulation (Soares-Filho et al., 2009).

Model Calibration

First, annual deforestation rates are calculated for each 3rd-order watershed based on conversion rates calculated for the period 1996 to 2005. The amount of change in each transition of interest is computed from a Markov matrix obtained through the comparison of land use/cover maps for the two dates.

Second, annual deforestation probability in relation to a set of spatial variables is obtained using "weights of evidence" analysis for each of the 6 sub-basins. I used the Weights of Evidence method (Soares-Filho et al., 2004) to select the variables most related to observed landscape changes as well as to quantify their influences on each of the modeled transitions. Weights of Evidence is a Bayesian method traditionally used to derive favorability maps for spatial point phenomena (Agterberg and Bonham-Carter, 1990; Bonham-Carter, 1994). In this study, weights of evidence (W_k^+) are calculated for every *k* category of each spatial variable under analysis and can be interpreted as the influence of that category on the chances of a deforestation event occurring. Since this method only applies to categorical data, it is necessary to categorize continuous gray-tone variables, such as distance-decay maps; this is done using a method adapted from Agterberg and Bonham-Carter (1990) in Dinamica EGO.

The variables I examined comprised a set of biophysical and socioeconomic (or proximate) factors that spatially determine the location of the changes. This set includes slope, elevation, soil type, protected areas (including indigenous territories), suitability for annual crops and cattle ranching, INCRA small-holder settlements, distances to rivers, major and secondary roads, and urban centers, and distances to forested, regenerating and deforested areas. A basic assumption for the Weights of Evidence method is that the variables must be spatially independent. I tested the spatial

independence of the aforementioned variables using the Crammer coefficient (V) and found that all variables, except the pair “soil type” and “suitability”, have values lower than an empirical threshold (V < 0.45), and thus are spatially independent (Almeida et al., 2003). I retained “suitability” as it encompassed more information than the soil variable.

Landcover Simulation

For each of the 6 sub-basins, I developed interacting spatial simulation models with customized parameters consisting of (1) a cellular automata type model that simulates the spatial patterns of (a) deforestation or clearing of *cerrado*, based on a probability map depicting the integrated influence of proximate drivers on the location of clearing at each time step (Soares-Filho et al., 2002), and (b) regeneration, according to a set of exogenous rules and assumptions; and (2) a *road constructor* model that projects the expansion of secondary road network, and thereby incorporates the effect of road expansion on the evolving spatial patterns of deforestation (Soares-Filho et al., 2004). Deforestation or clearing of *cerrado* occurs in accordance with the rate determined for each sub-basin by the Markovian transition matrix and the probability map determined by the set of spatial variables. As microbasins become saturated with deforestation, the model chooses the pixel in a neighboring microbasin with the next highest probability of being deforested within a given sub-basin. Depending on the assumptions of a given scenario, certain areas (e.g., protected areas) and individual micro-basins within a sub-basin can be forced to become saturated despite having a high probability of being deforested, thus causing the model to search for high-probability pixels in neighboring micro-basins that may still be cleared. The model includes regeneration only when a scenario requires that regeneration take place.

Regeneration occurs in one of two, not mutually exclusive, ways: (1) within pre-determined zones (e.g., riparian buffer zone, proposed protected areas), and/or (2) within individual micro-basins according to the requirements of the assumptions of a given scenario (e.g., according to state or federal legislation or some other proposal for private lands). In the latter case, regeneration preferentially occurs adjacent to the defined riparian zone, adjacent to remnant large blocks of forest, and/or farther away from roads and urban centers. At each time step, sub-basin level maps depicting saturation, time of residence (for pixels that have been subjected to a transition), land-cover, probability, and cumulative roads are merged and passed on to begin the next time step.

Baseline Scenario Development. The baseline (or Business as Usual, BAU) scenario assumes that the current rate of deforestation, level of compliance with environmental legislation, and accompanying land-use/land-cover change will continue, and thus serves as a baseline model against which to compare other regulatory options. The baseline scenario, referred to as the “business-as-usual” (BAU) scenario, assumes that historical trends will continue into the future, projecting regional rates using 1996-2005 figures. This scenario applies a Markovian approach, simply projecting the changes into the future using transition rates annualized from the 1996-2005 time-period transition matrix using the general spatial allocation approach described above (for details see Soares-Filho et al., 2002).

Model Validation

Finally, I validated the model by comparing its predictions to the observed data for 2007 using a fuzzy map comparison which compares simulated deforestation to observed deforestation (Soares-Filho et al., 2009; Almeida et al., 2008; Hagen, 2003).

Two-way fuzzy comparison using a constant decay function provides a detailed assessment both of categorical and spatial similarity of two maps that more closely mimics human visual comparison by overcoming restrictions induced by hard pixel limits like pattern quantification and exclusive cell state (Hagen-Zanker, 2006; Visser and Nijs, 2006). This method compares the number of cells of a certain class in a simulated map with the number of these cells in a reference map that fall within a central cell neighborhood, as defined by a window size. By using a constant decay function, if a matching cell is found within the window, fit is assigned to 1, otherwise 0. Windows with increasing sizes convolute over the map and a mean is computed for each window size. This method employs a reciprocal approach, comparing the match between the observed map and the simulated map, and vice versa, ultimately choosing the minimum mean in order to penalize random maps, which tend to overestimate the fit. In this manner, this method accounts for both omission and commission errors. Our comparison employed increasing windows sizes from 1 to 11 cells, which in terms of map resolution represent a range of 100x100-m to 1100x1100-m. The overall agreement between the two maps ranged from 60 to 77%.

APPENDIX C
ECONOMIC AND ECOLOGICAL ASSESSMENT

Economic Assessment

I estimated the economic costs of complying with the policy change by estimating (1) the potential forgone profits to producers over the whole landscape and by microbasin using net present value as a proxy for the potential value of agricultural lands in the region, and (2) the cost of riparian forest restoration over the whole landscape and by microbasin, where necessary.

Net Present Value

Net present value (NPV) for the region was estimated using spatially-explicit rent models for soy production (Vera Diaz et al., 2007; Nepstad et al., 2007a), cattle ranching (Nepstad et al., 2009), and sustainable timber harvest (Merry et al., 2009)—the three major economic activities in the region. These models estimate the potential rent of each economic activity on analyses of the costs of production (several of which are spatially-dependent, such as transportation costs), yields, and prices. For each of the three economic activities, the NPV was estimated for 30 years into the future assuming a 5% annual discount rate and a plausible schedule of highway paving (Soares et al., 2006). Agricultural land values are typically appraised by determining the production value of the land, as determined by the NPV of the specific use to which that land is or will be put (Van Kooten and Bulte, 2000). In this case, the range and distribution of NPV in the region is similar to that of actual land values for which prices are only available at the municipal level (FNP, 2005). Although the models do not account for short-term fluctuations, they use a set of assumptions that provide conservative projections of profit for each activity.

The layers derived from the rent models were combined such that, for any given pixel, the NPV of timber harvests (where this value was greater than 0) was subtracted from the highest value from either cattle or soy activities. Negative values resulting from this calculation were set equivalent to a net present value of zero. I used the resulting map of combined NPV of the Xingu headwaters to estimate total maximum potential NPV for both forested and cleared lands—*if they were to be cleared or remain cleared*—within the forest biome as well as per microbasin.

Restoration Costs

I estimated the costs to restore both legal reserve and riparian areas where required under the modeled scenarios. To estimate the cost of riparian zone restoration, I used figures generated in field trials carried out by non-governmental organizations active in the region: Instituto Socioambiental, Aliança da Terra, and Instituto de Pesquisa Ambiental da Amazônia. I adapted the riparian restoration costs to estimate legal reserve restoration costs, eliminating methods that involve out-planting and maintenance of seedlings as this would be extremely costly and labor-intensive over the relatively larger areas needing restoration. Restoration costs range from USD 536 ha^{-1} to USD 3217 ha^{-1}, depending on the type of adjacent land-use and the intensity of treatments required to restore forest (Table C-1). Although the cost could be as low as USD 0 ha^{-1}, regeneration would likely be slow and success would be highly dependent on the type and intensity of previous land-use, as well as distance to nearest seed source.

Ecological Assessment

I compared the final landscapes for the observed and modeled landscapes in terms of carbon stocks, river discharge, annual evapotranspiration, terrestrial habitat

quality, and water quality. Unlike other analyses in this study, which were restricted to private lands in the Xingu Basin, ecological consequences were assessed for the entire headwaters region, including all protected areas. Here, I describe how each indicator was assessed.

Carbon Stocks

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, adapted in Nepstad et al., 2009). I overlaid the map of biomass with the final outcome map for each scenario and assigned biomass values to each land-use/land-cover class in the simulated map. For intact native vegetation classes (intact forest, intact cerrado, native wetlands), the values in the aboveground biomass map were assigned directly. I converted biomass values to CO_2-equivalent (CO_2e) values for each pixel. For areas of agriculture or pasture, I assigned a value equivalent to 15% of the original carbon stock, which represents the reduction in biomass following clearing since sufficient information to accurately identify pasture and soy expansion was not available (Houghton et al., 2000). For areas of regenerating forest and cerrado, I assigned a value of 1.5 tC $ha^{-1}y^{-1}$ and 0.5 tC $ha^{-1}y^{-1}$ (Houghton et al., 2000; Zarin et al., 2001) per pixel, respectively, and multiplied by the number of years that regeneration in each pixel had taken place (ranging from 1 to 29 years).

