Seeing the forest for the streams: A multiscale study of land-use change and stream ecosystems in the Amazon's agricultural frontier

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Abstract

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Global demand for agricultural products is an increasingly important driver of deforestation in the Amazon Basin. This dissertation examines the consequences of agricultural expansion for stream ecosystems in the southern Amazon’s agricultural frontier. At regional scales, the removal of watershed forest cover is known to change the energy balance and influence hydro-climatic cycling by altering stream flow, regional rainfall patterns, and land surface temperatures. At the landscape scale, these physical changes may be further exacerbated by land management practices that lead to the degradation of riparian forest buffers; decreases in connectivity; changes in the amount of light, nutrient, and sediment inputs; and decreases in water quality. Together, land use and management influence the quality and distribution of aquatic habitats within stream networks, potentially decreasing stream biotic integrity and resilience to further disturbances.

Brazil’s Mato Grosso state is one of the most actively expanding agricultural frontiers in the world and represents an ideal case study for examining the linkages among tropical deforestation, agricultural expansion, and the conservation of freshwater ecosystems. Mato Grosso accounted for 40% of deforestation in the Brazilian Amazon during the early 2000s – primarily due to the expansion of soybeans and cattle ranching. Deforestation rates have since dropped throughout the Amazon, but there is a lack of spatially explicit information about the land use transitions accompanying this decline. To address this gap, I combined government data on deforestation and production with the MODIS satellite time series to quantify the spatial-

temporal dynamics of land use change in the region. Although agricultural expansion during this period slowed with declining commodity prices, the decline in deforestation is partly explained by a shift from soybean expansion into forests (26% of expansion from 2001-2005) to expansion into already cleared pasture lands (9% of expansion form 2006-2010). Beyond documenting these trends, the resulting dataset is a critical first step in evaluating the influence of land use and land use history on freshwater ecosystems at multiple scales.

In the headwaters region of the Xingu River Basin, the proportion of small watersheds (microbasins) dominated by agriculture (>60% of area) increased from 20 to 40% from 2001 to 2010. At the same time, the stream network became increasingly fragmented by the removal of riparian forest buffers and installation of farm impoundments. I used high resolution satellite data (ASTER) to produce the first landscape-level documentation of farm impoundments in the region, mapping approximately 10,000 impoundments (one per 7.6 km of stream length) in 2007. At the catchment scale, I collected field data in 12 headwater streams to examine the effect of land management on instream water quality. Watershed forest cover (from MODIS), the density of impoundments (from ASTER), and the percent forest in upstream riparian buffers (from Landsat) were all associated with substantial increases in stream temperature. These increases in fragmentation and water temperature may have large cumulative effects on the stream network and reduce the ability of downstream protected areas to conserve freshwater resources. At the scale of the Amazon Basin, my analysis indicates that 30% of indigenous lands and protected areas are highly vulnerable to future reductions in hydrologic connectivity, simply because of their location within their watersheds. These impacts could be substantially mitigated through enforcement of existing legislation to protect riparian buffers and new regulations to limit the number of impoundments in emerging agricultural landscapes.

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List of Abbreviations

AIC, Akaike Information Criterion
CONAB, National Food Supply Company of Brazil
EVI, Enhanced Vegetation Index
FUNAI, Brazilian National Indian Foundation
FGV, Getúlio Vargas Foundation
HCI, Hydrologic Connectivity Index
IBGE, Brazilian Institute of Geography and Statistics
ILPA, Indigenous Lands and Protected Areas
INPE, Brazilian National Institute for Space Research
LPDAAC, Land Processes Distributed Active Archive Center
MODIS, Moderate Resolution Imaging Spectroradiometer
MT, Mato Grosso
NASA, National Aeronautics and Space Administration
NDVI, Normalized Difference Vegetation Index
PRODES, Program for the Estimation of Deforestation in the Brazilian Amazon
REDD+, Reduced Emissions from Deforestation and Degradation in Developing Countries
UNFCCC, United Nations Framework Convention on Climate Change
ZOI, Zone of Influence

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Dedication

To my parents, Nelson and Lidia, for their love and support in all that I do, and to Marcelo, my little brother and best friend.

Chapter 1

Introduction

As I write this dissertation, the world population has just exceeded 7 billion people and is projected to climb to over 9 billion before the end of the century (UN, 2010). Growing enough food to feed all of these new mouths, while conserving tropical forests and the ecological services they provide, will be one of the great challenges of this generation. As demand for food, fiber, and fuel grows to unprecedented levels, global markets have become increasingly connected to tropical forest regions, which house the largest remaining supply of new land for agriculture. Converting tropical forests for agricultural production is not without consequence. These ecosystems not only support the highest levels of terrestrial and aquatic biodiversity on earth, but also provide important services in the form of carbon storage and sequestration (Ometto et al., 2011); regulation of stream flow and regional precipitation (Werth & Avissar, 2002); and the provision of clean water and food for local populations. This dissertation examines some of these tradeoffs in the Amazon's agricultural frontier, focusing on the linkages among deforestation, land management, and the conservation of freshwater ecosystems.

Agricultural expansion and intensification can degrade tropical stream ecosystems through a variety of mechanisms. Large-scale conversion of forests to croplands and pasture grasses alters the regional energy balance by reducing evapotranspiration and increasing surface albedo (Loarie et al., 2011a). This anthropogenic forcing results in a net increase in land surface temperature (Loarie et al., 2011b, Sampaio et al., 2007) and fundamental changes to the hydrological cycle (Coe *et al.*, 2009, Werth & Avissar, 2004). In the Amazon Basin, deforestation has been shown to increase stream discharge (Coe et al., 2009, Coe et al., 2011, Hayhoe et al., 2011) and decrease rainfall (Werth & Avissar, 2002) at local and regional scales, although the magnitude and direction of response depend on the level of deforestation and scale of analysis (Da Silva et al., 2008). In addition to regulating water quantity, watershed forest cover plays an important role in maintaining water quality by regulating the amount of light, nutrients, and sediments reaching streams from upland areas. Reductions in water quality due to watershed forest loss may be exacerbated by the activities that accompany subsequent agricultural land uses, including the installation of dams and farm impoundments, degradation of riparian areas, water diversion or withdrawal, and addition of chemical fertilizers and pesticides (Pringle, 2001).

While land cover change is known to alter stream hydrology, geomorphology, and the ecological integrity of streams in temperate regions (Snyder et al., 2005, Townsend et al., 1997), comparatively little research has been done on similar questions in tropical watersheds, where understanding the controls on fish abundance and diversity may be important for local livelihoods (Wright & Flecker, 2004). Tropical streams are a vital source of protein and clean water for local populations, not to mention the primary transportation network across much of the Amazon region. Even in areas that are otherwise sparsely populated, human populations live disproportionately near waterways (Sala et al., 2000). As the number of tropical protected areas has increased, so have the number of people living in and around these areas whose livelihoods depend on the freshwater resources they provide. By altering the physical parameters that define stream habitats, agricultural land uses may negatively impact the quality and distribution of fish habitat, and compromise the ability of protected areas to conserve freshwater resources.

There is a growing recognition of the urgent need for landscape management to mitigate the potential ecological and social impacts of agricultural expansion and intensification. Today, over 45% of the Brazilian Amazon is already under some form of protection (Soares et al., 2010), including strict protected areas, sustainable use areas, and indigenous lands. As demand for agricultural land grows, additional "set asides" for conservation will become increasingly difficult[1], and the success of existing areas will depend on sound management of the unprotected landscapes that surround them. This is particularly true in the case of freshwater ecosystems, given that the hydrological cycle transcends political boundaries and the flow of water in stream networks may directly link protected areas and their surrounding landscapes. Brazilian legislation provides a framework for this type of landscape management by designating watersheds as the basic unit of land use planning and creating a Forest Code that requires the conservation of forests and riparian buffers on private lands. While the regulation of public interests on private lands is a laudable step towards integrated land use management, creating the capacity and political will necessary for its implementation on the ground has proven extremely challenging.

The land-water interface

Land management plays an important role in mediating the influence of crop and cattle production on stream ecosystems. Mitigation strategies include: no tillage to prevent soil erosion; conservation of riparian forests to buffer streams against agricultural pollutants and sediments; fencing cattle out of streams (and providing artificial water sources) to prevent soil compaction and degradation of riparian areas; and management of watershed forest cover, particularly on

[1] Recent discussions in the Brazilian Senate have even considered decreasing the size of existing protected areas.

slopes, to control erosion and the cumulative impacts of agriculture in the landscape. The installation of infrastructure, such as roads, dams, and farm impoundments may also have important consequences for stream connectivity at the landscape scale. Following is a discussion of several management strategies relevant to this study.

Riparian buffers

The role of riparian vegetation in mitigating the negative impacts of agriculture on stream health is well documented (Barker *et al.*, 2006). Maintaining permanently vegetated corridors between pollutant sources and water bodies can effectively buffer against the degrading effects of sedimentation and non-point source pollution (Naiman & Decamps, 1997); regulate the amount of nitrogen and other nutrients reaching streams from upland areas (Karr & Schlosser, 1978); and prevent the erosion of stream banks (Narumalani *et al.*, 1997). Establishing riparian buffers in degraded areas has been shown to improve stream bank cover; decrease sedimentation, manure, and nutrients; and improve water clarity – physical attributes that are directly correlated to improvements in the biological integrity of fish assemblages (Teels *et al.*, 2006).

Watershed forest cover

Studies in deforested catchments with intensive agricultural systems suggest that land use alterations at the catchment scale can overwhelm the capacity of riparian buffers to support healthy stream habitats and associated biotic communities (Roth *et al.*, 1996). Both field-based studies and modeling indicate that the proportion of forest cover in a watershed is often related to the amount of pollutants observed downstream from agricultural areas (Basnyat *et al.*, 2000, Gergel *et al.*, 2002, Uriarte *et al.*, 2011). Similarly, the health of fish communities is negatively correlated with the amount of upstream agriculture and positively correlated with the amount of upstream forest cover. This relationship may exhibit a non-linear threshold response, whereby declines in fish fauna occur abruptly after a large proportion (e.g., >50%) of the catchment is converted to agricultural land uses (Wang *et al.*, 1997).

Impoundments

In the Xingu Basin, the installation of small farm impoundments is widespread, but their extent and distribution in the landscape has only recently been quantified (Chapter 3). These impoundments act as physical barriers, altering the flow of water, sediments, and organisms within headwater streams. Although the vast majority of the literature on dams focuses on large hydroelectric dams, several studies indicate that small dams can have a large cumulative impact on stream ecosystems. Small dams alter physical habitat by increasing water temperature (Cumming, 2004); changing current velocity, water volume, and depth above and below impoundments (Alexandre & Almeida, 2010, Lehner *et al.*, 2011); and trapping fine sediments as a result of the slackwater created behind reservoirs (Walter & Merritts, 2008). When coupled with agricultural land uses, which often increase the supply of sediments and pollutants to streams, small impoundments have the potential to fundamentally alter the geomorphology and quality of habitats within stream networks. Hence, they may facilitate the establishment of invasive species (Johnson *et al.*, 2008) and have a strong negative impact on stream biota, particularly fish (Alexandre & Almeida, 2010, Cumming, 2004, Wang *et al.*, 2011) and macroinvertebrate (Tiemann *et al.*, 2005) communities.

Connectivity

The configuration of land uses in a watershed and the management practices associated with each land cover type can have a profound effect on the hydrological connectivity[2] of a stream network. Even in areas where stream reaches remain physically connected, fluvial species may experience a functional decrease in connectivity. For example, riparian forests exert strong controls on the microclimate of streams, affecting the amount and type of solar radiation reaching streams, as well as inputs of organic matter, sediments, and other pollutants from the surrounding landscape (Naiman & Decamps, 1997, Naiman & Latterell, 2005). In agricultural landscapes, riparian forests provide not only lateral connectivity at the land-water interface, but also longitudinal connectivity for aquatic species. Spatial characteristics, such as length, width, and continuity of riparian buffers can have a strong influence on their effectiveness in conserving stream habitat (Gergel *et al.*, 2002), with well documented implications for fish abundance and diversity (Jones *et al.*, 2006, Lorion & Kennedy, 2009, Wright & Flecker, 2004). Similarly, the impoundment of a section of stream creates a lentic (still water) environment in place of a lotic (running water) environment. Even though small dams are often traversable barriers to dispersal and migration (March *et al.*, 2003), heavily impounded stream reaches may be functionally fragmented if they require fluvial species to repeatedly move through suboptimal habitat conditions (Schlosser *et al.*, 2000).