Surface Hydrology and Local and Regional Climate

To investigate the impact of each scenario on the surface hydrology of the Xingu River in the absence of atmospheric feedbacks to precipitation, simulations with a land surface model (IBIS; Kucharik et al., 2000) and a river transport model (THMB; Coe et al., 2009) were carried out. I carried out offline simulations (as described in Stickler et

al., 2009; Coe et al., 2009) for the landscape maps for all scenarios (3 historical, 2 modeled), as well as for a scenario describing potential (historical) land-cover in the region prior to settlement (referred as the Control (CTL) scenario). As the THMB model has a resolution of 5-min, whereas the land-cover maps have a resolution of 100 m, new 5-min cell size land-cover maps to serve as input to THMB were derived by calculating the fraction of each 5-min grid cell size represented by disturbed and undisturbed vegetation for each scenario. The land-cover maps corresponding to each scenario were first recoded such that they contained only 2 classes, disturbed and undisturbed vegetation (corresponding to the two extreme scenarios used by IBIS to calculate mean monthly surface and sub-surface runoff for THMB, IBIS-POT (which assumes all intact vegetation) and IBIS-GRASS (which assumes all disturbed or cleared vegetation). In the 2 theoretical Forest Code scenarios, regenerating forest and cerrado were reclassified as intact vegetation to reflect the intention that these lands will remain protected and allowed to continue to regenerate. The differences from CTL in simulated discharge of the three observed landscapes and the two theoretical landscapes quantify the sensitivity of the surface hydrology (discharge and flooding) to land cover changes.

For each scenario, I extracted simulated monthly discharge values for 32 years (1968-2000) for the main trunk of the Xingu River at the border between Mato Grosso and Pará states. I calculated annual mean discharge (N = 32) for each scenario. I calculated mean annual evapotranspiration by subtracting mean annual discharge from mean annual precipitation. Mean annual precipitation was derived for the same location from an interpolation of observed climate data over the same time period (1968-2000) over which the discharge simulations were carried out. For each scenario, the total

volume and the percent change from the potential in annual discharge and annual evapotranspiration for each scenario is presented.

Indicators of Water Quality

To compare the effects of native vegetation distribution for water quality, I assessed a series of landscape-level measures that associated with broad physical and chemical changes in water. The primary landscape measure associated with water quality is the presence of riparian zone vegetation. For each landscape, I calculated the amount of each land-cover type within the riparian zone by overlaying each landscape with the riparian buffer map described above. Next, I calculated the percent of forest or cerrado remaining in each of 2881 micro-basins comprising the headwaters region. Together with the presence of riparian vegetation cover, this percent forest cover serves 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 (Neill et al., 2006; Nepstad et al., 2007b).

Terrestrial Habitat

To evaluate differences in habitat quantity and quality among the scenarios, habitat fragmentation and the potential extent of edge effects were assessed for forest and cerrado cover associated with areas of grain or cattle production by calculating a series of simple landscape metrics for each landscape. I assessed quantity (total class 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). All analyses were carried out using Fragstats 3.3 spatial pattern analysis software (McGarigal et al., 2002). The proportion of “edge” vs. “interior” habitat for forest was calculated using edge influence

values derived from empirical observations by researchers studying edge effects related to the effects of fire on forests in the region (Balch, 2008). I applied an edge depth value of 150 m to forest patches adjacent to agricultural areas. This was rounded to 100 m (1 pixel) due to the resolution of the land-cover maps. For forest patches adjacent to cerrado or regenerating forest or cerrado patches, edge influence was considered to be negligible relative to map resolution. Similarly, edge influence depth in cerrado was considered to be negligible as cerrado is a more “open” land-cover type that is well-adapted to regular fire disturbance.

Table C-1. Estimated costs for restoration of native vegetation in legal reserves and riparian zones in the Xingu River headwaters region.

Description of Restoration Method	Legal Reserve (USD ha^{-1})	Riparian Zone (USD ha^{-1})
Natural regeneration, adjacent to forest or cropland	0	0
Natural regeneration, adjacent to active pasture (fencing, fire breaks only)	536	536
Natural regeneration with enrichment, adjacent to active pasture (fencing, fire breaks, enrichment with native seeds)	644	644
Mechanized restoration, adjacent to forest or cropland (grading, leveling, planting native and some exotic seeds)	791	791
Mechanized restoration, adjacent to active pasture (fencing, fire breaks, grading, leveling, planting native and some exotic seeds)	1327	1327
Restoration with seedlings (fencing, fire breaks, seedlings, planting labor, maintenance labor)	NA	3217
2nd year enrichment (seedlings, native seeds, planting labor)		

Exchange rate: 0.53 BRL/USD (August 28, 2009; Source: Instituto Socioambiental & Aliança da Terra)

LIST OF REFERENCES

Abell, R., Allan, J.D., 2002. Riparian shade and stream temperatures in an agricultural catchment, Michigan, USA. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie 28, 232–237.

Achard, F., Defries, R., 2007. Pan-tropical monitoring of deforestation. Environmental Research Letters 2, 045022. doi: 10.1088/1748-9326/2/4/045022

Acevedo, M.A., Villanueva-Rivera, L.J., 2006. Using automated digital recording systems as effective tools for the monitoring of birds and amphibians. Wildlife Society Bulletin 34, 211-214.

Adams, J.B., Sabol, D.E., Kapos, V., (1995) Classification of multispectral images based on fractions of endmembers – application to land-cover change in the Brazilian Amazon 52(2), 137-154.

Ad-Hoc Working Group on Long Term Cooperative Action, 5th Session (AWGLTCA5), 2009. Reducing Emissions from Deforestation and Forest Degradation and the role of Conservation, Sustainable Management of Forests, and the Enhancement of Forest Carbon Stocks. Paper No. 1. Bonn, April 2009, UNFCCC. Accessible at http://unfccc.int/resource/docs/2009/awglca5/eng/misc01a04.pdf (accessed on November 2, 2009)

Agencia Nacional de Águas (ANA), 2009.. *Ottobacias*. Accessible at http://www.ana.gov.br/bibliotecavirtual/ottobacias.asp (Accesed on November 2, 2009)

Agterberg, F.P., Bonham-Carter, G.F., 1990. Deriving weights of evidence from geoscience contour maps for the prediction of discrete events. Proceedings of the 22nd APCOM Symposium, Technical University of Berlin, Berlin, Germany, 381-395.

Akcakaya, H.R., 2000. Viability analyses with habitat-based metapopulation models. Population Ecology 42, 45-53.

Alencar, A., Nepstad D., McGrath, D., Moutinho, D., Pacheco, P., Vera Diaz, M.D.C., and Soares Filho, B., 2004. Desmatamento na Amazônia: Indo Além da "Emergência Crônica". Instituto de Pesquisa Ambiental da Amazônia (IPAM), Belem, Pará, Brasil.

Alencar, A.A.C., Nepstad, D.C., Vera-Diaz, M.D.C., 2006. Forest Understory Fire in the Brazilian Amazon in ENSO and Non-ENSO Years: Area Burned and Committed Carbon Emissions. Earth Interactions 10(6), 1-17.

Allan, J.D., 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology and Systematics 35, 257–284.

Almeida, C.M., Batty, M., Monteiro, A.M.V., Camara, G., Soares-Filho, B.S., Cerqueira, G.C., Pennachin, C.L., 2003. Stochastic cellular automata modeling of urban land use dynamics: empirical development and estimation. Computers, Environment, and Urban Systems 27, 481-509.

Almeida, C.M., Gleriani, J.M., Castejon, E.F., Soares Filho, B.S., 2008. Neural networks and cellular automata for modeling intra-urban land use dynamics. International Journal of Geographical Information Science 22, 943-963.

Alston, L.J., Libecap, G.D., and Mueller, B., 1999. Titles, conflict, and land use: the development of property rights and land reform on the Brazilian Amazon frontier. University of Michigan Press, Ann Arbor.

Ames, B., Keck, M., 1998. The politics of sustainable development: environmental policy making in four Brazilian states. Journal of Interamerican Studies and World Affairs 39 (4), 5-40.

Andreae, M.O., Rosenfeld, D., Artaxo, P., Costa, A.A., Frank, J.P., Longo, K.M., Silva-Dias, M.A.F., 2004. Smoking rain clouds over the Amazon. Science 303, 1337–1342. (doi:10.1126/science.1092779)

Angelsen, A., Streck, C., Peskett, L., Brown, J., Luttrell, C. 2008. What is the right scale for REDD? The implications of national, subnational and nested approaches. Center for International Forestry Research (CIFOR), Bogor, Indonesia. Accessible at http://www.cifor.cgiar.org/carbofor/projects/globalredd/Publications.htm (accessed on November 2, 2009)

Ankersen, T.T., Ruppert, T.R. 2006. Tierra y Libertad: the social function doctrine and land reform in Latin America. Tulane Environmental Law Journal 69, 70-120.

Ascher, W., 1999. Why Governments Waste Natural Resources: Policy Failures in Developing Countries. The Johns Hopkins University Press, Baltimore.

Asner, G.P, Lobell, D.B., 2000. A biogeophysical approach for automated SWIR unmixing of soils and vegetation. Remote Sensing of Environment 74 (1), 99-112.

Asner, G.P., Vitousek, P.M., 2005. Remote analysis of biological invasion and biogeochemical change. Proceedings of the National Academy of Sciences of the United States of America 102 (12), 4383-4386.

Asner, G.P., Martin, R.E., 2009. Airborne spectranomics: mapping canopy chemical and taxonomic diversity in tropical forests. Frontiers in Ecology and the Environment 7 (5), 269-276.

Asner G.P., Knapp D.E., Broadbent E.N., 2005. Selective logging in the Brazilian Amazon. Science 310 (5747), 480-482.

Automated Remote Biodiversity Monitoring Network (ARBIMION), 2009. Overview. University of Puerto Rico, San Juan, Puerto Rico. Accessible at http://arbimon.uprrp.edu/arbimonweb/index.php (accessed on November 2, 2009)

Azevedo, A.A., 2009. Legitimação da insustentabilidade? Análise do Sistema de Licenciamento Ambiental de Propriedades Rurais - SLAPR (Mato Grosso). Ph.D. Dissertation, Federal University of Brasília, Brasília, DF.

Azevedo, A.A., Pasquis, R., 2006. PROJETO DIÁLOGOS. Identificação de temas e espaços de formação e diálogo pelo setor produtivo ligado à cadeia da soja no estado do Mato Grosso. WWF, Brasília. Accessible at www.dialogos.org (accessed on November 2, 2009)

Azevedo-Ramos, C., Kalif, K., de Carvalho, O., 2004. As madeiras e a conservação da fauna. Revista Ciência Hoje 34, 111-120.

Baatz, M., Benz, U., Dehghani, S., 2004. *eCognition: User Guide 4*. Definiens Imaging GmbH, Munich, Germany. 72 pp. Accessible at http://www.gis.unbc.ca/help/software/ecognition4/ELuserguide.pdf (accessed on November 29, 2009)

Baccini, A., Laporte, N., Goetz, S.J., Sun, M., Dong, H.,2008. A first map of tropical Africa's above-ground biomass derived from satellite imagery. Environmental Research Letters 3, 045011.

Baguette, M., Schtickzelle, N., 2003. Local population dynamics are important to the conservation of metapopulations in highly fragmented landscapes. Journal of Applied Ecology 40 (2), 404-412.

Balch, J.K. 2008. Effects of Recurrent Fire on Transitional Forest Dynamics in the Amazon's Wildfire Frontier in Mato Grosso, Brazil. PhD dissertation, Yale University, New Haven, CT.

Ballester, M.V.R., Victoria, D.D.C., Krusche, A.V., Coburn, R., Victoria, R.L., Richey, J.E., Logsdon, M.G., Mayorga, E., Matricardi, E., 2003. A remote sensing/GIS-based physical template to understand the biogeochemistry of the Ji-Paraná river basin (Western Amazon). Remote Sensing of the Environment 87, 429-445.

Banco Nacional de Desenvolvimento (BNDES), 2009. Fundo Amazônia. Accessible at http://www.bndes.gov.br/SiteBNDES/bndes/bndes_pt/Fundo_Amazonia/Fundo/ (accessed November 2, 2009)

Barbier, E.B., Burgess, J.C., 1997. Economic Analysis of Forest Land Use Options. Land Economics 73 (2), 174-195.

Barlow J., 2007. Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. Proceedings of the National Academy of Sciences of the United States of America 104 (47), 18555-18560.

Barlow, J., Haugaasen, T., Peres, C.A., 2002 Effects of ground fires on understory bird assemblages in Amazonian forests. Biological Conservation 105, 157-169.

Barton, D.R., Taylor, W.D., Biette, R.M., 1985. Dimensions of riparian buffer strips required to maintain trout habitat in southern Ontario streams. North American Journal of Fisheries Management 5, 364–378.

Bauer, E., Kohavi, R., 1999. An empirical comparison of voting classification algorithms: Bagging, boosting, and variants. Machine Learning 36, 105-139.

Beaulac, M.N., Reckhow, K.H., 1982. An examination of land use-nutrient export relationships. Water Resources Bulletin 18, 1013-1024

Benatti, J.H., 2003. Direito de Propriedade e Proteção Ambiental no Brasil: apropriação e o uso dos recursos naturais no imóvel rural. Ph.D. Dissertation, Center for Advanced Amazonian Studies, Federal University of Pará, Belém, Brasil.

Beissinger, S.R., Westphal, M.I., 1998. On the use of demographic models of population viability in endangered species management. Journal of Wildlife Management 62 (3), 821-841.

Beissinger, S.R., McCullough, D.R., 2002. Population viability analysis. University of Chicago Press, Chicago.

Benstead, J.P., Pringle, C.M., 2004. Deforestation alters the resource base and biomass of endemic stream insects in eastern Madagascar. Freshwater Biology 49, 490–501.

Bierregaard, R.O., 2001. Lessons from Amazonia: the ecology and conservation of a fragmented forest. Yale University Press, New Haven.

Biggs, T.W., Dunne, T., Domingues, T. F., Martinelli, L.A., 2002. The relative influence of natural watershed properties and human disturbance on stream solute concentrations in the southwestern Brazilian Amazon basin. Water Resources Research 38, 10.1029/2001WR000271.

Biggs, T.W., Dunne, T., Martinelli,L.A., 2004. Natural controls and human impacts on stream nutrient concentrations in a developing region of the Brazilian Amazon basin, Biogeochemistry 68, 227-257.

Blackard, J.A., Finco, M.V., Helmer, E.H., 2008. Mapping U.S. forest biomass using nationwide forest inventory data and moderate resolution information. Remote Sensing of Environment 112, 1658-1677.

Bojsen, B.H., Barriga R., 2002. Effects of deforestation on fish community structure in Ecuadorian Amazon streams. Freshwater Biology 47, 2246–2260.

Bonham-Carter, G., 1994. Geographic Information Systems for Geoscientists: Modelling with GIS. Pergamon, Kidlington, United Kingdom. 398 pp.

Bonan, G.B., Pollard, D., Thompson, S.L., 1992. Effects of boreal forest vegetation on global climate. Nature 359 (6397), 716-718.

Bormann, F.H., Likens, G.E., 1979. Pattern and Process in a Forested Ecosystem. Springer-Verlag, New York, 253 pp.

Bosch, J.M., Hewlett, J.D., 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. Journal of Hydrology 103, 323-333.

Bowman, M.S., Amacher, G.S., Merry F.D., 2008. Fire use and prevention by traditional households in the Brazilian Amazon. Ecological Economics 67 (1), 117-130.

Breiman, L., 2001. Random forests. Machine Learning 45**,** 5–32.

Brito, B., and Barreto, P. A Eficácia da aplicação da lei de crimes ambientais pelo IBAMA para proteção de florestas no Pará. IMAZON, Belém, Brasil, 33 pp.

Brown, D., Seymour, F., Peskett, L., 2008. How do we achieve REDD co-benefits and avoid doing harm? In Moving Ahead with REDD (Ed. A. Angelsen). CIFOR, Bogor, Indonesia, 107-118. Accessible at http://www.cifor.cgiar.org/publications/pdf_files/Books/BAngelsen0801.pdf (accessed on November 2, 2009)

Bruijnzeel, L.A., 1990. Hydrology of moist tropical forests and effects of conversion: a state of knowledge review. UNESCO, 224.

Burcham, J., 1988. Fish communities and environmental characteristics of two lowland streams in Costa Rica. Revista de Biologia Tropical 36, 273–285.

Busch, J., Strassburg, B., Cattaneo, A., Comparing REDD mechanism design options with an open source economic model. Ecological Economics, in review.

Camarga, 2008. http://www.inesc.org.br/noticias/noticias-gerais/2008/julho/bancada-ruralista-a-um-passo-da-vitoria/ (accessed on November 2, 2009)

Campari, J.S., 2005. The Economics of Deforestation in the Amazon: Dispelling The Myths. Edward Elgar Publishing.

Campos, M.T., Nepstad, D.C., 2006. Smallholders, The Amazon's New Conservationists. Conservation Biology 20 (5), 1553-1556.

Carpenter, S.R., Caraco, N.F., Correll, D.L., Howarth, R.W., Sharpley, A.N., Smith, V.H., 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications 8, 559-568.

Carter, N., 2001. The Politics of the Environment: Ideas, Activism, Policy. The Cambridge University Press, Cambridge.

Chan, K.M.A., Pringle, R.M., Ranganathan, J., 2007. When agendas collide: Human welfare and biological conservation. Conservation Biology 21 (1), 59-68.

Chaves, J., 2008. Land management impacts on runoff sources in small Amazon watersheds, Hydrological Processes 22, 1766-1775.

Chomitz, K.M., 2004. Transferable development rights and forest protection: An exploratory analysis. International Regional Science Review 27 (3), 348-373.

Climate, Community and Biodiversity Alliance (CCBA), 2008. Climate, Community and Biodiversity Project Design Standards Second Edition. CCBA, Arlington, VA. Accessible at http://www.climate-standards.org/standards/pdf/ccb_standards_second_edition_december_2008.pdf (accessed on November 2, 2009)

Cochrane, M.A., 2003. Fire Science for Rainforests. Nature 421, 913-919.

Cochrane, M.A., Souza, C.M., 1998. Linear mixture model classification of burned forests in the Eastern Amazon. International Journal of Remote Sensing 19 (17), 3433-3440.

Coe, M.T., 2000. Modeling terrestrial hydrological systems at the continental scale: Testing the accuracy of an atmospheric GCM. Journal of Climate 13, 686-704.

Coe, M.T., Costa,M.H., Soares-Filho,B.S., 2009. The Influence of historical and potential future deforestation on the stream flow of the Amazon River - Land surface processes and atmospheric feedbacks. Journal of Hydrology doi:10.1016/j.jhydrol.2009.02.043.

Colby, K.E., 2003. Brazil and the MST: land reform and human rights. New York International Law Review 1,10-11.

Cook, W.M., Lane, K.T., Foster, B.L., Holt, R.D. 2002. Island theory, matrix effects and species richness patterns in habitat fragments. Ecology Letters 5, 619–623.

Congresso Nacional (CN), 2000. Commisão Mista destinada a examinar e emitir parecer sobre a Medida Provisória Nº 1.956-48. Congresso Nacional, Brasilia, April 17. Accessible at http://webthes.senado.gov.br/sil/Comissoes/Mistas/MP/Comissoes/20001956048/Atas/20000417RO006.rtf (accessed on November 2, 2009)

Costa, M.H., Foley,J.A., 1997. Water balance of the Amazon Basin: Dependence on vegetation cover and canopy conductance. Journal of Geophysical Research-Atmospheres 102 (D20), 23,973-23,989.

Costa, M.H., Foley, J.A., 2000. Combined effects of deforestation and doubled atmospheric CO_2 concentrations on the climate of Amazonia, Journal of Climatology 13, 18– 34.

Costa, M.H., Botta, A., Cardille, J.A., 2003. Effects of large-scale changes in land cover on the discharge of the Tocantins River, southeastern Amazonia. Journal of Hydrology 283, 206-217. dio-10.1016/S0022-1694(03)00267-1.

Costa, M.H., Yanagi, S.N.M., Souza, P.J.O.P., Ribeiro, A., Rocha, E.J.P., 2007. Climate change in Amazonia caused by soybean cropland expansion, as compared to caused by pastureland expansion. Geophysical Research Letters 34, L07706. doi:10.1029/2007GL029271

Costanza, R., D'arge, R., De Cross, R., Farber, S., Hannon, B., Raskin, R.C., Sutton, P., 1997. The value of the world's ecosystem services and natural capital. Nature 338: 253-260.

Cox, P.M., Betts, R.A., 2000. Acceleration of global warming due to carbon cycle feedbacks in a coupled climate model. Nature 408, 184-187.