Effective management of stream ecosystems requires systematic planning and a monitoring system that can link spatial patterns and ecological processes. The fields of spatial and landscape ecology have greatly improved our understanding of how spatial patterns across landscapes can influence ecosystems locally (Leitão et al., 2006, Turner et al., 2001, Wiens,

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[2] Here I adopt the definition of hydrologic connectivity first introduced by Pringle (2001): "the water-mediated transport of matter, energy and organisms within and between elements of the hydrological cycle".

2002), but these principles have only recently been applied to stream ecosystems (Allan, 2004, Gergel *et al.*, 2002, Grant *et al.*, 2007), particularly in the tropics (Uriarte *et al.*, 2011). This dissertation combines remote sensing and field-based inventories to further our understanding of the linkages between land use and stream ecosystem health in the Amazon's agricultural frontier. Specifically, I employ GIS and remote sensing tools to examine land use transitions through time (Chapter 2), as well as the distribution of land use, riparian forests, and impoundments in space (Chapter 3). Armed with this landscape-scale perspective, I am able to assess the consequences for stream ecosystems at multiple scales and identify opportunities for improved management (Chapter 3). At the Amazon scale, I assess the vulnerability of Amazon indigenous lands and protected areas to land use changes within their watershed (Chapter 4), based on current and predicted deforestation in the surrounding landscape. Finally, I summarize the results of the dissertation at catchment, landscape, and regional scales and introduce some of the relevant policies and institutions at each scale (Chapter 5).

Research objectives

My research aims to understand how land use and land management influence tropical streams, while producing information that is of practical relevance for the management of freshwater resources in agricultural landscapes. This dissertation lays the groundwork for addressing this goal by accomplishing the following objectives:

(1) Examines the dynamics of deforestation and subsequent land use transitions in an ecologically and economically important frontier landscape.

(2) Determines how land use history influences the spatial distribution of riparian forests and impoundments.

(3) Quantifies the consequences of land use and land use history for stream temperature and connectivity.

(4) Assesses the potential implications of current and modeled future land uses for the management of freshwater resources within Amazon indigenous lands and protected areas.

The remainder of this introduction provides an overview of the study system, describes the rationale and significance of the study, and outlines how the dissertation is structured to address these four objectives.

Study system

This dissertation focuses on the headwaters of the Xingu River Basin in the Brazilian state of Mato Grosso (MT). Located in the southern Amazon's agricultural frontier, the upper Xingu occurs along the ecotone between cerrado woodlands and dense tropical forests. The Xingu Indigenous Park (PIX) lies at the center of this region, protecting 2.6 million hectares of forest and a long stretch of the Xingu River. The PIX was created for the protection of several indigenous groups living within its borders, as well as the forests and freshwater resources they depend on. Despite its large size, it is susceptible to influences from outside land uses because nearly all of the headwater streams lie outside of its boundaries. The establishment of intensive agriculture in these headwater areas, which led to illegal deforestation of riparian forests, has reportedly increased turbidity and altered water quality to the point that the residents of the reserve have recounted noticeable impacts on fisheries (ISA, 2003).

Socio-political context

The Xingu Basin lies just to the east of highway BR-163, which is in the process of being paved from the southern border of Pará north to the port city of Santarém. Government plans to pave BR-163 were the focus of vigorous public debate in the early 2000s. Proponents of the road argued that this infrastructure would bring jobs and opportunities for development in an otherwise depressed region. Indeed, the highway will greatly reduce the cost of getting agricultural goods produced in Mato Grosso and Pará to markets outside the Amazon via the international port of Santarém (Fearnside, 2006a, Soares *et al.*, 2006). Others have argued that paving the road without adequate land use planning and strong governance would result in unprecedented deforestation (Soares *et al.*, 2004) and create a conduit for unregulated occupation of the central Amazon.

The promise of a new road and development along the highway brought massive immigration into the region. In Mato Grosso, this immigration took the form of planned cities, funded and occupied by *gauchos* from the south of Brazil, who were seeking new frontiers for agricultural expansion. In Pará the occupation was characterized by land speculation, illegal land occupation, land title conflicts, and violence[3] (Fearnside, 2007). Concern over the disordered occupation of the region brought together a diverse set of governmental and non-governmental stakeholders to develop an economic and ecological zoning plan that aimed to mitigate the potential deforestation impacts of the road. For all of its problems, the process eventually led to the creation of a 7 million hectare mosaic of new protected areas to the east of the highway in Pará, which have had a measurable effect in containing deforestation in the region (Soares *et al.*, 2010).

More recently, the socio-environmental debate in Brazil has shifted to focus on two particularly controversial policy initiatives, both of which have important implications for the Xingu Basin. First is a proposal by Senator Aldo Rebelo (PCdoB - Communist Party of Brazil) to modify the Brazilian Forest Code, which has recently been the subject of hot debate in the

[3] The murder of Sister Dorothy Stang on February 12, 2005 was a particularly high-profile example of the type of violence that plagued the region in the face of these land battles.

Brazilian Congress and in the national media. The proposal would reduce the amount of forest conservation required on private lands, reduce the riparian buffer width requirements, and provide amnesty for landowners who were not in compliance with the code prior to a certain date (Chapter 5, IPAM, 2011). Second is recent approval of the Belo Monte dam, which is slated for construction on the Xingu River near the city of Alta Mira in Pará. Belo Monte will be the third largest hydroelectric dam in the world and will cost over US$16 billion to build. The mega-project will affect an area of 1,500 km^2 and flood an area of 650 km^2, displacing between 20,000 and 40,000 people, including several indigenous groups, in the process (International Rivers, 2010). Approval of the Belo Monte resulted in an international outcry opposing it, including high profile campaigns by national and international celebrities. Opponents of the dam argue that, without additional dams upstream, Belo Monte will only operate at a third of its capacity and that the social and environmental costs of the project far outweigh its energy benefits (Fearnside, 2006b, Hurwitz *et al.*, 2011).

Land cover and land use change

The globalization of the Amazon soy and beef industries – coupled with gains in productivity from locally adapted plant varieties and the promise of decreased transportation costs – was a strong driver of deforestation in the upper Xingu Basin over the last decade (Bowman *et al.*, 2012, Nepstad *et al.*, 2006). Agribusiness now accounts for roughly a quarter of Brazil's gross domestic product (FAS, 2009) and the state of Mato Grosso is the country's leading producer of soy and beef. Deforestation in Mato Grosso totaled some 63,000 km^2 from 2000-2010, accounting for 35% of the deforestation in the Brazilian Amazon during the same period (INPE, 2011). Expansion of extensive cattle ranching remains the primary driver of

Amazon deforestation, although expansion of intensive, mechanized soy production became an important new driver of deforestation in Mato Grosso in the early 2000s. At its peak in 2003, soybean expansion was responsible for between 18 and 23% of annual deforestation in the state (Macedo *et al.*, 2012, Morton *et al.*, 2006). The latter half of the 2000s saw a marked decrease in deforestation (Nepstad *et al.*, 2009) and a shift from soy expansion into forested lands to expansion into already cleared (pasture) lands (Chapter 2).

This landscape-scale conversion of native vegetation to mechanized soybeans and pasture grasses may have immediate impacts on aquatic ecosystems by changing hydrology, sediment delivery rates, and a number of other physical variables. However, it is the long-term management practices associated with these new land uses that will ultimately determine the overall health of freshwater systems in the region. Although land use practices vary across the landscape, there are several emergent trends. The summary below is based on personal observations during the course of fieldwork from 2007-2010:

(1) Land is generally cleared using a combination of fire and heavy machinery to remove woody vegetation and level the land. In the case of mechanized soy production, fields are usually completely cleared of biomass in the first year and a transitional crop (usually rice) may be planted in the year of conversion. Large-scale cattle ranching operations may remove woody biomass (i.e., stumps, roots) using fire over the course of several years, since livestock can graze in the interim period.

(2) Because soils are nutrient poor, chemical fertilizers are usually applied prior to crop cultivation. Roughly 50 kg/ha of phosphorus and 80 kg/ha of potassium are applied to

fields under soy cultivation[4] each year. Significant amounts of lime are also applied every few years to reduce the acidity of the soil. The majority of cattle ranches do not invest in such soil improvements because the costs are prohibitive. As a result cattle pastures in the region are generally extensive and of low productivity.

(3) The majority of agricultural land is held in large farms and ranches, with individual landholders often controlling several thousand hectares. In cropland areas, pesticides, herbicides, and chemical desiccants are sprayed using a combination of crop dusting airplanes and specialized tractors. This generally requires several applications during the course of a growing season. Such chemical inputs are generally absent in pasture areas.

(4) The system of crop cultivation is capital intensive, highly mechanized, and primarily rain-fed. The result is a landscape characterized by large tracts of monoculture croplands in the rainy season and great expanses of bare soil in the dry season, with the concomitant problems of wind-driven soil erosion. Although livestock operations maintain some pasture grass cover throughout the year, they are highly susceptible to escaped fires during the long dry season.

(5) In addition to the primary crop (usually soybeans), producers are increasingly intensifying production by incorporating a second crop towards the end of the dry season (Galford *et al.*, 2008), usually corn, sorghum, millet, or cotton.

(6) Cattle production in the Amazon is generally extensive and of low productivity, with an average stocking density of 0.7 heads of cattle per hectare (Arima *et al.*, 2005). The Brazilian Ministry of Agriculture's Low Carbon Agriculture (ABC) program has

[4] In the case of soy cultivation, phosphorous is the limiting nutrient, as soy is a nitrogen-fixing legume.

identified intensification of the cattle sector as a major priority because of its potential to reduce pressure on forests and prevent deforestation-related greenhouse gas emissions.

(7) Soil conservation is increasingly a priority on well-managed farms and ranches. For example, the practice of no-till agriculture is gaining widespread adoption in the region, which has the potential to improve soil carbon storage and conservation (Galford *et al.*, 2011). Likewise, planting secondary crops and cover crops in the dry season has a positive impact in controlling soil erosion, which can lead to increased sedimentation in freshwater ecosystems.

(8) In some areas riparian forest buffers have been completely removed or severely degraded, particularly where cattle are allowed direct access to streams. Although all riparian buffers are legally protected under the Brazilian Forest Code, the degree of compliance with this law is highly variable (Stickler, 2009).

(9) Farm ponds or impoundments are commonly used in the upper Xingu Basin to provide water for cattle, generate hydroelectric power at the farm-scale, or as a result of new roads (and culverts) in the landscape. The installation of impoundments on private farms and ranches is unregulated and likely reduces connectivity in freshwater systems.

This dissertation employs a multiscale approach to examine the impacts of land use change (1), riparian buffer removal (8) and impoundment installation (9) on aquatic systems, as well as the influence of land management on water quality (4-7). The primary focus of the research is on changes in the quality and connectivity of aquatic habitats arising from the land management practices outlined above. The impact of other non-point source pollutants, including chemical fertilizers and pesticides (2-3), are beyond the scope of this dissertation.

Research implications

Investigating the causes of land use change in the Amazon's agricultural frontier and its consequences for freshwater ecosystems can provide important insights for improved land management in this and other emerging agricultural frontiers. Scale emerges as a prominent theme in this dissertation, and one that has particular relevance to the management of freshwater resources. Freshwater systems have a built-in asymmetry among potential users, such that those upstream inherently have more power to degrade or conserve a resource than those downstream (Lebel *et al.*, 2005), creating the potential for transboundary management issues (e.g., pollution, overfishing). Although human-environment systems are generally structured across temporal, spatial, and jurisdictional (or institutional) scales, research and management efforts often overlook this complexity, leading to dramatic failures in resource management (e.g., fisheries collapses; Cash *et al.*, 2006, Ostrom, 1999).