Cumberlidge, N., Ng, P.K.L., Yeo, D.C.J., Magalhães, C., Campos, M.R., Alvarez, F., Naruse, T., Daniels, S.R., Esser, L.J., Attipoe, F.Y.K., Clotilde-Ba, F.L., Darwall, W., McIvor, A., Baillie, J.E.M., Collen, B., Ram, M., 2009. Freshwater crabs and the biodiversity crisis: Importance, threats, status, and conservation challenges. Biological Conservation 142:1665–1673.

Curran, L.M., Trigg, S.N., McDonald, A.K., Astiani, D., 2004. Lowland forest loss in protected areas of Indonesian Borneo. Science 303 (5660),1000-1003.

Daly, H.E., Farley, J., 2003. Ecological Economics: Principles and Applications. Island Press, Washington, DC.

Daily, G. C., 1997. Introduction: What are ecosystem services? In Nature's Services: Societal Dependence on Nature's Ecosystems (Ed. G. Dailey). Island Press, Washington, D.C.

Davidson, E.A., Neill, C., Krusch, A.V., Ballester, V.V.R., Markewitz, D., Figueiredo,R.O., 2004. Loss of nutrients from terrestrial ecosystems to streams and the atmosphere following land use change in Amazonia. In Ecosystems and Land Use Change (Eds. DeFries, R., Asner, G., Houghton R.). Geophysical Monograph Series 153, 147-158. American Geophysical Union, Washington, D.C.

Davidson, E.A., de Carvalho, C.J.R., Figueira, A.M., Ishida, F.Y., Ometto, J.P.B., Nardoto, G.B., Sabá, R.T., Hayashi, S.N., Leal, E.C., Vieira, I.C.G., Martinelli, L.A., 2007. Recuperation of nitrogen cycling in Amazonian forests following agricultural abandonment. Nature 447, 995-998.

Daviet, F., McMahon, H., Stolle, F., 2007. REDD Flags: what we need to know about the options. WRI, Washington, D.C. Accessible at http://pdf.wri.org/redd-flags.pdf (accessed on November 2, 2009)

Dean, W., 1995. With Broadax and Firebrand: Destruction of the Brazilian Atlantic Forest. University of California Press, Berkeley, CA.

Delire, C., Behling, P., Coe, M.T., Foley, J.A., Jacob, R., Kutzbach, J., Liu, Z., Vavrus, S., 2001. Simulated response of the atmosphere-ocean system to deforestation in the Indonesian Archipelago. Geophysical Research Letters 28 (10), 2081-2084.

Dickinson, R.E., Henderson-Sellers, A., 1988. Modelling tropical deforestation: a study of GCM land-surface parameterizations. Quaternary Journal of the Royal Metereological Society 114, 439-62.

Dietterich, T.G., 2000. An experimental comparison of three methods for constructing ensembles of decision trees: bagging, boosting, and randomization. Machine Learning 40, 139-157.

Dietz, J.M., Dietz, L.A., 1994. The effective use of flagship species for conservation of biodiversity: the example of lion tamarins in Brazil. In Creative Conservation (Eds. P. Olney, G. Mace), 145-161.

Dietz, T., Ostrom, E., Stern, P.C., 2003. The struggle to govern the commons. Science 302, 1907-1912.

Dooley, K., Griffiths, T., Leake, H., Ozinga, S., 2008. Cutting Corners World Bank's forest and carbon fund fails forests and peoples. FERN / Forest Peoples Programme, London. Accessible at http://www.fern.org/media/documents/document_4312_4313.pdf (accessed on November 2, 2009)

Engel, S., Pagiola, S., Wunder, S., 2008. Designing payments for environmental services in theory and practice: an overview of the issues. Ecological Economics 65 (4), 663-674.

Fearnside, P.M., 2003. Deforestation control in Mato Grosso: a new model for slowing the loss of Brazil's Amazon forest. Ambio 32, 343-345.

Fearnside, P.M., 2005. Deforestation in Brazilian Amazonia: History, Rates, and Consequences. Conservation Biology 19 (3), 680–688.

Fearnside, P.M. 2008. The roles and movements of actors in the deforestation of Brazilian Amazonia. Ecology and Society 13: 23.

Fimbel, R.A., Grajal, A., Robinson, J.G., 2001. Logging-wildlife issues in the tropics: an overview. In The Cutting Edge: Conserving Wildlife in Logged Tropical Forests (Eds. R.A. Fimbel, A. Grajal, J.G. Robinson), 3-9, Columbia University Press, New York.

Fischer, J., Lindenmayer, D.B., 2006. Beyond fragmentation: the continuum model for fauna research and conservation in human-modified landscapes. Oikos 112, 473–480.

Flecker, A. S. 1992. Fish trophic guilds and the structure of a tropical stream: weak direct vs. strong indirect effects. Ecology 73, 927–940.

FNP Consultoria e Comércio (FNP), 2005. Análise de Mercado de Terras: Relatório Bimestral. Accessible at www.fnp.com.br (accessed on December 1, 2009)

Foley, J.A., Prentice, I.C., Ramankutty, N., Levis, S., Pollard, D., Sitch, S., Haxeltine, A., 1996. An integrated biosphere model of land surface processes, terrestrial carbon balance, and vegetation dynamics. Global Biogeochemical Cycles 10 (4), 603-628.

Folha de São Paulo, October 15, 2009. Ruralistas obtêm comando de debate sobre Código Florestal. Accessible at http://www1.folha.uol.com.br/fsp/brasil/fc1510200915.htm (accessed on October 29, 2009)

Food and Agriculture Organization (FAO), 2009. The State of Agricultural Commodity Markets 2009: High food prices and the food crisis—experiences and lessons learned. Accessible at http://www.fao.org/docrep/012/i0854e/i0854e00.htm (accessed on November 29, 2009)

Forest Carbon Partnership Facility (FCPF), 2009. About the FCPF. Accessible at http://www.forestcarbonpartnership.org/fcp/node/12 (accessed on November 2, 2009)

Friends of the Earth International (FOEI), 2008. REDD myths: a critical review of proposed mechanisms to reduce emissions from deforestation and degradation in developing countries. FOEI, London. www.foei.org.uk (accessed on November 2, 2009)

Gibbs, H.K, Brown, S.L., 2007. Monitoring and estimating tropical forest carbon stocks: making REDD a reality. Environmental Research Letters 2, 045023. doi: 10.1088/1748-9326/2/4/045023 (accessed on November 2, 2009)

Goetz, S.J., Bunn, A.G., Fiske, G., Houghton, R.A., 2005. Satellite-observed photosynthetic trends across boreal North America associated with climate and fire disturbance. Proceedings of the National Academy of Sciences 102, 13521-13525.

GOFC-GOLD, 2009. Reducing greenhouse gas emissions from deforestation and degradation in developing countries: a sourcebook of methods and procedures for monitoring, measuring and reporting, GOFC-GOLD Report version COP14-2, GOFC-GOLD Project Office, Natural Resources Canada, Alberta, Canada. Accessible at http://www.gofc-gold.uni-jena.de/redd/sourcebook/Sourcebook_Version_July_2009_cop14-2.pdf (accessed on November 2, 2009)

Goldammer, J.G., ed., 1990. Fire in the tropical biota. Ecosystem processes and global challenges. Ecological Studies 84, Springer-Verlag, Berlin-Heidelberg-New York, 497 p.

Goodwin, R.A., Pandey, V., Kiker, G.A., 2007. Spatially-explicit population models with complex decisions.. In Environmental Security in Harbors and Coastal Areas, (Linkov, I., Kiker, G.A., et al. Eds.), 293-306, Springer Netherlands.

Goulding, M., 1980. The fishes and the forest: explorations in Amazonian natural history. University of California Press, Berkeley, California.

Government of Norway (GON), 2009. International Climate and Forest Initiative Funding Scheme. Accessible at http://www.norway.org.vn/development/Environment/Norway+Intl+Climate+and+Forest+Initiative.htm (accessed on November 2, 2009)

Government of Brazil (GOB), 2008. Política Nacional sobre Mudança do Clima. Accessible at http://www.mma.gov.br/estruturas/imprensa/_arquivos/96_11122008040728.pdf (accessed on November 2, 2009)

Government of Mato Grosso (GOMT), 2009. Reference Document for the Development of Mato Grosso State's REDD Program. November 26, 2009, Cuiabá, Mato Grosso.

Green, G.M., Schweik, C., Hanson, M., 2002. Radiometric calibration of LANDSAT multi-spectral scanner and thematic mapper images:Guidelines for the global change community. Center for the Study of Institutions, Population, and Environmental Change (CIPEC), Indiana University, Bloomington, Indiana.

Gregory, S.V., Swanson, F. J., McKee, W.A., Cummins, K.W., 1991. An ecosystem perspective of riparian zones. BioScience 41, 540–551.

Gullison, R.E., Frumhoff, P.C., Canadell, J.G., 2007. Tropical forests and climate policy. Science 316 (5827), 985-986.

Guimaraes, J. Almeida, O., 2007. Análise de custos de produção da pecuária e da soja na região nordeste do Mato Grosso. CNPq/ IPAM, Belém, PA.

Gustafson, E.J., Crow, T.R., 1996. Simulating the effects of alternative forest management strategies on landscape structure. Journal of Environmental Management 46, 77-94.

Hagen, A., 2003. Fuzzy set approach to assessing similarity of categorical maps. International Journal of Geographic Information Science 17, 235-249.

Hagen-Zanker, A., 2006. Map comparison methods that simultaneously address overlap and structure. Journal of Geographic Systems 8, 165-185.

Hansen, M.C., Stehman, S.V., Potapov, P.V., 2008. Humid tropical forest clearing from 2000 to 2005 quantified by using multitemporal and multiresolution remotely sensed data. Proceedings of the National Academy of Sciences 105, 9439-9444.

Hardin, G., 1968. The Tragedy of the Commons. Science 162, 1243-1248.

Hecht, S., 2005. Soybeans, Development and Conservation on the Amazon Frontier. Development and Change 36(2), 375–404.

Herold, M., Johns, T. 2007. Linking requirements with capabilities for deforestation monitoring in the context of the UNFCCC-REDD process. Environmental Research Letters 2, 045025, doi:10.1088/1748-9326/2/4/04502

Hochstettler, K., Keck, M.E., 2007. Greening Brazil: Environmental Activism in State and Society. Duke University Press, Durham, NC.

Hoffmann, W.A., Orthen, B., 2003. Comparative fire ecology of tropical savanna and forest trees. Functional Ecology 17, 720-726.

Holdsworth, A.R., Uhl, C., 1997. Fire in Amazonian selectively logged rain forest and the potential for fire reduction. Ecological Applications 7, 713–725.

Holmes, T.P., Blate, G.M., Zweede, J.C., Pereira, R., Barreto, P., Boltz, F., Bauch, R., 2002. Financial and ecological indicators of reduced impact logging performance in the eastern Amazon. Forest Ecology and Management 163 (1-3), 93-110.

Hölscher, Sá, D., T.D.d.A., Bastos, T.X., Denich, M., Fölster, H., 1997. Evaporation from young secondary vegetation in eastern Amazônia. Journal of Hydrology 193, 293-305.

Hooijer, A., Silvius, M., 2006. PEAT-CO_2, Assessment of CO_2 emissions from drained peatlands in SE Asia. Delft Hydraulics report Q3943, 7. Accessible at www.wetlands.org/ckpp/publication.aspx?ID=f84f160f-d851-45c6-acc4-d67e78b39699 (accessed on November 2, 2009)

Houghton, R., Skole, D., Nobre, C., Hackler, J., Lawrence, K., Chementowski, W., 2000. Annual fluxes of carbon from deforestation and regrowth in the Brazilian Amazon. Nature 403: 301–304.

Howarth, R.W., Billen, G., Swaney, D., Townsend, A., Jaworski, N., Lajtha, K., Downing, J.A., Elmgren, R., Caraco, N., Jordan, T., Berendse, F., Freney, J., Kudeyarov, V., Murdoch, P., Zhao-Liang, Z., 1996. Regional nitrogen budgets and riverine N & P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. Biogeochemistry 35, 75-139

Howe, H.F., Smallwood,J.F., 1982. Ecology of seed dispersal. Annual Review of Ecology and Systematics 13, 201-228.

HR 2454, American Clean Energy and Security Act of 2009, 111th Congress, 2009.

Instituto Brasileiro de Geografia e Estatística (IBGE), 1981. Projeto RADAMBRASIL, Vol. 22, Folha SC.22 Tocantins. Brasilia, Brasil.

Instituto Nacional de Pesquisa Espaciais (INPE), 2009. Projeto Desmatamento (Prodes): Monitoramento Da Floresta Amazonica Por Satelite. Accessible at http://www.obt.inpe.br/prodes/.(accessed on November 2, 2009)

Instituto de Pesquisa Ambiental da Amazônia (IPAM), 2009. Xingu Social Carbon Program: Potential for REDD. Presentation at the Forum for REDD Readiness South-South Collaboration Workshop, March 15-20, Manaus, Brazil. Accessible at http://www.whrc.org/Policy/REDD/Reports/FlaviaGabriela-Introduction_Ipam.pdf (accessed on November 2, 2009)

Instituto Sociambiental (ISA), 2005. Campanha `Y Ikatu : Mobilização para salvar nascentes do Rio Xingu. Accessible at http://www.socioambiental.org/inst/camp/xingu/enc/acarta.html (accessed on November 2, 2009)

Intergovernmental Panel on Climate Change (IPCC), 2007. Climate change 2007: the physical science basis. Contribution of working group I to the fourth assessment report of the IPCC. Cambridge University Press, Cambridge.

Jansen, P.A., Zuidema, P.A., 2001. Logging, seed dispersal by vertebrates, and natural regeneration of tropical timber trees. In The Cutting Edge: Conserving Wildlife in Logged Tropical Forests (Eds. R.A. Fimbel, A. Grajal, J.G. Robinson), 35-59. Columbia University Press, New York.

Janssen, L.L.F., Jaarsma, M.N., van der Linden, E.T.M., 1990. Integrating topographic data with remote sensing for land-cover classification. Photogrammetric Engineering and Remote Sensing 56, 1503-1506.

Jaramillo, C., Kelly, T., 1997. Deforestation and Property Rights in Latin America. Inter-American Development Bank, Washington, DC. Accessible at: http://www.iadb.org/sds/publication/publication_1030_e.htm (accessed on November 2, 2009)

Jensen, J.R., 2004. Introductory Digital Image Processing, 3rd Ed. Prentice Hall Series in Geographic Information Science (Ed. K.C. Clark), Prentice Hall, Upper Saddle River, New Jersey. 526 pp.

Jepson, W., 2006. Private agricultural colonization on a Brazilian frontier, 1970-1980. Journal of Historical Geography 32, 839-863.

Jipp, P., Nepstad, D., Cassel, K., de Carvalho, C.R., 1998. Deep soil moisture storage and transpiration in forests and pastures of seasonally-dry Amazônia. Climatic Change 39, 395-412.

Johns, T., Merry, F., Stickler, C., Nepstad, D., Laporte, N., Goetz, S., 2008. A three-fund approach to incorporating government, public and private forest stewards into a REDD funding mechanism. International Forestry Review 10 (3), 458-464.

Justice, C.O., Giglio, L., Korontzi, S., Owens, J., Morisette, J.T., Roy, D., Descloitres, J., Alleaume, S., Petitcolin, F., Kaufman, Y., 2003. The MODIS fire products. Remote Sensing of Environment 83, 244-262.

Karr, J.R. 1991. Biological integrity: A long-neglected aspect of water resource management. Ecological Applications 1, 66–84.

Karr, J. R., Schlosser, I.J., 1978. Water resources and the land–water interface. Science 201, 229–234.

Kauffman, J.B., Cummings, D.L., Ward, D.E., Babbitt, R., 1995. Fire in the Brazilian Amazon: biomass, nutrient pools, and losses in slashed primary forests. Oecologia 104, 397-409.

Kellndorfer, J.M., Shimada, M., Rosenqvist, A., Walker, W.S., Kirsch, K, Nepstad, D.C., Laporte, N., Stickler, C.M., Lefebvre, P., 2007. New Eyes in the Sky: Cloud-Free Tropical Forest Monitoring for REDD with the Japanese Advanced Land Observation Satellite (ALOS). Report for the United Nations Framework Convention on Climate Change (UNFCCC) Conference of the Parties (COP), Thirteenth session, 3-14 December, The Woods Hole Research Center, Falmouth, MA. Accessible at http://whrc.org/BaliReports/assets/Bali_ALOS.pdf (accessed on November 2, 2009)

Kindermann, G., Obersteiner, M., Sohngen, B., Sathaye, J., Andrakso, K., Rametsteiner, E., Schlamadinger, B., Wunder, S., Beach, R., 2008. Global cost estimates for reducing carbon emissions through avoided deforestation. Proceedings of the National Academy of Science of the Unites States of America. http://www.pnas.org/content/105/30/10302.full (accessed on November 2, 2009)

Krishnaswamy, J., Bawa, K.S., Ganesahiah, K.N., 2009. Quantifying and mapping biodiversity and ecosystem services: Utility of a multi-season NDVI based Mahalanobis distance surrogate. Remote Sensing of Environment 113 (4), 857-867.

Kucharik C.J., Foley J.A., Delire C., Fisher V.A., Coe M.T., Lenters J., Young-Molling C., Ramankutty N., Norman J.M., Gower S.T., 2000. Testing the performance of a dynamic global ecosystem model: Water balance, carbon balance and vegetation structure. Global Biogeochemical Cycles 14 (3), 795-825.

Lal, R., 1995. Global soil erosion by water and carbon dynamics. In Soils and global change (Eds. R. Lal, J. Kimble, E. Levine, B.A. Stewart), 131-142. CRC/Lewis Publishers, Boca Raton, FL.

Landell-Mills, N., Porras, T.I., 2002. Silver bullet or fools' gold? A global review of markets for forest environmental services and their impact on the poor. Instruments for sustainable private sector forestry series. International institute for Environment and Development, London. www.iied.org (accessed on November 2, 2009)

Lang, C., 2009. Halt climate change. Halt forest destruction. Halt plantations. REDD Monitor, June 10. Accessible at http://www.redd-monitor.org/2009/06/10/halt-climate-change-halt-forest-destruction-halt-plantations/ (accessed on November 2, 2009)

Larsen, J., Heilmayr, R., 2009. Emissions Reductions under the American Clean Energy and Security Act of 2009. World Resources Institute, Washington, D.C. Accessible at http://pdf.wri.org/usclimatetargets_2009-05-19.pdf (accessed on November 26, 2009)

Latrubesse, E.M., Chen, Z., Stevaux, J.C. Short and long term processes, landforms and responses in large rivers. Geomorphology, in press. doi:10.1016/j.geomorph.2009.03.018

Laurance, W.F., Laurance, S.G.W., 1996. Responses of five arboreal marsupials to recent selective logging in tropical Australia. Biotropica 28, 310-322.

Laurance, W.F., Bierregaard, R.O., Jr., eds. 1997. Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities. University of Chicago Press, Chicago.

Laurance, W. F., Cochrane, M.A., Bergen, S., Fearnside, P.M., Delamonica, P., Barber, C., D'Angelo, S., Fernandes, T., 2001. The future of the Brazilian Amazon. Science 291, 438-439.

Lentini, M., Veríssimo, A., Sobral,L., 2003. Fatos Florestais da Amazônia 2003. Imazon, Belém, Brasil. Accesible at: http://www.imazon.org.br/upload/im_livros_002.pdf (accessed on November 2, 2009)

Li,W., Fu, R., Dickinson, R.E., 2006. Rainfall and its seasonality over the Amazon in the 21st century as assessed by the coupled models for the IPCC AR4. Journal of Geophysical Research 111, D02111, doi:10.1029/2005JD006355.

Liaw, A., Wiener, M., 2002. Classification and regression by randomForest. R News 2/3, 18-22.

Lima, A., Irigaray, C.T., Silva, R.T., Guimaraes, S., Araujo, S., 2005. Sistema de Licenciamento Ambiental em Propriedades Rurais do Estado de Mato Grosso: Análise de Lições na Sua Implementação (Relatório Final). Ministério do Meio Ambiente/Secretaria de Coordenação da Amazônia/Programa Piloto para a Proteção das Florestas Tropicais do Brasil/Projeto de Apoio ao Monitoramento e Análise (AMA), Brasilia, July. (Projeto PNUD: BRA 98/0005).