Cash et al. (2006) highlight three common challenges associated with management of resources in complex systems: 1) failure to recognize important scale and level[5] interactions; 2) mismatches between levels and scales in human-environment systems, and 3) failure to distinguish disparities in the way that different actors perceive and value resources at a given scale. The Xingu landscape provides a telling example of how these scale mismatches play out in the real world. Powerful agribusiness interests have the power to induce rapid, large-scale change in the landscape, but their collective actions and land management practices have downstream consequences for the freshwater resources used by indigenous groups, as well as global consequences in terms of greenhouse gas (GHG) emissions (Brondizio *et al.*, 2009). In

[5] Scale is used here to mean the "spatial, temporal, quantitative, or analytical dimensions used to measure and study a phenomenon." Level refers to "the units of analysis that are located at different positions on a scale (Gibson *et al.*, 2000).

this context, sustainable management of resources will likely require developing new governance structures that enable less powerful stakeholders (e.g., indigenous groups) to shift across scales and influence environmental outcomes (Lebel *et al.*, 2005, Ostrom, 2010). The development of powerful alliances among indigenous groups and coordination with NGOs in the region provides one example of such a shift (Brondizio *et al.*, 2009). Another example is the mobilization of local and regional institutions to improve monitoring and enforcement of national environmental regulations on private properties. One particularly promising initiative is the development of a voluntary land registry that educates rural landowners about best practices and is working towards creating financial incentives (and accountability) for implementing them on individual properties. By examining land-water interaction across several scales, this dissertation aims to provide science-based information that is relevant to these management initiatives and takes an important step towards bridging the scale mismatches described above.

Dissertation structure

Substantial progress has been made in advancing the theory of freshwater conservation planning. However, our practical knowledge of how to manage the negative impacts of land conversion and intensification in tropical systems is still quite rudimentary. This is due in part to a lack of multiscale research into the phenomena that drive the connections between land cover change and instream ecosystem processes. Research ranges from large-scale modeling of hydrology and land use change to small watershed studies. What is generally lacking is research that looks across scales to better understand how cumulative alterations to hydrologic connectivity in the landscape influence ecological patterns locally. This dissertation begins to address this gap by integrating satellite-based analyses with field-based surveys to inform management at the scale of individual catchments, the Xingu Basin, and the Amazon Basin. Chapters 2 - 4 of this document serve as stand-alone research articles that contribute to this goal. Following is a brief overview of the questions addressed in each.

Chapter 2

In Chapter 2, I examine the tradeoffs between forest conservation and food production in the forested region of Mato Grosso. Specifically, I develop annual land use classifications, using the MODIS[6] time series to examine the dynamics of deforestation from 2001–2010 and the land use transitions associated with declining deforestation after 2005. The chapter addresses the following questions:

- What land-use transitions – cropland expansion into forest, expansion into already cleared lands, or changes in yields – occurred during the 2000s? How do trends vary between the first and second halves of the decade?
- Was declining deforestation from 2006-2010 associated with fluctuations in commodity markets, policy interventions, or both?

By quantifying the spatial and temporal dynamics of soybean and cattle expansion in the region and the degree to which they drive deforestation through time, this chapter lays the groundwork for the following chapters.

Chapter 3

In Chapter 3, I examine the influence of watershed forest cover, riparian buffers, and impoundments on headwater stream temperature. Building on the data outputs from Chapter 2, I

[6] The Moderate Resolution Imaging Spectroradiometer (MODIS) is an imaging instrument flying aboard the Terra satellite, launched in December 1999 by the National Aeronautics and Space Administration (NASA). It provides daily images of the Earth's surface at 250-m resolution.

use higher resolution satellite products (ASTER[7] and Landsat 5 [8]) to map the distribution of impoundments and riparian forests in the landscape and combine these with 16 months' worth of field measurements in 12 headwater catchments. The chapter addresses the following questions:

- What is the spatial distribution of soybeans and cattle ranching within the Xingu Basin and how has this changed over the last decade? How is land use history associated with the distribution of farm impoundments in the landscape?

- What is the relationship between land management (i.e., forest cover, riparian buffers, and impoundments) and stream temperature at the catchment scale?

- How might current management strategies be modified to mitigate the impacts of agricultural expansion on headwater streams?

By integrating field data with landscape information derived from multiple satellite sensors, I am able to assess the impact agricultural management on stream temperature at catchment and landscape scales and discuss potential mitigation strategies.

Chapter 4

In Chapter 4, I use the case of the Xingu Indigenous Park to illustrate how stream fragmentation in agricultural landscapes can threaten freshwater resources within protected areas. I then examine the vulnerability of the Amazon network of indigenous lands and protected areas (ILPAs) to similar losses in fragmentation in the future. This chapter addresses the following questions:

- To what extent has land use change in the headwaters of the Xingu Basin already altered hydrologic connectivity in the zone of influence of the PIX?

- How many Amazon Basin ILPAs are vulnerable to similar hydrologic fragmentation in the future?

[7] The Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) is a high resolution (15-m) sensor, flying aboard NASA's Terra satellite.
[8] NASA's Landsat 5 satellite was launched in 1984 and provides images of the Earth at 30-m resolution, approximately every 16 days.

- Given limited resources, how can we prioritize management interventions based on the likely timing of development?

By combining the data outputs from Chapters 2 and 3 with existing datasets on regional hydrology and projected future deforestation (Soares *et al.*, 2006), I am able to assess threats to individual protected areas based on their location within the hydrological landscape and the likely timing of development.

Chapter 5

In Chapter 5, I summarize the main results of the dissertation at each scale of study and place them in the context of the major institutions and policies operating at each scale. Finally, I highlight some of the cross-scale challenges inherent in managing agricultural landscapes for the long-term sustainability of freshwater systems.

Freshwater ecosystems are at the forefront of the global biodiversity crisis, with more declining and extinct species than terrestrial or marine environments (Abell *et al.*, 2008) (Johnson *et al.*, 2008). Understanding the response of these systems to human-induced forcing requires a holistic approach that couples site-specific information with a broad understanding of land use patterns in the watershed. By combining landscape level indicators derived from remote sensing with field-based measures of disturbance, this dissertation helps to elucidate the relationship between land use and aquatic ecosystems and provide the scientific basis for improved management of these systems.

References

Abell R., Thieme M. L., Revenga C., Bryer M., Kottelat M., Bogutskaya N., Coad B., Mandrak N. *et al.* (2008) Freshwater ecoregions of the world: A new map of biogeographic units for freshwater biodiversity conservation. *Bioscience,* 58, 403-414.

Alexandre C. M., Almeida P. R. (2010) The impact of small physical obstacles on the structure of freshwater fish assemblages. *River Research and Applications*, 26, 977-994.

Allan J. D. (2004) Landscapes and riverscapes: The influence of land use on stream ecosystems. *Annual Review of Ecology Evolution and Systematics*, 35, 257-284.

Arima E., Barreto P., Brito M. (2005) *Pecuária na Amazônia: Tendências e implicações para a conservação ambiental*, Belém, Pará, Instituto do Homem e Meio Ambiente da Amazônia.

Barker L. S., Felton G. K., Russek-Cohen E. (2006) Use of Maryland Biological Stream Survey data to determine effects of agricultural riparian buffers on measures of biological stream health. *Environmental Monitoring and Assessment*, 117, 1-19.

Basnyat P., Teeter L. D., Lockaby B. G., Flynn K. M. (2000) The use of remote sensing and GIS in watershed level analyses of non-point source pollution problems. *Forest Ecology and Management*, 128, 65-73.

Bowman M. S., Soares-Filho B. S., Merry F. D., Nepstad D. C., Rodrigues H., Almeida O. T. (2012) Persistence of cattle ranching in the Brazilian Amazon: A spatial analysis of the rationale for beef production. *Land Use Policy,* 29, 558-568.

Brondizio E. S., Ostrom E., Young O. R. (2009) Connectivity and the governance of multilevel social-ecological systems: The role of social capital. *Annual Review of Environment and Resources,* 34, 253-278.

Cash D. W., Adger W. N., Berkes F., Garden P., Lebel L., Olsson P., Pritchard L., Young O. (2006) Scale and cross-scale dynamics: Governance and information in a multilevel world. *Ecology and Society*, 11.

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. *J Hydrology*, 369, 165-174.

Cumming G. S. (2004) The impact of low-head dams on fish species richness in Wisconsin, USA. *Ecological Applications*, 14, 1495-1506.

FAS (2009) Agricultural economy and policy report - Brazil. Foreign Agriculture Service - U.S. Department of Agriculture.

Fearnside P. (2007) Brazil’s Cuiabá- Santarém (BR-163) highway: The environmental cost of paving a soybean corridor through the Amazon. *Environmental Management,* 39, 601-614.

Fearnside P. F. (2006a) Containing destruction from Brazil's Amazon highways: now is the time to give weight to the environment in decision-making. *Environmental Conservation,* 33, 181-183.

Fearnside P. M. (2006b) Dams in the Amazon: Belo Monte and Brazil's hydroelectric development of the Xingu River basin. *Environmental Management*, 38, 16-27.

Galford G. L., Melillo J. M., Kicklighter D. W., Mustard J. F., Cronin T. W., Cerri C. E. P., Cerri C. C. (2011) Historical carbon emissions and uptake from the agricultural frontier of the Brazilian Amazon. *Ecological Applications*, 21, 750-763.

Galford G. L., Mustard J. F., Melillo J., Gendrin A., Cerri C. C., Cerri C. E. P. (2008) Wavelet analysis of MODIS time series to detect expansion and intensification of row-crop agriculture in Brazil. *Remote Sensing of Environment*, 112, 576-587.

Gergel S. E., Turner M. G., Miller J. R., Melack J. M., Stanley E. H. (2002) Landscape indicators of human impacts to riverine systems. *Aquatic Sciences,* 64, 118-128.

Gibson C. C., Ostrom E., Ahn T. K. (2000) The concept of scale and the human dimensions of global change: a survey. *Ecological Economics*, 32, 217-239.

Grant E. H. C., Lowe W. H., Fagan W. F. (2007) Living in the branches: population dynamics and ecological processes in dendritic networks. *Ecology Letters*, 10, 165-175.

Hurwitz Z., Millikan B., Monteiro T., Widmer R. (2011) Belo Monte. Mega-projeto, Mega-riscos: Análise de riscos para investidores no complexo hidrelétrico Belo Monte. [Belo Monte. Mega-project, Mega-risks: Risk Analysis for investores in the Belo Monte hydroelectric complex.]. São Paulo, Brazil, Amigos da Terra - Amazônia Brasileira. International Rivers.

INPE (2011) Program for the Estimation of Amazon Deforestation (Projeto PRODES Digital). Available at http://www.dpi.inpe.br/prodesdigital/prodes.php. Accessed on January 20, 2011. Brazilian National Agency for Space Research.

International Rivers (2010) Belo Monte: Massive dam project strikes at the heart of the Amazon. Berkeley, CA.

IPAM (2011) Reforma do Código Florestal: qual o caminho para o consenso? Contribuições para o Relatório da Commissão de Meio Ambiente do Senado Federal sobre a reforma do Código Florestal Brasileiro [Reform of the Forest Code: what is the path to consensus? Contributions to the Report of the Senate's Environment Commission on the reform of the Brazilian Forest Code.] Brasília, Brazil, Amazon Environmental Research Institute.

ISA (2003) O Xingu na Mira da Soja. São Paulo, Brazil, Instituto Sociambiental.

Johnson P. T. J., Olden J. D., Zanden M. J. V. (2008) Dam invaders: impoundments facilitate biological invasions into freshwaters. *Frontiers in Ecology and the Environment*, 6, 359-365.

Jones K. L., Poole G. C., Meyer J. L., Bumback W., Kramer E. A. (2006) Quantifying expected ecological response to natural resource legislation: a case study of riparian buffers, aquatic habitat, and trout populations. *Ecology and Society*, 11.

Karr J. R., Schlosser I. J. (1978) Water-Resources and Land-Water Interface. *Science*, 201, 229-234.

Lebel L., Garden P., Imamura M. (2005) The politics of scale, position, and place in the governance of water resources in the Mekong region. *Ecology and Society*, 10.

Lehner B., Liermann C. R., Revenga C., Vörösmarty C., Fekete B., Crouzet P., Döll P., Endejan M. *et al.* (2011) High-resolution mapping of the world's reservoirs and dams for sustainable river-flow management. *Frontiers in Ecology and the Environment*, 9, 494-502.

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, 468-479.

Macedo M., DeFries R., Morton D., Stickler C., Galford G., Shimabukuro Y. (2012) Decoupling of deforestation and soy production in the southern Amazon during the late 2000s. *PNAS*, 109, 1341-1346.

March J. G., Benstead J. P., Pringle C. M., Scatena F. N. (2003) Damming tropical island streams: Problems, solutions, and alternatives. *Bioscience*, 53, 1069-1078.

Morton D. C., DeFries R. S., Shimabukuro Y. E., Anderson L. O., Arai E., Espirito-Santo F. D., Freitas R., Morisette J. (2006) Cropland expansion changes deforestation dynamics in the southern Brazilian Amazon. *Proceedings of the National Academy of Sciences of the United States of America*, 103, 14637-14641.

Naiman R. J., Decamps H. (1997) The ecology of interfaces: Riparian zones. *Annual Review of Ecology and Systematics*, 28, 621-658.

Naiman R. J., Latterell J. J. (2005) Principles for linking fish habitat to fisheries management and conservation. *Journal of Fish Biology*, 67, 166-185.

Narumalani S., Zhou Y. C., Jensen J. R. (1997) Application of remote sensing and geographic information systems to the delineation and analysis of riparian buffer zones. *Aquatic Botany*, 58, 393-409.

Nepstad D., Soares B. S., Merry F., Lima A., Moutinho P., Carter J., Bowman M., Cattaneo A. *et al.* (2009) The end of deforestation in the Brazilian Amazon. *Science*, 326, 1350-1351.

Nepstad D. C., Stickler C. M., Almeida O. T. (2006) Globalization of the Amazon soy and beef industries: Opportunities for conservation. *Conservation Biology*, 20, 1595-1603.

Ostrom E. (1999) Coping with tragedies of the commons. *Annual Review of Political Science,* 2, 493-535.
Ostrom E. (2010) Polycentric systems for coping with collective action and global environmental change. *Global Environmental Change-Human and Policy Dimensions,* 20, 550-557.
Pringle C. M. (2001) Hydrologic connectivity and the management of biological reserves: A global perspective. *Ecological Applications,* 11, 981-998.
Roth N. E., Allan J. D., Erickson D. L. (1996) Landscape influences on stream biotic integrity assessed at multiple spatial scales. *Landscape Ecology,* 11, 141-156.
Schlosser I. J., Johnson J. D., Knotek W. L., Lapinska M. (2000) Climate variability and size-structured interactions among juvenile fish along a lake-stream gradient. *Ecology,* 81, 1046-1057.
Soares B., Alencar A., Nepstad D., Cerqueira G., Diaz M. D. V., Rivero S., Solorzano L., Voll E. (2004) Simulating the response of land-cover changes to road paving and governance along a major Amazon highway: the Santarem-Cuiaba corridor. *Global Change Biology,* 10, 745-764.
Soares B., Moutinho P., Nepstad D., Anderson A., Rodrigues H., Garcia R., Dietzsch L., Merry F. *et al.* (2010) Role of Brazilian Amazon protected areas in climate change mitigation. *Proceedings of the National Academy of Sciences of the United States of America,* 107, 10821-10826.
Soares B. S., Nepstad D. C., Curran L. M., Cerqueira G. C., Garcia R. A., Ramos C. A., Voll E., McDonald A. *et al.* (2006) Modelling conservation in the Amazon basin. *Nature,* 440, 520-523.
Stickler C. M. (2009) Defending public interests in private forests: land use policy alternatives for the Xingu River headwaters region of southeastern Amazônia. Ph.D., Geography. University of Florida, Gainesville, FL, 199 pp.
Teels B. M., Rewa C. A., Myers J. (2006) Aquatic condition response to riparian buffer establishment. *Wildlife Society Bulletin*, 34, 927-935.
Tiemann J. S., Gillette D. P., Wildhaber M. L., Edds D. R. (2005) Effects of lowhead dams on the ephemeropterans, plecopterans, and trichopterans group in a North American river. *Journal of Freshwater Ecology*, 20, 519-525.
Uriarte M., Yackulic C. B., Lim Y., Arce-Nazario J. A. (2011) Influence of land use on water quality in a tropical landscape: a multi-scale analysis. *Landscape Ecology,* 26, 1151–1164.
Walter R. C., Merritts D. J. (2008) Natural streams and the legacy of water-powered Mills. *Science,* 319, 299-304.
Wang L. Z., Infante D., Lyons J., Stewart J., Cooper A. (2011) Effects of dams in river networks on fish assemblages in non-impoundment sections of rivers in MIchigan and Wisconsin, USA. *River Research and Applications*, 27, 473-487.
Wang L. Z., Lyons J., Kanehl P., Gatti R. (1997) Influences of watershed land use on habitat quality and biotic integrity in Wisconsin streams. *Fisheries,* 22, 6-12.
Werth D., Avissar R. (2004) The regional evapotranspiration of the Amazon. *Journal of Hydrometeorology,* 5, 100-109.
Wright J. P., Flecker A. S. (2004) Deforesting the riverscape: the effects of wood on fish diversity in a Venezuelan piedmont stream. *Biological Conservation*, 120, 439-447.

Chapter 2

Decoupling of deforestation and soy production in the southern Amazon during the late 2000s[9]

Abstract

From 2006-2010 deforestation in the Amazon frontier state of Mato Grosso decreased to 30% of its historical average (1996-2005) whereas agricultural production reached an all-time high. This study combines satellite data with government deforestation and production statistics to assess land-use transitions and potential market and policy drivers associated with these trends. In the forested region of the state, increased soy production from 2001-2005 was entirely due to cropland expansion into previously cleared pasture areas (74%) or forests (26%). From 2006 to 2010, 78% of production increases were due to expansion (22% to yield increases), with 91% on previously cleared land. Cropland expansion fell from 10 to 2% of deforestation between the two periods, with pasture expansion accounting for most remaining deforestation. Declining deforestation coincided with a collapse of commodity markets and implementation of policy measures to reduce deforestation. Soybean profitability has since increased to pre-2006 levels while deforestation continued to decline, suggesting that anti-deforestation measures may have influenced the agricultural sector. We found little evidence of direct leakage of soy expansion into cerrado in Mato Grosso during the late 2000s, although indirect land use changes and leakage to more distant regions are possible. This study provides evidence that reduced deforestation and increased agricultural production can occur simultaneously in tropical forest frontiers, provided that land is available and policies promote the efficient use of already-cleared

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lands (intensification), while restricting deforestation. It remains uncertain whether government- and industry-led policies can contain deforestation if future market conditions favor another boom in agricultural expansion.

Introduction

Global markets for commodities such as oil palm and soybeans are increasingly replacing local demand as the primary driver of tropical forest conversion for agriculture (DeFries *et al.*, 2010, Gibbs *et al.*, 2010, Rudel *et al.*, 2009a). As global demand for food, fiber, and biofuels grows to unprecedented levels, the supply of available land continues to shrink (Lambin & Meyfroidt, 2011). Most of this land is concentrated in tropical forest regions, fueling debate about how to reconcile the need for agricultural production with forest conservation and maintenance of ecosystem services such as carbon storage, climate regulation, and biodiversity conservation. Many argue that intensification and the productive use of already cleared lands is a pathway to meeting these objectives (DeFries *et al.*, 2004, DeFries *et al.*, 2010, Matson & Vitousek, 2006, Nepstad *et al.*, 2009, Tilman *et al.*, 2002). Others conclude that intensification itself does not reduce pressure on forests and that, in the absence of effective conservation policies, increased yields can stimulate additional deforestation (Angelsen, 2010, Rudel *et al.*, 2009b) via direct agricultural encroachment or displacement of other land uses (Arima *et al.*, 2011, Lambin & Meyfroidt, 2011). To date, empirical examples that test these assertions have been limited to national-scale analysis and scenarios (Lambin & Meyfroidt, 2011, Rudel *et al.*, 2009a), with few concrete cases where increased production and forest conservation occurred simultaneously. Here we combine satellite data with government statistics on deforestation and production to track forest clearing and postdeforestation land uses during a decade of historic agricultural expansion in the state of Mato Grosso (MT), Brazil. The resulting dataset enables a spatially-explicit analysis of trends in production and deforestation, whether and where intensification and reduced deforestation occurred simultaneously, and the accompanying market and policy context.

The Amazon's "arc of deforestation" has been the world's most active deforestation frontier in recent decades. The frontier states of Mato Grosso, Rondônia, and Pará accounted for 85% of all Amazon deforestation from 1996-2005, converting an average of 16,600 km^2yr^{-1} of forest (INPE, 2011). The underlying forces driving agricultural expansion in the region shifted dramatically in the last two decades (DeFries & Rosenzweig, 2010, Nepstad *et al.*, 2006). Deforestation in the 1970s and 1980s was driven by a combination of government subsidies for Amazon development, investments in road infrastructure, unclear land tenure, and policies that promoted land speculation by rewarding deforesters with formal land titles (Fearnside, 2005). The last decade saw the removal of many policies that stimulated deforestation and an increasing influence of global markets on the Amazon economy (Cattaneo, 2008, Nepstad *et al.*, 2009).

From 2006 to 2010 deforestation in the Amazon declined dramatically, particularly in Mato Grosso. The state is situated in the agricultural frontier and occupies 900,000 km^2, divided between tropical forest (Amazon) and savanna/grassland (Cerrado) ecosystems (Fig. A.1). Mato Grosso is Brazil's leader in soy and beef production, responsible for 31% of the nation's soy production and over 13% of its cattle herd in 2009 (IBGE, 2009). From 2000 to 2005 it also led in deforestation, accounting for 40% of deforestation in the Brazilian Amazon. In the ensuing years, deforestation in Mato Grosso declined substantially, reaching an estimated 850 km^2 by 2010 (INPE, 2011) – just 11% of its historical average (7,600 km^2yr^{-1} from 1996 to 2005; Fig. 2.1). These declines in deforestation coincided with fluctuations in commodity markets and the implementation of several high-profile policy initiatives aimed at restricting credit for deforesters, improving monitoring and enforcement, and excluding deforesters from the supply chains of major exporters.

Although expansion of cattle ranching continues to be the primary proximate driver of deforestation, the expansion of mechanized agriculture (croplands) altered deforestation dynamics, both directly by increasing conversion of forests for soy cultivation (Morton *et al.*, 2006) and indirectly by replacing existing cattle pastures, some of which moved into other forested regions (Nepstad *et al.*, 2006). The replacement of extensive land uses (e.g., cattle pastures) with intensive production (e.g., soybeans) is often referred to as "intensification", whereas the replacement of natural vegetation (e.g., forest or cerrado) with extensive land uses is termed "extensification". This terminology is complicated by the case of direct conversion of natural vegetation for intensive agriculture, which incorporates some elements of both. In lieu of this terminology we distinguish among cropland expansion into already-cleared lands, cropland expansion into forests, and pasture expansion into forests.