Lindenmayer, D.B., Margules, C.R., Botkin, D.R., 2000. Indicators of Biodiversity for Ecologically Sustainable Forest Management. Conservation Biology 14 (4), 941–950. doi:10.1046/j.1523-1739.2000.98533.x

Lorion, C.M., Kennedy, B.P., 2009. Relationships Between Deforestation, Riparian Forest Buffers And Benthic Macroinvertebrates In Neotropical Headwater Streams. Freshwater Biology 54, 165–180.

Lorion, C. M., Kennedy, B.P., 2009. Riparian forest buffers mitigate the effects of deforestation on fish assemblages in tropical headwater streams. Ecological Applications 19 (2), 468–479.

Lowrance, R., Altier, L.S., Newbold, J.D., Schnabel, R.R., Groffman, P.M., Denver, J.M., Correll, D.L., Gilliam, J.W., Robinson, J.L., BRinsfield, R.B., 1997. Water quality functions of riparian forest buffers in Chesapeake Bay watersheds. Environmental Management 21, 687–712.

Malhi, Y., Timmons, R.J., 2008. Climate change, deforestation and the fate of the Amazon. Science 319, 169–172. doi:10.1126/science.1146961

Malhi, Y., Aragao, L.E.O.C., Galbraith, D., 2009. Exploring the likelihood and mechanism of a climate-change-induced dieback of the Amazon rainforest. Proceedings of the National Academy of Science of the United States of America

Mansourian, S., Vallauri, D., Dudley, N., 2005. Forest restoration in landscapes: beyond planting trees. Springer, New York.

Markewitz, D., Davidson, E. A., Figueiredo, R., Victoria, R. L., Krusche, A.V., 2001. Control of cation concentrations in stream waters by surface soil processes in an Amazonian watershed. Nature 410, 802-805.

May, P.H., 1999. Natural Resource Valuation and Policy in Brazil. Columbia University Press, New York.

McFarland, A.M.S., Hauck, L.M., 1999 Relating agricultural land uses to in-stream stormwater quality. Journal of Environmental Quality 28, 836-844.

McGarigal K., Cushman, S.A., Neel, M.C., Ene, E., 2002. FRAGSTATS: Spatial Pattern Analysis Program for Categorical Maps. University of Massachusetts, Amherst. Accesible at: www.umass.edu/landeco/research/fragstats/fragstats.html (accessed on November 2, 2009)

McGrath, D.A., Smith, C.K., Gholz, H.L., Oliveira, F.D.A. 2001. Effects of land-use change on soil nutrient dynamics in Amazônia, Ecosystems 4, 625-645.

Medida Provisoria (MP) 2.166-67, 2001. Presidência da República, Casa Civil, Brasilía, Brasil, August 24, 2001. Accessible at http://www.planalto.gov.br/ccivil_03/MPV/2166-67.htm#art1 (accessed on November 29, 2009)

Melo, C.E., Machado, F.A., Pinto-Silva, V., 2003. Diversidade de peixes em um córrego de Cerrado no Brasil Central. Brazilian Journal of Ecology 1-2, 17-23.

Mendonça, F.P., Magnusson, W.E., Zuanon, J. 2005. Relationships Between Habitat Characteristics and Fish Assemblages in Small Streams of Central Amazonia. Copeia 2005 (4), 751–764

Meridian Institute, 2009. Reducing Emissions form Deforestation and Degradation (REDD): an options assessment report. Prepared for the Government of Norway. Meridian Institute, Washington, DC. Accessible at http://www.REDD-OAR.org (accessed on November 2, 2009)

Merry, F.D., Soares-Filho, B.S., Nepstad, D.C., Amacher, G., Rodrigues, H.O., 2009. Balancing conservation and economic sustainability: the future of the Amazon timber industry. Environmental Management 44 (3), 395-407.

Ministério da Ciência e Tecnologia (MCT), 2009. Inventário Brasileiro das Emissões e Remoções Antrópicas de Gases de Efeito Estufa. Accessible at http://mct.gov.br (accessed November 25, 2009)

Miles, L., Kapos, V., 2008. Reducing Greenhouse Gas Emissions from Deforestation and Forest Degradation: Global Land-Use Implications. Science 320 (5882), 1454-1455.

Molnar, A., Scherr, S., Khare, A., 2003. Who conserves the world's forests? A new assessment of conservation and investment trends. Ecoagriculture Partners, Washington, D.C.

Montgomery, D.R., 2007. Soil erosion and agricultural sustainability. Proceedings of National Academy of Sciences 104 (33), 13268–13272.

Moorcroft, P.R., Hurtt, G.C., Pacala, S.W., 2001. A method for scaling vegetation dynamics: the ecosystem demography model (ED). **Ecological Monographs 71**, 557-585.

Moraes, J.M. de, Schuler, A.E., Dunne, T., Figueiredo, R. de O., Victoria, R.L., 2006. Water storage and runoff processes in plinthic soils under forest and pasture in eastern Amazonia. Hydrological Processes 20, 2509-2526.

Moreira, A.G., 2000. Effects of fire protection on savanna structure in Central Brazil. Journal of Biogeography 27, 1021-1029.

Myers, N., 1988. Threatened biotas: “Hot spots” in tropical forests. The Environmentalist 8, 1–20.

Myers, N., 1990. The biodiversity challenge: Expanded hot spots analysis. The Environmentalist 10, 243–256.

Nagendra, H., Rocchini, D., 2008. High resolution satellite imagery for tropical biodiversity studies: the devil is in the detail. Biodiversity and Conservation 17 (14), 3431-3442.

Navulur, K., 2007. Multispectral Image Analysis Using the Object-Oriented Paradigm. CRC Press, Boca Raton, Florida. 184 pp.

Naiman, R.J., Décamps, H., 1997. The ecology of interfaces: riparian zones. Annual Review of Ecology and Systematics 28, 621–658.

Neeff, T., Ascui, F., 2009. Lessons from carbon markets for designing an effective REDD architecture. Climate Policy 9 (3), 306-315.

Neill C., Davidson, E.A., 2000. Soil carbon accumulation or loss following deforestation for pasture in the Brazilian Amazon. In Global climate change and tropical ecosystems (Eds. Lal, R. J.M. Kimble, B.A. Stewart), 197-211.CRC Press, Boca Raton.

Neill, C., Deegan, L.A., Thomas, S.M., Cerri, C.C., 2001. Deforestation for pasture alters nitrogen and phosphorus in soil solution and streamwater of small Amazonian watersheds. Ecological Applications 11, 1817-1828.

Neill, C., Deegan, L.A., Thomas, S.M., Haupert, C.L., Krusche, A.V., Ballester, V.M., Victoria, RL., 2006. Deforestation alters channel hydraulic and biogeochemical characteristics of small lowland Amazonian streams. Hydrological Processes 20, 2563-2580.

Neill C., Piccolo, M.C., Steudler, P.A., Melillo, J.M., Feigl, B.J., Cerri, C.C., 1995. Nitrogen dynamics in soils of forests and active pastures in the Western Brazilian Amazon basin, Soil Biology and Biochemistry 27, 1167-1175.

Nepstad, D., de Carvalho, C.R., Davidson, E., Jipp, P., Lefebvre, P., Hees Negreiros, G., Silva, E., Stone, T., Trumbore, S., Vieira, S., 1994. The role of deep roots in the hydrologic and carbon cycles of Amazonian forests and pastures. Nature 372, 666-669.

Nepstad, D.C., Alencar, A.C., McGrath, D.G., Moutinho, P., Barros, A.C. 2001. Road paving, fire regime feedbacks, and the future of Amazon forests. Forest Ecology and Management 154, 395–407. doi:10.1016/S0378-1127(01)00511-4

Nepstad, D.C., Schwartzmann, S., Bamberger, B., Santilli, M., Ray, D., Schlesinger, P., Lefebvre, P., Alencar, A., Prinz, E., Fiske, G., Rolla, A., 2006. Inhibition of Amazon deforestation and fire by parks and indigenous lands. Conservation Biology 20 (1), 65-73.

Nepstad, D.C., Soares-Filho, B., Merry, F., Moutinho, P., Rodrigues, H.O., Bowman, M., Schwartzman, S., Almeida, O., Rivero, S., 2007a. The Costs and Benefits of Reducing Carbon Emissions from Deforestation and Forest Degradation in the Brazilian Amazon. The Woods Hole Research Center/Instituto de Pesquisa Ambiental da Amazonia, Falmouth, MA.

Nepstad, D.C., Carvalho Jr., O., Carter, J., Moita, A., Neu, V., Cardinot, G., 2007b. Manejo e Recuperação de Mata Ciliar em Regiões Florestais da Amazônia. (Management and restoration of riparian zone forests of Amazon forest regions).Instituto de Pesquisa Ambiental da Amazônia (IPAM), Belém, Brazil. 68 pp.

Nepstad, D.C., Stickler, C.M., 2008. Managing the Tropical Agriculture Revolution Journal of Sustainable Forestry 27 (1-2), 43 – 56.

Nepstad, D.C., Stickler, C.M., Soares Filho, B.S., Merry, F., 2008. Ecological, economic, and climatic tipping points of an Amazon forest dieback. Philosophical Transactions of the Royal Society B 363, 1737.

Nepstad, D.C., Soares, B.S., Merry, F.D., Lima, A., Moutinho, P., Carter, J., Bowman, M.S., Cattaneo, A., Rodrigues, H., Schwartzman, S., McGrath, D.G., Stickler, C., Lubowski, R., Piris-Cabezas, P., Rivero, S., Alencar, A., Stella, O., Almeida, O. 2009. The end of deforestation in the Brazilian Amazon. Science 326, 1350-1351.

Nobre, C.A., Sellers, P.J., Shulka, J., 1991. Amazonian deforestation and regional climate change. Journal of Climatology 4, 957–988.

Noss, R., 1990. Indicators for monitoring biodiversity. A hierarchical approach. Conservation Biology 4, 355-364.