As deforestation in Mato Grosso decreased after 2005, soy production in the state continued its upward trend (Fig. 2.1), following a dip in 2006 and 2007 when commodity prices dropped precipitously. This decoupling of soy production from deforestation is a departure from trends during the first half of the decade, when deforestation tracked changes in soy and cattle production (Galford *et al.*, 2010). Whereas the first half of the decade contradicts the hypothesis that intensification inevitably leads to land sparing, the latter half suggests that in certain contexts it can. This study combines satellite and field data with Brazilian government data on deforestation and production to quantify land-use transitions in the forested region of Mato

Grosso from 2001 to 2010[10]. We analyze Moderate Resolution Imaging Spectroradiometer (MODIS) data to develop spatially and temporally explicit estimates of transitions from forest to pasture or cropland and from already-cleared land (primarily pasture) to cropland. This analysis extends our previous time series of land-use transitions (Morton *et al.*, 2006) and allows us to examine the changing dynamics associated with substantial declines in deforestation in the latter half of the decade. We focus on two central questions: 1) What land-use transitions – cropland expansion into forest, expansion into already cleared lands, or changes in yields – occurred during the 2000s? How do trends vary between the first and second halves of the decade? 2) Was declining deforestation from 2006 to 2010 associated with fluctuations in commodity markets, policy interventions, or both?

Results and Discussion

Trends in soy production

Land-use transitions differed dramatically between the periods 2001-2005 and 2006-2010 (Fig. 2.2). The first period corresponded to a boom in cropland (primarily soy) expansion, with the area planted in soy doubling from 3 to 6 million ha (Fig. A.2) and production increasing by 85% (Fig. 2.1), or 8 million tons (IBGE, 2011b). A third of that increase in area (~1 million ha) and production (~3 million tons) occurred in the Amazon forest biome, where planted area more than tripled during the same period (Fig. A.3; IBGE, 2011b). Rising demand for soy was primarily related to export markets for animal fodder in Europe and Asia (Nepstad *et al.*, 2006, Nepstad *et al.*, 2008). While the majority of soy expansion replaced cattle pastures, an average of 12% of the area in large clearings (> 25 ha) each year was directly converted from forest to

[10] Growing years span from August in the year of planting through July in the year of harvest. Unless otherwise specified, the years of analysis refer to growing years and are labeled by the harvest year.

cropland (Fig. 2.3). Our results support those previously reported for 2001-2005 (Morton *et al.*, 2006), with a clear peak in deforestation for soy (18.5%) in 2003.

The second half of the decade paints a very different picture. Soy planted area in Mato Grosso contracted by nearly 1 million ha, as commodity prices crashed in 2006 and 2007. The area planted in soy increased each year since, but by 2010 still had not recovered to the highest levels recorded in 2005 (Fig. A.2). After its peak in 2003, our analysis indicates that the percentage of large-scale (>25 ha) deforestation due to soy expansion decreased steadily, reaching 1% in 2009 (Fig. 2.3). The number of large clearings decreased markedly during the second half of the study period, representing an average of 85% of the deforested area from 2001 to 2005 and 65% from 2006 to 2009 (Fig. A.4). This is consistent with previous work showing that deforestation during this latter period occurred primarily at the edge of existing fields or pastures (Rudorff *et al.*, 2011), rather than through new large-scale expansion into forests. Despite overall reductions in deforestation and a temporary contraction in area planted, the forested region of MT saw a net increase in annual production of 750,000 tons between the 2005 and 2009 harvests (Fig. 2.4), roughly 25% of the increase observed in the first half of the decade.

Using our MODIS-derived soy distribution data and the state vegetation map (Fig. A.1), we spatially allocated annual data on municipal soy production and area planted (IBGE, 2011b) by biome. The resulting land-use transition maps allowed us to examine whether annual changes in production within Mato Grosso's forested region were due to deforestation, expansion into already-cleared areas, or changes in yield (Fig. 2.4 and Fig. A.5). Short-term changes in yield may be influenced by several factors, including rainfall variability, emergence of crop diseases, changes in planting technology, and the time required to build up soil fertility (~ 2-3 y). As expected, the boom from 2001 to 2005 was largely due to cropland expansion, with increases in area planted accounting for steady increases in production. This pattern shifted noticeably in 2006 and 2007, when area planted and overall production decreased. The next two years saw a recovery in production attributable to increases in yield (2008) and area planted (2009). During the latter half of the decade, cropland expansion in Mato Grosso's forested region occurred largely in previously cleared lands (primarily pasture), which accounted for 91% (318,000 ha) of expansion from 2006 to 2010, in contrast to 74% (800,000 ha) during the boom period (Fig. 2.4).

Trends in pasture expansion

As soy became more profitable in the region the price of land increased, as did the opportunity cost of holding land for livestock production (Cattaneo, 2008). During the boom period in soy expansion (2001-2005), the incentive for cattle producers was to sell their land at a profit and clear more land elsewhere (Nepstad *et al.*, 2006). This displacement effect is difficult to quantify, although it is clear that the two sectors are strongly interconnected (Nepstad *et al.*, 2008). Recent studies suggest that soy expansion and intensification in Mato Grosso during the early part of the decade displaced cattle ranching northward into neighboring states (Arima *et al.*, 2011, Barona *et al.*, 2010). This phenomenon may have been partially mitigated by improvements in livestock technology introduced in the center-west to keep up with the profitability of soy in the region (Cattaneo, 2008). Improvements in pasture management and phyto-sanitary measures aimed at keeping the herd free of hoof and mouth disease may have been crucial to limiting indirect impacts of soy expansion, avoiding an estimated 6,000-10,000 km^2yr^{-1} of additional deforestation (Cattaneo, 2008).

Our MODIS-based analysis indicates that large-scale clearings of forest for pasture decreased rapidly after 2005, dropping over 70% from 2005 to 2006 alone (Fig. 2.3). These reductions in cattle expansion made the biggest contribution to deforestation reductions observed after 2005, suggesting that market signals and policy measures aimed at reducing illegal deforestation may have had a broad impact. The increasing costs of expansion were concurrent with a move towards intensification, as many of the state's cattle producers replaced extensive grazing (< 1 head of cattle per ha) with confinement of animals in feedlots for part of the growing period – a practice that grew by 286% from 2005 to 2008 (ACRIMAT, 2010). Such intensification allows for local consumption of second-harvest crops (millet, sorghum, and corn) and potentially releases land for other agricultural uses.

Market trends

From 2001 to 2009 deforestation for soy was weakly correlated with the profitability per 60 kg sack of soy (Fig. A.6a; R^2=0.39, n=9), defined as the difference between the variable costs of production and the price received by producers in Mato Grosso (Fig. 2.3). The farm gate price of cattle showed virtually no correlation with deforestation for pasture (Fig. A.6b; R^2=0.04, n=9) during the same period. These relationships become considerably stronger if we consider only the years prior to 2008, with soybean profitability and cattle prices explaining significantly more of the variation in cropland deforestation (Fig. A.6a; R^2=0.64, n=7) and pasture deforestation (Fig. A.6b; R^2=0.89, n=7), respectively. Although based on a limited number of observations, these trends suggest that high profitability was a strong incentive for soy and cattle expansion into forested areas during the boom period and that decreases in deforestation from 2003 to 2007 were at least partially due to declines in profitability. This trend is supported by a recent econometric analysis for 783 municipalities, indicating that fluctuations in meat and soybean prices drove deforestation in the region from 2002 to 2007 (Silva, 2009). Decreased profitability in the latter half of the decade was associated with a global crash in commodity markets and increases in the variable costs (CONAB, 2011) of soy production (e.g., seeds and fertilizers), which may have temporarily removed incentives for expansion. Despite the recovery of soy and cattle prices after 2007, deforestation did not increase as in the early part of the decade (Rudorff *et al.*, 2011). Rather, expansion of soy during this period occurred almost exclusively on previously cleared (pasture) lands (Fig. 2.4). Expansion of cattle ranching also decreased during this period, presumably as a result of the market contraction and a move towards intensification in Mato Grosso (ACRIMAT, 2010).

Policy initiatives

Although profitability and macroeconomic trends almost certainly affect the short-term decision-making of producers, it is difficult to isolate their impact from that of government- and industry-led policies introduced during the same period. In response to increasing deforestation in the mid-1990s and the decentralization of environmental regulatory powers, Mato Grosso implemented an integrated system of environmental licensing and management, which introduced regular satellite-based monitoring of deforestation (Azevedo, 2009, Fearnside, 2003, Stickler, 2009). Despite implementation of this system, deforestation rates continued to climb. As they reached their peak in 2004, the federal government established a national plan to control deforestation in the Amazon, requiring states to develop and implement their own deforestation control programs (Abdala *et al.*, 2008). In an attempt to curtail corruption related to licensing for logging and clearing, the federal government implemented real-time monitoring of deforestation and carried out raids, which led to the imprisonment of employees in several state and federal agencies and reorganization of the Mato Grosso state environmental agency (Nepstad *et al.*, 2009, Stickler, 2009). Finally, in 2008 the federal government created a “black list” of municipalities with high deforestation rates, imposing a series of sanctions on producers in those municipalities, including eliminating subsidies, restricting credit, halting all (legal) deforestation, and issuing fines for illegal clearing and burning (Nepstad *et al.*, 2009, Stickler & Almeida, 2008).

Two agroindustry-led initiatives to reduce deforestation accompanied the government-led enforcement initiatives described above. The first was a 2006 “soy moratorium” (ABIOVE, 2010), which excluded all soy cultivated in areas deforested after that date from the supply chains of major exporters (Stickler & Almeida, 2008). Prompted by pressure from international environmental organizations and demand from environmentally conscious consumers, it served as a model for a similar moratorium in the beef and leather industry, declared in 2009 by the four largest cattle producers and traders. These demand-driven disincentives to deforestation are relatively new forces in the region, complementing government enforcement measures and bolstering existing certification schemes to reward environmentally responsible production (Nepstad *et al.*, 2006, Nepstad *et al.*, 2008).

The land-use transitions observed during the postboom period – and the case of 2009 in particular – suggest that when market conditions favored expansion, producers expanded into areas previously cleared for pasture rather than forest areas (Fig. 2.4 and Fig. A.5). These patterns are consistent with the outcomes expected by many of the recent policy interventions, providing some support for the hypothesis that they have helped to suppress deforestation. An alternate explanation is that, even in the absence of policy reforms, the market-induced contraction in soy area planted provided sufficient fallow cropland to absorb soy expansion in the years following the market decline. Had this been the case, we would expect no increase in the cumulative area planted from 2006 to 2010 (i.e., no new cropland). In fact, our MODIS estimates indicate that there was a steady increase in cumulative area planted after 2005 (Fig. A.7a) while deforestation was suppressed, suggesting a shift (proportionally) from soy expansion into forest to soy expansion into previously cleared lands during this period (Fig. 2.2). Combining our satellite analyses with PRODES data on the year of clearing (INPE, 2011) provides further evidence that this shift was not simply due to a glut of land cleared during the boom period (Nepstad *et al.*, 2009). Rather, about two-thirds of non-forest areas converted to soy during this period were cleared prior to 2000 (50% prior to 1997) and the remaining third was cleared from 2001 to 2005 (Fig. 2.4b). Because Mato Grosso had little mechanized crop production prior to 2000 (Fig. A.3; IBGE, 2011b), we assume that lands cleared prior to that date were originally cleared for pasture.

Leakage

One potential byproduct of reductions in deforestation and cropland expansion in the forested region of Mato Grosso is leakage into the state's cerrado or into forested areas of neighboring states. Theoretically, such leakage can occur at multiple scales (Lambin & Meyfroidt, 2011) and could take the form of direct conversion of natural vegetation for cropland or indirect land-use changes associated with the displacement of cattle ranching (Arima *et al.*, 2011). To examine direct leakage within Mato Grosso, we used our satellite-derived data on cropland area and the state vegetation map (Fig. A.1) to assess whether decreased deforestation in the postboom period displaced soy expansion into the state's cerrado region. Based on patterns of soy area planted in each biome, we saw no evidence of an overall increase in soy expansion into the state's cerrado since 2005. Planted area in both biomes exhibited similar trends throughout the study period (Fig. A.7). Moreover, an analysis of deforestation polygons in Mato Grosso's cerrado (Ferreira *et al.*, 2007) indicates that deforestation for cropland decreased from 2003 to 2006 and remained relatively low for the remainder of the decade (Fig. A.8). These trends provide indirect evidence that reduced deforestation in the forest region did not provoke an immediate increase in clearing of cerrado for soy in the latter half of the decade.