Oliveira, P.J.C., Asner, G.P., Knapp, D.E., Almeyda, A., Galvan-Gildemeister, R., Keene, S., Raybin, R.F., Smith, R.C., 2007. Land-use allocation protects the Peruvian Amazon. Science 317 (5842), 1233-1236.

Oliveira, P.S., Marquis, R.J., 2002. The Cerrados of Brazil: Ecology and Natural History of a Neotropical Savanna. Columbia University Press, New York.

Olson, D.M., Dinerstein, E., 1998. The Global 200: A representation approach to conserving the Earth's most biologically valuable ecoregions. Conservation Biology 12, 502–515.

Olson, D.M., Dinerstein, E., Wikramanayake, E.D., 2001. Terrestrial ecoregions of the world: a new map of life on Earth. Bioscience 51 (11), 933-938.

Omernik, J.M., 1995. Ecoregions – a framework for environmental management, In Biological assessment and criteria-tools for water resource planning and decision making, (Eds. W.S. Davis, T.P. Simon), 49-62, Lewis Publishers, Boca Raton, Florida.

Ostrom, E., 1990. Governing the Commons: The Evolution of Institutions for Collective Action. Cambridge University Press, Cambridge.

Ostrom, E., Burger, J., Field, C.B., Norgaard, R.B., Policansky, D., 1999. Revisiting the Commons: Local Lessons, Global Challenges. Science 284, 278-282.

Oyama, M.A., Nobre, C.A., 2003. A new climate-vegetation equilibrium state for Tropical South America . Geophysical Research Letters 30 (23), 2199. doi:10.1029/2003GL018600

Page, S.E., Siegert, F., Rieley, J.O., Boehm, H.D.V., Jayak, A., Limink, S., 2002 The amount of carbon released from peat and forest fires in Indonesia during 1997. Nature 420, 61-65.

Parrotta, J.A., Turnbull, J.W. 1997. Catalyzing native forest regeneration on degraded tropical lands. Forest Ecology and Management 99 (1-2), 1-7.

Peres, C.A., Barlow, J., Haugaasen, T., 2003. Vertebrate responses to surface fires in a central Amazonian forest. Oryx 37, 97-109.

Perz, S., Brilhante, S., Brown, F., Caldas, M., Ikeda, S., Mendoza, E., Overdevest, C., Reis, V., Reyes, J.F., Rojas, D., Schmink, M., Souza, C., Walker, R., 2008. Road building, land use and climate change: prospects for environmental governance in the Amazon. Philosophical Transactions of the Royal Society of London B. 363 (1498),1889-1895. doi: 10.1098/rstb.2007.0017

Peskett, L., Luttrell, C., Iwata, M., 2007. Can standards for voluntary carbon offsets ensure development benefits? ODI Forestry Briefing 13, Overseas Development Institute, London. Accessible at http://www.odi.org.uk/resources/download/11.pdf (accessed on November 2, 2009)

Peskett, L., Huberman, D., Bowen-Jones, E., Edwards, G., Brown, J., 2008. Making REDD work for the poor. London, Overseas Development Institute. Accessible at http://www.odi.org.uk/ccef/resources/reports/s0179_redd-_final_report.pdf (accessed on November 2, 2009)

Pickett, S.T.A., Parker, V.T., Fiedler, P.L., 1992. The new paradigm in ecology: implications for conservation biology above the species level. In Conservation Biology: The Theory and Practice of Nature Conservation, Preservation, and Management (Eds. P.L. Fiedler, S. K. Jain), 37-56, Chapman and Hall, New York.

Pielke, R.A., Eastman, J., Chase, T. N., Knaff, J., Kittel, T.G. F., 1998. The 1973-1996 trends in depth-averaged tropospheric temperature, Journal of Geophysical Research 103, 16,927-16,933.

Plumptre, A.J., 2001. The effects of habitat change due to selective logging on the fauna of forests in Africa, in African Rain Forest Ecology and Conservation (Eds. W. Webber, L.J.T. White, A. Vedder, L. Naughton-Treves), 463-479, Yale University Press, New Haven, CT.

Poiani, K.A., Richter, B.D., Anderson, M.G., Richter, H.E., 2000. Biodiversity conservation at multiple scales: functional sites, landscapes, and networks. Bioscience 50 (2), 133-146.

Point Carbon, 2009. Carbon Market Monitor January 2009: a review of 2008. Point Carbon, Oslo. Accessible at http://www.pointcarbon.com/research/carbonmarketresearch/monitor/1.1034635 (accessed on November 2, 2009)

Pringle, C.M., Hamazaki, T., 1997. Effects of fishes on algal response to storms in a tropical stream. Ecology 78, 2432–2442.

Pusey, B.J., Arthington, A.H., 2003. Importance of the riparian zone to the conservation and management of freshwater fish: a review. Marine and Freshwater Research 54, 1–16.

Putz, F.E., Sirot, L.K., Pinard, M.A., 2001. Tropical forest management and wildlife: silvicultural effects on forest structure, fruit production, and locomotion of arboreal animals. In The Cutting Edge: Conserving Wildlife in Logged Tropical Forests (Eds. R.A. Fimbel, A. Grajal, J.G. Robinson), 11-34, Columbia University Press, New York.

Putz, F.E., Sist, P., Fredericksen, T.S., Dykstra, D. 2008. Reduced-impact logging: Challenges and opportunities. Forest Ecology and Management 256, 1427-1433.

Ray, D.G., Nepstad, D.C., Moutinho, P., 2005. Micrometeorological and canopy controls of fire susceptibility in mature and disturbed forests of an east-central Amazon landscape. Ecological Applications 15 (5), 1664-1678.

Rosenthal, E., 2009. In Brazil, Paying Farmers to Let the Trees Stand. New York Times, August 22.

Saatchi, S.S., Houghton, R.A., Dos Santos Alvala, R.C., Soares, J.V., Yu, Y., 2007. Distribution of aboveground live biomass in the Amazon basin. Global Change Biology 13 (4), 816-837.

Sahin, V., Hall, M.J., 1996. The effects of afforestation and deforestation on water yields. Journal of Hydrology 178, 293-309.

Sampaio, G., Nobre, C., Costa, M.H., Satyamurty, P., Soares-Filho, B.S., Cardoso, M., 2007. Regional climate change over eastern Amazonia caused by pasture and soybean cropland expansion. Geophysical Research Letters 34, L17709. doi:10.1029/2007GL030612

Sampson, G.P., 2000. Trade, Environment and the WTO: the Post-Seattle Agenda. Overseas Development Council, Washington, DC. 151 pp.

Sanches, R.A., 2002. Desmatamento na região dos formadores do Rio Xingu, Mato Grosso, Brasil. ISA, São Paulo.

Sanchez-Azofeifa, G.A., Daily, G.C., Pfaff, A.S.P., Busch, C., 2003. Integrity and isolation of Costa Rica's national parks and biological reserves: examining the dynamics of land-cover change. Biological Conservation 109, 123-135.

Sanchez-Azofeifa, G.A., Pfaff, A., Robalino, J.A., Boomhower, J.P., 2007. Costa Rica's Payment for Environmental Services Program: Intention, Implementation, and Impact. Conservation Biology 21 (5),1165-73.

Sanderson, E.W., Redford, K.H., Vedder, A., Coppolillo, P., Ward, S.E., 2002. A conceptual model for conservation planning based on landscape species requirements. Landscape and Urban Planning 58, 51-56.

Santilli, M., Moutinho, P., Schwartzman, S., Nepstad, D.C., Curran, L., 2005 Tropical deforestation and the Kyoto Protocol: an editorial essay. Climate Change, 71, 267–76.

Sawaya, K.E., Olmanson, L.G., Heinert, N.J., Brezonik, P.L., Bauer, M.E., 2003. Extending satellite remote sensing to local scales: land and water resource monitoring using high-resolution imagery. Remote Sensing of Environment 88 (1-2), 144-156.

SBI Reports, 2009. Carbon Emissions Trading Markets Worldwide. SBI Reports, Rockville, MD. 186 pp. Accessible at http://www.sbireports.com/Carbon-Emissions-Trading-1926753/ (accessed on November 2, 2009)

Scanlon, T.M., Caylor, K.K., Levin, S.A., Rodriguez-Iturbe, I., 2007. Positive feedbacks promote power-law clustering of Kalahari vegetation. Nature 449, 209-213.

Schafer, J.S., Eck, T.F., Holben, B.N., Artaxo, P., Yamasoe, M.A., Procopio, A.S., 2002. Observed reductions of total solar irradiance by biomass- burning aerosols in the Brazilian Amazon and Zambian Savanna. Geophysical Research Letters, 29.

Schafer, J.S., Holben, B.N., Eck, T.F., Yamasoe, M.A., Artaxo, P., 2002. Atmospheric effects on insolation in the Brazilian Amazon: Observed modification of solar radiation by clouds and smoke and derived single scattering albedo of fire aerosols. Journal of Geophysical Research-Atmospheres, 107.

Schwarzenbach, R.P., Escher, B.I., Fenner, K., Hofstetter, T.B., Johnson, C.A., von Gunten, U., Wehrli, B., 2006. The challenge of micropollutants in aquatic systems. Science, 313, 1072 – 1077.

Secretaria de Estado de Planejamento e Coordenação Geral de Mato Grosso (SEPLAN-MT), 2008. *Projeto Zoneamento Socioeconômico Ecológico do Estado de Mato Grosso*. Cuiába, Mato Grosso. Accessible at http://www.geo.seplan.mt.gov.br/zsee/ (accessed on November 29, 2009)

Soares-Filho, B.S., Cerqueira, G.C., Pennachin, C.L., 2002. DINAMICA—a stochastic cellular automata model designed to simulate the landscape dynamics in an Amazonian colonization frontier. Ecological Modelling 154, 217-235.

Soares-Filho, B., Alencar, A., Nepstad, D., Cerqueira, G., Diaz, M.C.V., Rivero, S., Solorzano, L., Voll, E., 2004. Simulation of deforestation and forest regrowth along a major Amazon highway: the case of the Santarém-Cuiabá highway. Global Change Biology 10, 745-764.

Soares-Filho, B.,Nepstad, D., Curran, L.. Cerqueira, G., Garcia, R., Ramos, C., Voll, E., McDonald, A., Lefebvre, P., Schlesinger, P., 2006. Modeling Amazon conservation. Nature 440, 520–523.

Soares-Filho, B.S., Rodrigues, H.O., Costa, W.L., 2009. Modeling Environmental Dynamics with Dinamica EGO. Accessible at: http://www.csr.ufmg.br/dinamica (accessed on November 28, 2009)

Sonzogni, W.C., Chesters, G., Coote, D.R., Jeffs, J.C., Robinson, J.B., 1980, Pollution from land runoff. Environmental Science and Technology 14, 148-153.

Southgate, D., Whitaker, M., 1992. Promoting Resource Degradation in Latin America: Tropical Deforestation, Soil Erosion, and Coastal Ecosystem Disturbance in Ecuador. Economic Development and Cultural Change 49, 787-807.

Sterner, T., 2003. Policy instruments for environmental and natural resource management. RFF Press, Washington, DC.

Stickler, C.M., Nepstad, D.C., Coe, M.T., Rodrigues, H.O., McGrath, D.G., Walker, W.S., Soares-Filho, B.S., Davidson, E.A. 2009. The potential ecological costs and co-benefits of REDD: a critical review and case study from the Amazon region. Global Change Biology 15, 2803-2824.

Stocking, M.A., 2003. Tropical soils and food security: the next 50 years. Science 302, 1356. DOI: 10.1126/science.1088579

Sweeney, B.W., Bott, T.L., Jackson, J.K., Kaplan, L.A., Newbold, J.D., Standley, L. J., Hession, W.C., Horwitz, R.J., 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. Proceedings of the National Academy of Sciences 101, 14132–14137.

Tarnocai, C., Canadell, J.G., Schuur, E.A.G., Kuhry, P., Mazhitova, G., Zimov, S., 2009. Soil organic carbon pools in the northern circumpolar permafrost region. Global Biogeochemical Cycles 23, GB2023; doi:10.1029/2008GB003327.

Terborgh, J., 1999. Requiem for Nature. Island Press, Washington, DC, 426 pp.

Thanapakpawin, P., Richey, J., Thomas, D., Rodda, S., Campbell, B., Logsdon, M., 2007. Effects of landuse change on the hydrologic regime of the Mae Chaem river basin, NW Thailand. Journal of Hydrology 334, 215-230.

Thiollay, J.M., 1992. Influence of selective logging on bird species diversity in a Guianan rain forest. Conservation Biology 6, 47-63.

Tietenberg, T., 1996. Private Enforcement of Environmental Regulations in Latin America and the Caribbean: An Effective Instrument for Environmental Management? Inter-American Development Bank, Washington, D.C., June. No. ENV-101. Accessible at http://www.iadb.org/sds/doc/env-101e.pdf (accessed on November 2, 2009)

Tollefson, J., 2009. Paying to save the rainforests. Nature 460, 936-937.

Torgersen, C.E., Faux, R.N., McIntosh, B.A., Poage, N.J., Norton, D.J., 2001. Airborne thermal remote sensing for water temperature assessment in rivers and streams. Remote Sensing of Environment 76, 386-398.

Tortajada, C., 2001. Institutions for Integrated River Basin Management in Latin America. Water Resources Development 17 (3), 289-301.

Turner, M.G., 1989. Landscape ecology: the effect of pattern and process. Annual Review of Ecology and Systematics 20, 171-197.

Van Kooten, G.C., Bulte, E.H., 2000. The Economics of Nature: Managing Biological Assets. Oxford: Blackwell Publishers.

Vannote, R.L., Minshall, J.V., Cummins, K.W., Seddell, J.R., Cushing, C.E., 1980. The river continuum concept. Canadian. Journal of Fisheries and Aquatic Science 37, 130-137.

Vayda, A.P., 2006 Causal explanation of Indonesia's forest fires: concepts, applications, and research priorities. Human Ecology 34 (5), 615-635.

Vera-Diaz, M.D.C., Kaufmann, R.K., Nepstad, D.C., Schlesinger, P., 2007. An interdisciplinary model of soybean yield in the Amazon Basin: the climatic, edaphic, and economic determinants. Ecological Economics 13, 134-141.

Vira, B., Adams, W.M., 2009. Ecosystem services and conservation strategy: beware the silver bullet. Conservation Letters 49, 6-7.

Visser, H., de Nijs, T., 2006. The map comparison kit. Environmental Modelling and Software 21, 346-358.

Vogelmann, J.E., Sohl, T.L., Campbell, P.V., Shaw, D.M., 1998. Regional land cover characterization using Landsat Thematic Mapper data and ancillary data sources. Environmental Monitoring and Assessment 51, 415-428.

Vollenweider, R.A., 1971. The scientific fundamentals of lake and stream eutrophication with particular reference to phosphorus and nitrogen as eutrophication factors. OECD, Paris.

Walker, W.S., Stickler, C.M., Kellndorfer, J.M., Kirsch, K., Nepstad, D.C. Cloud-free mapping of tropical forests with satellite radar. Nature Geoscience, in preparation.

Walker, W.S., Kellndorfer, J.M., Lapoint, E., Hoppus, M., Westfall, J., 2007. An empirical InSAR-optical fusion approach to mapping vegetation canopy height. Remote Sensing of Environment 109, 482-499.

Watchman, P., Delfino, A., Addison, J., 2007. EP 2: The revised Equator Principles: Why hard-nosed bankers are embracing soft law principles. Law and Financial Markets Review 1 (2), 79-90.

Weiss, T.G., 2000. Governance, Good Governance and Global Governance: Conceptual and Actual Challenges. Third World Quarterly 21 (5), 795-814.

Werth, D., Avissar, R., 2002. The local and global effects of Amazon deforestation. The Journal of Geophysical Research, 107 (D20), 8087.

Werth, D., Avissar, R., 2005. The local and global effects of Southeast Asian deforestation. Geophysical Research Letters, 2005GL022970

Werth, D., Avissar, R., 2005. The local and global effects of African deforestation. Geophysical Research Letters, 12704, 2005GL022969

Wilcox, B.A., 1984. In situ conservation of genetic resources: Determinants of minimum area requirements. In National Parks, Conservation and Development, Proceedings of the World Congress on National Parks (Eds. JA McNeely, KR Miller), 18-30, Smithsonian Institution Press, Washington, D.C.

Williams, M.R., Melack, J.M., 1997. Solute export from forested and partially deforested catchments in the central Amazon. Biogeochemistry 38, 67-102.

Williams, E., Rosenfeld, D., Madden, N., Gerlach, J., Gears, N., Atkinson, L., Dunnemann, N., Frostrom, G., Antonio, M., Biazon, B., Camargo, R., Franca, H., Gomes, A., Lima, M., Machado, R., Manhaes, S., Nachtigall, L., Piva, H., Quintiliano, W., Machado, L., Artaxo, P., Roberts, G., Renno, N., Blakeslee, R., Bailey, J., Boccippio, D., Betts, A., Wolff, D., Roy, B., Halverson, J., Rickenbach, T., Fuentes, J., Avelino, E., 2002. Contrasting convective regimes over the Amazon: Implications for cloud electrification. Journal of Geophysical Research-Atmospheres, 107.

Winterbottom, R. 1990. Taking stock: the Tropical Forestry Action Plan after five years. World Resources Institute (WRI), Washington, DC, 59 pp. http://db.jhuccp.org/ics-wpd/exec/icswppro.dll?BU=http://db.jhuccp.org/ics-wpd/exec/icswppro.dll&QF0=DocNo&QI0=075491&TN=Popline&AC=QBE_QUERY&MR=30%25DL=1&&RL=1&&RF=LongRecordDisplay&DF=LongRecordDisplay (accessed on November 2, 2009)

Woodroffe, R., Ginsberg, J.R., 1998. Edge effects and the extincition of populations inside protected areas. Science 280, 2126-2128.

Wunder, S. 2006. Are direct payments for environmental services spelling doom for sustainable forest management in the tropics? Ecology and Society 11 (2), 23. URL: http://www.ecologyandsociety.org/vol11/iss2/art23/

Wunder, S., 2007. The efficiency of payments for environmental services in tropical conservation. Conservation Biology 21 (1), 48-58.

Y ikatu Xingu. 2009. Campaign to save the Xingu River. Accessible at http://www.yikatuxingu.org.br/home (accessed on November 2, 2009)

Zakia, M.J., 2005. A silvicultura em áreas declivosas e o topo de morro. Ministério do Meio Ambiente, Brasília, DF.

Zarin, D.J., Ducey, M.J., Tucker, J.M., Salas, W.A., 2001. Potential biomass accumulation in Amazonian regrowth forests. Ecosystems 4(7), 658-668.

Zarin, D. J., Davidson, E.A., Brondizio, E., Vieira, I.C.G., Sá, T., Feldpausch, T., Schuur, E.A.G., Mesquita, R., Moran, E., Delamonica, P., Ducey, M.J., Hurtt, G.C., Salimon, C., Denich, M., 2005. Legacy of fire slows carbon accumulation in Amazonian forest regrowth. Frontiers in Ecology and the Environment. 3, 365-369.

BIOGRAPHICAL SKETCH

Claudia Stickler received her B.S. degree in biology and international political economy from the University of Puget Sound, Tacoma, Washington, in May 1996. She spent three years in Cameroon, Central Africa, first as an agroforestry extension agent with the Peace Corps and then as a research assistant on a San Francisco State University project focusing on tropical forest regeneration and plant-animal interactions. In 2000, she worked in Suriname, South America, as a research assistant on a long-term University of Florida study of primate behavior and ecology. She received her M.S. degree from the College of Natural Resources and Environment, University of Florida, in 2004. Her M.S. research focused on the effect of logging on habitat selection by primates in Kibale National Park in Uganda. She is affiliated with the Department of Geography, the Tropical Conservation and Development Program, and the Land Use and Environmental Change Institute at the University of Florida. Since 2004, she has been working as a Visiting Scholar with the Amazon Environmental Research Institute (*Instituto de Pesquisa Ambiental da Amazônia*) in Brazil and as a Graduate Fellow with the Woods Hole Research Center in Falmouth, Massachusetts.