Previous studies have linked soybean expansion in Mato Grosso to the displacement of pastures into Pará (Barona *et al.*, 2010), Rondônia, and Amazonas (Arima *et al.*, 2011) based on municipality-level agricultural statistics. At the state level, annual deforestation rates in Pará and Rondônia (INPE, 2011) decreased considerably after 2005 (Fig. A.8) and do not suggest substantial leakage (direct and indirect) from Mato Grosso in the short term. However, small or isolated leakage effects may be masked by a number of other factors affecting deforestation at the state level, including changing markets, governance (Mandemaker *et al.*, 2011), enforcement capacity, agrarian reform, and land speculation. Prevention of leakage in the cattle sector is of particular concern, given the Brazilian government's commitment to decreasing deforestation and land-use related carbon emissions (Gouvello, 2010). The present study focuses on the soybean sector because it played a catalytic role in increasing deforestation during the first half of the decade, but this is only part of the equation. Controlling deforestation over the long term will likely hinge on what happens in the cattle sector, where there are greater opportunities for gains in efficiency through intensification (Gouvello, 2010). The information presented here does not preclude lagged effects, whereby recent land use dynamics result in future leakage, or eliminate the possibility that leakage may already be underway at finer scales or in more distant regions. Establishing that leakage is occurring from Mato Grosso would require more in-depth analysis of the political context, migration patterns, and socioeconomic motivation of producers in those regions.

Conclusions

The combination of MODIS-derived land-use information with government agricultural and deforestation statistics allowed a spatially explicit analysis of land-use transitions associated with declining deforestation and increasing production in Mato Grosso's forested region from 2006 to 2010. The analysis leads to three conclusions. First, after 2005 the increase in soy production was partially due to relatively high yields (e.g., 2008), but mainly to a proportional increase in soy expansion onto previously cleared land compared to the first half of the decade. The observed patterns provide evidence that it is possible to achieve the dual objectives of forest conservation and agricultural production (Angelsen, 2010, DeFries & Rosenzweig, 2010, Matson & Vitousek, 2006) in contexts where there is a sufficient supply of previously cleared land and incentives that encourage productive use of that land instead of expansion into forests. Although this outcome is positive for forests and food production, there are likely additional synergies and tradeoffs inherent in the expansion of intensive production, even if constrained to previously cleared lands. On one hand is the synergistic potential for improved farm-level management (e.g., no tillage, cover crops) to enhance crop productivity and soil carbon storage. On the other are potential trade-offs with biodiversity loss, altered hydrological function, and runoff of agrochemicals. Furthermore, the observed decreases in deforestation do not guarantee that remaining forests are pristine, considering recent evidence that forest degradation in the region is increasing due to logging (Asner *et al.*, 2006) and fire (Aragao & Shimabukuro, 2010). This may diminish the benefits of reduced deforestation for climate and forest conservation.

Second, deforestation for cropland in Mato Grosso remained low even when profitability favored soy expansion. In 2008, profitability peaked to levels comparable to those during the 2000-2005 boom, yet deforestation for soy continued to decrease (Fig. 2.3). These decreases may be partially explained by increases in the variable cost of soy production, which decreased profitability relative to the first half of the decade. These trends were also concurrent with the implementation of policies aimed at restricting credit for deforesters, improving monitoring and enforcement, and excluding deforesters from the supply chains of major exporters. Observed patterns suggest that they have had some success. However, the implementation of the policies mentioned here occurred at a time when market conditions already favored a slowing in deforestation. Whether this coincidence was strategic or serendipitous, it likely helped in achieving deforestation reductions during the late 2000s. Quantifying the relative influences of concurrent market drivers and policy interventions requires more detailed analyses of landholder responses to different incentives.

Finally, Mato Grosso's reduction of deforestation after 2005 did not result in a net increase of soy expansion into the state's cerrado. Deforestation in Pará and Rondônia also declined, suggesting that the patterns observed in Mato Grosso did not provoke a major net increase in clearing in adjacent Amazonian states during the study period. It is possible that the advancing wave of soy production into the Amazon has already exhausted suitable lands for agricultural production in Mato Grosso´s cerrado or that forested areas in neighboring states are less suitable for cropland, neither of which is captured by the data presented here. Over the last decade, expansion into previously cleared lands and intensification of crop (Galford *et al.*, 2010) and cattle production may also have mitigated potential leakage into other regions (Cattaneo, 2008). Although the large supply of low productivity pasture land presents an opportunity for gains in efficiency and mitigation of future leakage, this result is by no means guaranteed. There is already evidence of recent soy expansion into cerrado areas further east and northeast in the country, particularly in the states of Bahia, Maranhão, Piauí, and Tocantins (IBGE, 2011b), although it is unclear if these trends reflect leakage from the southern Amazon.

The Brazilian government's investments in monitoring and enforcement of deforestation have created powerful disincentives for expansion into forest lands (Silva, 2009), complemented by voluntary industry initiatives. While these efforts have had some success, our results suggest that preventing deforestation over the long-term will require parallel efforts to modernize the cattle sector and create strong new policy incentives that promote efficient use of degraded lands. Recent efforts to model Brazil's low-carbon development alternatives indicate that the implementation of existing technologies to restore degraded lands and increase pasture productivity could free enough additional land to accommodate projected growth through 2030 (Gouvello, 2010), although achieving this would be challenging and require substantial private and public investments.

Mato Grosso has considerable remaining forest land that is suitable for agricultural production (Fig. A.10), and advances in infrastructure and technology will likely increase access to these and other Amazon forests (Nepstad *et al.*, 2008). Reports of increased deforestation in Mato Grosso during the first semester of 2011 have already raised concerns that soaring commodity prices and proposed changes to Brazil's Forest Code may soon reverse recent trends in deforestation. If Brazil is to build on its successes in reducing deforestation and continue the trend towards becoming one of the world's major food producers, it will require continued implementation of policies that conserve standing forests while directing agricultural expansion onto previously cleared lands. If successful, initiatives like the UNFCCC REDD+ program (UNFCCC, 2010) and national efforts to promote low carbon development could help sustain lower deforestation rates by providing mechanisms to compensate actors for avoiding deforestation and increasing productivity. Although our results pertain to the specific context of Mato Grosso in the last decade, the approach of tracking postclearing land uses can yield insights into the changing drivers of deforestation in other tropical forest regions. Demands for export-oriented agricultural products will likely continue to exert pressure for expansion into forested regions (DeFries *et al.*, 2010) at the same time that carbon markets and consumer demand call for decreased deforestation. National, state, and local governments will need to consider context-specific strategies and policy incentives to balance these objectives.

Methods

Data

Data on soy production and area planted came from the Brazilian Institute of Geography and Statistics (IBGE, 2011b), and annual deforestation data from Brazilian National Institute for Space Research (INPE, 2011). Data on the farm gate price of soy and cattle in Mato Grosso came from the Getúlio Vargas Foundation (FGV, 2011a, FGV, 2011b), and cost data from the National Food Supply Company of Brazil (CONAB, 2011). The IBGE provided historical data on the expanded consumer price index (IBGE, 2011a) and 2007 municipal boundaries[11].

[11] Available at http://www.ibge.gov.br/home/geociencias/geografia/. Accessed on December 10, 2010.

Collection 5 MODIS enhanced vegetation index (EVI) data for the study area came from the Land Processes Distributed Active Archive Center (LPDAAC)[12].

Remote sensing

We used the MODIS EVI product (MOD13Q1) to perform annual land-use classifications based on differences in vegetation phenology, an approach that is conceptually similar to that of previous studies (Galford *et al.*, 2008, Morton *et al.*, 2006). Given changes in the MODIS data (collection 4-5) and variation in the details of our methodology, we processed the entire 10-y time series for this analysis. First, we eliminated cloud-contaminated pixels and replaced missing data values using a spline interpolation in the time (z) dimension. For each growing year we calculated SD, annual mean, dry season mean (July), wet season mean (December-February), and wet season maximum. Based on these metrics, we developed a decision tree classifier using 326 ground data points collected in 2006 (Stickler *et al.*, 2009) to classify cropland, forest, and pasture/cerrado for each year (Fig. A.11). Finally, we filtered the classified time series using a 3-y filter to remove unlikely land-use transitions through time. This correction affected at least one observation in 13% of the pixels monitored (< 2% of all observations).

The final land-use classification (Fig. A.12) was validated using 317 data points collected in 2010 and distinguishes the three classes of interest with an overall accuracy of 92% (Table A.1). Given the moderate resolution of MODIS data (250-m), we cannot reliably evaluate edge pixels or areas smaller than 25 ha (Morton *et al.*, 2005), which accounted for an increasing proportion of deforestation during the study period (Rudorff *et al.*, 2011). As a result, we may

[12] Available at http://mrtweb.cr.usgs.gov/. Accessed on January 14, 2011.

underestimate deforestation for cropland, particularly towards the end of the time series. Nevertheless, most of the area in production occurs in clearings considerably larger than 25 ha (Morton *et al.*, 2006), allowing us to characterize overall land-use trends.

Postdeforestation land use

To determine the postdeforestation land use, we combined INPE's high resolution (30-m) deforestation data with our land-use classification, a method similar to that published by Morton and coauthors (Morton *et al.*, 2006). First, we used the state vegetation map (Fig. A.1) to mask out areas that were not historically forest. For each deforestation year (September through August), we selected large deforestation polygons (> 25 ha) and classified each according to the majority land use within its boundaries in the subsequent 3 y. Polygons identified as cropland in any of the following 3 y were classified as deforestation for cropland. Polygons identified as pasture in the 3 y after clearing were classified as deforestation for pasture. Polygons identified as forest in all 3 postdeforestation y were classified as not in production and likely include damaged forests that were never fully cleared (e.g., logged or burned), edge effects from adjacent forest cover, and regrowth (Morton *et al.*, 2006). We used the same approach for analysis of cerrado deforestation polygons (Ferreira *et al.*, 2007).

Planted area and production

We combined IBGE municipal boundaries and the potential vegetation map (Fig. A.1) to allocate production and planted area data to the Cerrado and Amazon forest biomes. Municipalities with most of their area in one biome (> 80%) were automatically assigned to that biome (~ 70% of municipalities). Remaining municipalities were evaluated according to the majority biome, municipal area, and cumulative area planted during the study period to identify cases where assignment to the majority biome could result in misallocation of croplands. In these cases we used our annual land-use classification to determine the proportion of soy area located in each biome in a given year. This correction affected 10% of all municipalities and reduced errors that would have occurred had we assumed that mixed municipalities were in a single biome based on the majority vegetation type. Performing the same allocation using state level data did not change the results substantially (r = 0.98) and we have reported municipality level results here.

Market trends

Our calculation of soy profitability is based on the variable costs of production – those costs associated with planting, harvest, storage, and transport of a single soy crop. Our analysis excludes fixed costs (e.g., depreciation of machinery), which are less likely to influence short-term decisions (Angelsen, 2010). After using an expanded consumer price index (IBGE, 2011a) to adjust price and cost data to the July 2010 Real, we calculated the difference between soy price and production costs. The resulting index estimated the profit per 60-kg sack of soybeans in each growing year. In the absence of comparable data on the cost of cattle production, we used inflation-adjusted data on the farm gate cattle price (FGV, 2011a) to examine the relationship between markets and deforestation for cattle. To assess the influence of temporal autocorrelation on the statistical models, we compared parameter estimates from models with and without an autocorrelation error structure. Because including the autocorrelation structure did not change the results, we present only the estimates derived from ordinary least squares regression.

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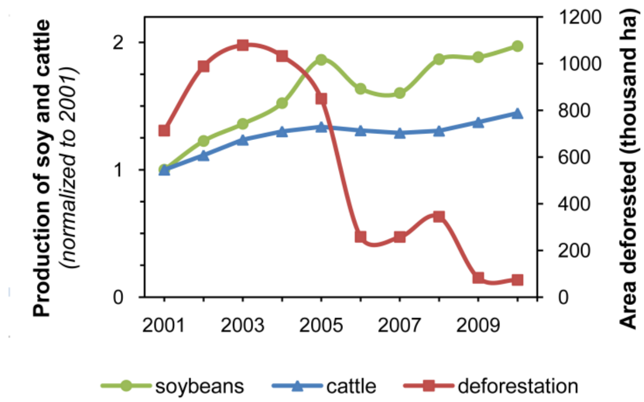

Figure 2.1: Deforestation in Mato Grosso (INPE, 2011), tons of soy produced (IBGE, 2011b), and number of heads of cattle produced (IBGE, 2009) from 2001-2010. Production was normalized to 2001. Production increases correspond to an area increase of 3 million ha for cropland (soy) and 10 million ha for pasture (assuming one head of cattle per ha).

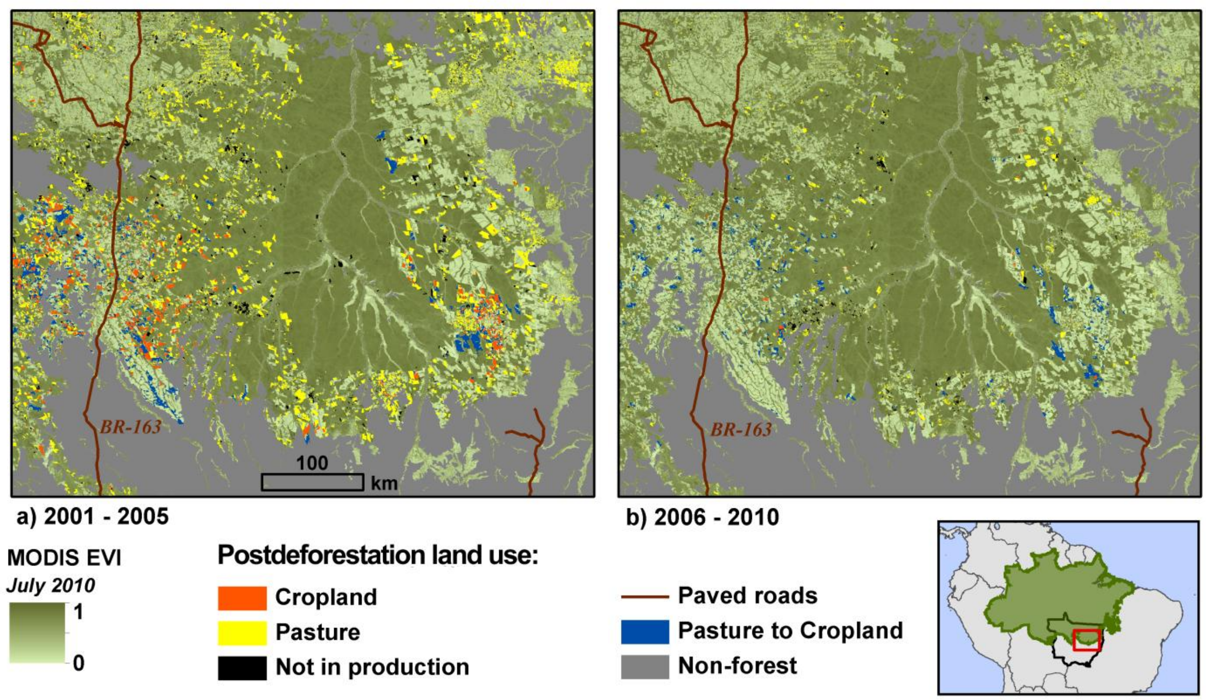

Figure 2.2: Postdeforestation land uses in a subset of the study region (inset) from 2001 to 2005 (a) and 2006 to 2010 (b). Deforestation areas >25 ha were derived from the PRODES dataset (INPE, 2011), and land use from analysis of the MODIS EVI time series. The Brazilian Amazon forest biome is shaded in green (*Lower Right*).

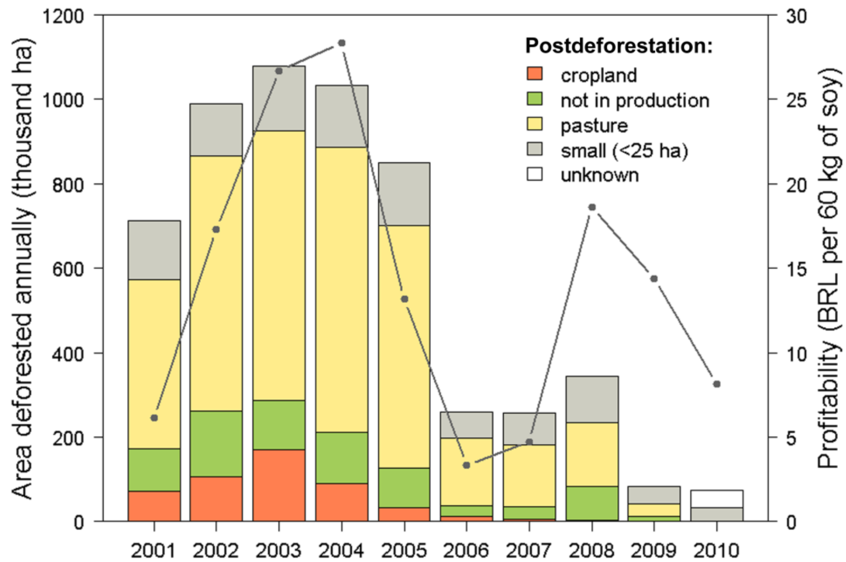

Figure 2.3: Deforestation in Mato Grosso from 2001 to 2010. Postdeforestation land uses for large (>25 ha) clearings were derived from the PRODES dataset (INPE, 2011) and the MODIS EVI time series. Profitability was calculated from state-level data on price received for soy (FGV, 2011b) and cost of production (CONAB, 2011), in Brazilian Reais (BRL). Soy profitability was correlated with cropland deforestation until 2007 (R^2 = 0.64, n=7).

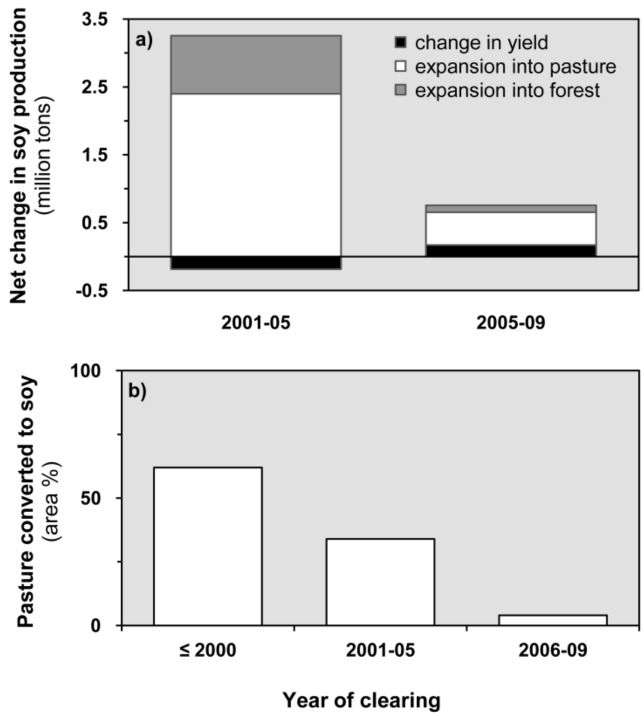

Figure 2.4: Trends in soy expansion during the study period. (a) Attribution of net changes in soy production in the forested region of Mato Grosso to yield, expansion into forest, and expansion into previously cleared (primarily pasture) land. From 2001 to 2005, increases in production were due entirely to expansion into forest (26%) and pasture (74%). From 2005 to 2009, increases in yield accounted for 22% of production increases and most (91%) cropland expansion occurred into pasture. (b) Of the pasture converted to soy from 2005 to 2009, about two-thirds represented old clearings deforested prior to 2000. These results were based on IBGE municipal agricultural data (IBGE, 2011b) and PRODES deforestation data (INPE, 2011), spatially allocated using the MODIS time series.

References

Abdala G. C., Lima A., Pires M. O., Carvalho F., Soares N., Resende R., Abirached C. F., Borges L. M. *et al.* (2008) Plano de Ação para Prevenção e Controle do Desmatamento na Amazônia Legal – PPCDAM: Documento de Avaliação 2004-2007 [Action Plan for the Prevention and Control of Deforestation in the the Legal Amazon - PPCDAM: Evaluation Document 2004-2007]. Brasilia, Brazil, Ministry of Environment: DPCD-SECEX-MMA.

ABIOVE (2010) Sustainability - Soy Moratorium. Available at http://www.abiove.com.br/english/ss_moratoria_us.html. Accessed on January 07, 2011. Brazilian Association of Vegetable Oil Industries

ACRIMAT (2010) Characterization of cattle ranching in the state of Mato Grosso. Cuiaba, MT, Brasil, Cattle Raisers Association of Mato Grosso.

Angelsen A. (2010) Policies for reduced deforestation and their impact on agricultural production. *Proceedings of the National Academy of Sciences of the United States of America*, **107**, 19639-19644.

Aragao L. E. O. C., Shimabukuro Y. E. (2010) The incidence of fire in Amazonian forests with implications for REDD. *Science,* **328**, 1275-1278.

Arima E. Y., Richards P., Walker R., Caldas M. M. (2011) Statistical confirmation of indirect land use change in the Brazilian Amazon. *Environmental Research Letters,* **6**, 024010.

Asner G. P., Broadbent E. N., Oliveira P. J. C., Keller M., Knapp D. E., Silva J. N. M. (2006) Condition and fate of logged forests in the Brazilian Amazon. *Proceedings of the National Academy of Sciences of the United States of America,* **103**, 12947-12950.

Azevedo A. A. (2009) Legitimating Unsustainability? Analysis of the Environmental Licensing System for Rural Properties - SLAPR (Mato Grosso). Ph.D. in Sustainable Development, Center for Sustainable Development (CDS). University of Brasília, Brasília, Brasil, 325 pp.

Barona E., Ramankutty N., Hyman G., Coomes O. T. (2010) The role of pasture and soybean in deforestation of the Brazilian Amazon. *Environmental Research Letters,* **5**.

Cattaneo A. (2008) Regional comparative advantage, location of agriculture, and deforestation in Brazil. *J of Sustainable Forestry,* **27**, 25-42.

CONAB (2011) Cost of production of summer crops - time series. Available at http://www.conab.gov.br. Accessed on March 15, 2011., Companhia Nacional do Abastecimento.

DeFries R., Rosenzweig C. (2010) Toward a whole-landscape approach for sustainable land use in the tropics. *Proceedings of the National Academy of Sciences of the United States of America,* **107**, 19627-19632.

DeFries R. S., Foley J. A., Asner G. P. (2004) Land-use choices: balancing human needs and ecosystem function. *Frontiers in Ecology and the Environment,* **2**, 249-257.

DeFries R. S., Rudel T., Uriarte M., Hansen M. (2010) Deforestation driven by urban population growth and agricultural trade in the twenty-first century. *Nature Geosci,* **3**, 178-181.

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**, 680-688.

Ferreira N., Ferreira L., Huete A., Ferreira M. (2007) An operational deforestation mapping system using MODIS data and spatial context analysis. *International Journal of Remote Sensing,* **28**, 47-62.

FGV (2011a) Index of Prices Received (IPR) for Cattle - Mato Grosso Getúlio Vargas Foundation - Brazilian Institute of Economics.

FGV (2011b) Index of Soy Prices Received (IPR) - Mato Grosso Getúlio Vargas Foundation - Brazilian Institute of Economics.

Galford G. L., Melillo J., Mustard J. F., Cerri C. E. P., Cerri C. C. (2010) The Amazon frontier of land-use change: Croplands and consequences for greenhouse gas emissions. *Earth Interactions,* **14**.

Galford G. L., Mustard J. F., Melillo J., Gendrin A., Cerri C. C., Cerri C. E. P. (2008) Wavelet analysis of MODIS time series to detect expansion and intensification of row-crop agriculture in Brazil. *Remote Sensing of Environment*, **112**, 576-587.

Gibbs H. K., Ruesch A. S., Achard F., Clayton M. K., Holmgren P., Ramankutty N., Foley J. A. (2010) Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. *Proceedings of the National Academy of Sciences of the United States of America*, **107**, 16732-16737.

Gouvello C. (2010) Brazil low-carbon country case study. The World Bank Group.

IBGE (2009) Municipal Cattle Production. In: *Produção Pecuária Municipal.* Rio de Janeiro, Brazil, Brazilian Institute of Geography and Statistics.

IBGE (2011a) Extended Consumer Price Index (IPCA). Available at http://seriesestatisticas.ibge.gov.br/. Accessed on March 15, 2011. Brazilian Institute of Geography and Statistics.

IBGE (2011b) Municipal Agricultural Production. Available at http://seriesestatisticas.ibge.gov.br/. Accessed on March 1, 2011. Brazilian Institute of Geography and Statistics.

INPE (2011) Program for the Estimation of Amazon Deforestation (Projeto PRODES Digital). Available at http://www.dpi.inpe.br/prodesdigital/prodes.php. Accessed on January 20, 2011. Brazilian National Agency for Space Research.

Lambin E. F., Meyfroidt P. (2011) Global land use change, economic globalization, and the looming land scarcity. *Proceedings of the National Academy of Sciences*, **108**, 3465-3472.

Mandemaker M., Bakker M., Stoorvogel J. (2011) The role of governance in agricultural expansion and intensification: a global study of arable agriculture. *Ecology and Society*, **16**, 8.

Matson P. A., Vitousek P. M. (2006) Agricultural intensification: Will land spared from farming be land spared for nature? *Conservation Biology*, **20**, 709-710.

Morton D. C., DeFries R. S., Shimabukuro Y. E., Anderson L. O., Arai E., Espirito-Santo F. D., Freitas R., Morisette J. (2006) Cropland expansion changes deforestation dynamics in the southern Brazilian Amazon. *Proceedings of the National Academy of Sciences of the United States of America*, **103**, 14637-14641.

Morton D. C., DeFries R. S., Shimabukuro Y. E., Anderson L. O., Espirito-Santo F. D. B., Hansen M., Carroll M. (2005) Rapid assessment of annual deforestaion in the Brazilian Amazon using MODIS data. *Earth Interactions*, **9**.

Nepstad D., Soares B. S., Merry F., Lima A., Moutinho P., Carter J., Bowman M., Cattaneo A. *et al.* (2009) The end of deforestation in the Brazilian Amazon. *Science*, **326**, 1350-1351.

Nepstad D. C., Stickler C. M., Almeida O. T. (2006) Globalization of the Amazon soy and beef industries: Opportunities for conservation. *Conservation Biology*, **20**, 1595-1603.

Nepstad D. C., Stickler C. M., Soares B., Merry F. (2008) Interactions among Amazon land use, forests and climate: prospects for a near-term forest tipping point. *Philosophical Transactions of the Royal Society B-Biological Sciences*, **363**, 1737-1746.

Rudel T. K., Defries R., Asner G. P., Laurance W. F. (2009a) Changing drivers of deforestation and new opportunities for conservation. *Conservation Biology*, **23**, 1396-1405.

Rudel T. K., Schneider L., Uriarte M., Turner B. L., DeFries R., Lawrence D., Geoghegan J., Hecht S. *et al.* (2009b) Agricultural intensification and changes in cultivated areas, 1970-2005. *Proceedings of the National Academy of Sciences of the United States of America*, **106**, 20675-20680.

Rudorff B. F. T., Adami M., Aguiar D. A., Moreira M. A., Mello M. P., Fabiani L., Amaral D. F., Pires B. M. (2011) The soy moratorium in the Amazon biome monitored by remote sensing images. *Remote Sensing*, **3**, 185-202.

Silva J. H. (2009) Economic causes of deforestation in the Brazilian Amazon: An empirical analysis. Master's Thesis, Economics and Politics Program. University of Freiburg, Freiburg, Germany, 46 pp.

Stickler C. M. (2009) Defending public interests in private forests: land use policy alternatives for the Xingu River headwaters region of southeastern Amazônia. Ph.D., Geography. University of Florida, Gainesville, FL, 199 pp.

Stickler C. M., Almeida O. T. (2008) Harnessing international finance to manage the Amazon agro-industrial explosion? The case of International Finance Corporation loans to Grupo Maggi. *J of Sustainable Forestry*, **27**, 57-86.

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

Tilman D., Cassman K. G., Matson P. A., Naylor R., Polasky S. (2002) Agricultural sustainability and intensive production practices. *Nature,* **418**, 671-677.

UNFCCC (2010) The Cancun Agreements: Outcome of the work of the Ad Hoc Working Group on Long-term Cooperative Action under the Convention. In: *Decision 1/CP.16.* (ed Parties TCOT).

Chapter 3

In hot water: The influence of agricultural land management on headwater stream temperature in the southern Amazon

Abstract

Large-scale cattle and soybean production are the primary drivers of deforestation in the Amazon's agricultural frontier. These land cover and land use changes (LCLUC) can degrade stream ecosystems by reducing hydrologic connectivity, changing the amount of light and nutrient inputs, and altering the quality and quantity of water flowing within streams. This study integrates field data and satellite-derived information in the Xingu Basin, a rapidly changing agricultural landscape in Mato Grosso, Brazil, to assess how recent (2001-2010) agricultural expansion has affected the temperature of headwaters streams. We document the extent of LCLUC at the landscape scale, quantify how these changes influence stream temperature in 12 catchments, and evaluate how the presence of riparian buffers and impoundments influence stream temperature patterns. By 2010, over 40% of small catchments outside protected areas were dominated by agriculture (> 60% of area), with an estimated 10,000 impoundments (one per 7.6 km of stream) in the upper Xingu landscape. At the catchment scale, we monitored stream temperature in 12 soy, pasture, and forest watersheds and explored its relationship with land management (riparian forest buffers, watershed forest cover, impoundments) and environmental variables (precipitation, light, air temperature). Streams in pasture and soy watersheds were significantly warmer than those in forested watersheds, with average daily maxima more than 4^{o}C (16.5%) higher in pasture and 3^{o}C (12.1%) higher in soy. The density of impoundments upstream, percent forest in upstream riparian buffers (500-m upstream), and watershed forest cover were significant ($p<0.01$) predictors of stream temperature. Numerous

studies have demonstrated that such temperature increases can exert a negative influence on the distribution and abundance of aquatic organisms. Our results suggest that these impacts could be substantially mitigated through enforcement of existing legislation to protect riparian buffers and new regulations to limit the number of impoundments in emerging agricultural landscapes.

Introduction

The expansion of cattle ranching and soybeans has fundamentally changed the landscape of the southern Amazon by replacing native vegetation with pasture grasses and croplands. These land-cover and land-use changes (LCLUC) can have a number of consequences for freshwater ecosystems, including degrading riparian areas (Deegan *et al.*, 2011), altering hydrological cycles (Bruijnzeel, 2004, Coe *et al.*, 2011, Costa *et al.*, 2003, Hayhoe *et al.*, 2011), and decreasing hydrologic connectivity (Freeman *et al.*, 2007, Pringle, 2003). Some of these impacts are inevitable tradeoffs associated with agricultural development in the tropics, while others could be substantially mitigated through improved land management. Numerous studies have demonstrated the impact of agriculture on streams and explored the important mitigating function of riparian buffers (Benstead *et al.*, 2003, Gergel *et al.*, 2002, Lorion & Kennedy, 2009b, Uriarte *et al.*, 2011), although relatively few have been focused in the tropics. Most field studies in the Amazon have been limited to the scale of small watersheds (Neill *et al.*, 2006) and fail to consider the cumulative, landscape-scale effects of agricultural expansion.

This study integrates field-based data and satellite-derived information to explore the influence of recent agricultural expansion (2001-2010) on headwater streams in the Xingu Basin (Fig. 3.1), a rapidly changing agricultural landscape in Mato Grosso, Brazil. We focus on stream temperature because it is an important determinant of habitat quality (Buisson *et al.*, 2008, Eaton

& Scheller, 1996) and, in small streams, is directly influenced by management decisions at the farm level. Specifically, increased stream temperatures have been linked to deforestation (Caissie, 2006), failure to conserve riparian forest buffers (Lorion & Kennedy, 2009b), and thermal pollution from farm impoundments, or small dams (Webb & Nobilis, 2007).

In small headwater streams, roughly 82% of heat exchange occurs at the air/water surface, with the remainder occurring at the streambed/water interface (Fig. 3.2; Evans *et al.*, 1998). Diel variations are generally small in forested headwater streams, where riparian vegetation provides shade and shelter that maintains relatively cool and steady temperatures throughout the day. These streams are particularly vulnerable to agricultural land-use changes, which often reduce streamside vegetation, exposing streams to increased solar radiation (Caissie, 2006). The resulting temperature increases may be exacerbated by the fact that, compared to forests, pasture grasses and soybeans have a higher surface albedo (Loarie *et al.*, 2011a), lower leaf area and shallower rooting depth, which leads to reduced evapotranspiration (Bruijnzeel, 2004, Costa & Foley, 1997) and increased surface temperatures (Costa *et al.*, 2007). In Brazil's agricultural frontier, these changes have been associated with increased stream discharge (Coe *et al.*, 2009, Coe *et al.*, 2011, Hayhoe *et al.*, 2011) and air temperatures (Loarie *et al.*, 2011b), which in turn may influence diel and annual water temperature cycles.

Stream temperature patterns exert a strong influence on the evolution, distribution, and ecology of aquatic organisms in stream ecosystems (Ward, 1985, Ward & Stanford, 1982). This is due, in large part, to the fact that most aquatic organisms are strict ectotherms, meaning they lack the anatomical and physiological means to regulate their body temperature relative to the environment. As a result, each species has evolved to occupy a specific thermal niche within which it may function well, but outside of which it may fail to survive (Hochachka & Somero, 2002). Even when not lethal, higher water temperatures can alter basic life history parameters, including incubation and development time (Gillooly *et al.*, 2002, Gillooly *et al.*, 2008), growth rates (Neuheimer *et al.*, 2011), and the metabolism of organisms ranging from microbes to fish (Gillooly *et al.*, 2001, Gillooly *et al.*, 2002, Huston *et al.*, 2003). These metabolic changes come with increased energetic requirements, which may or may not be met in degraded streams. In addition to direct impacts on species metabolism and survival, increased water temperature can have indirect effects by facilitating the spread of invasive species and disease (Roth *et al.*, 2010), increasing the toxicity of environmental contaminants (Rehwoldt *et al.*, 1972), and constraining the abundance and spatial distribution of species (Caissie, 2006, Vannote *et al.*, 1980).

Studies on the effects of stream warming on fish in temperate systems indicate that temperature thresholds are critical determinants of suitable fish habitat and that the influence of human activity on stream thermal regimes has had a strong negative influence on the quality and quantity of available habitat (Brett, 1956, Franco & Budy, 2005, Malcolm *et al.*, 2004, Myrick & Cech, 2004, Theurer *et al.*, 1985). Even modest increases in stream temperature can cause dramatic declines in salmonids, for example. Likewise, macroinvertebrate abundance has been projected to decline by 21% for every 1°C rise in water temperature (Durance & Ormerod, 2007, Kaushal *et al.*, 2010). These thresholds have not been well explored in tropical streams, but there is some evidence that they are governed by similar mechanisms. In Costa Rica, riparian forest removal and associated increases in stream temperature have been shown to alter the taxonomic composition of benthic macroinvertebrates, reduce diversity, and eliminate the most sensitive taxa (Lorion & Kennedy, 2009a). Similarly, the removal of riparian buffers in pasture areas increased temperature, reduced allochthonous inputs, and altered fish community composition compared to streams with intact buffers (Lorion & Kennedy, 2009b). A recent study in the

Xingu Basin suggests that the diversity of *Odonata* also decreases with increasing temperature (Batista, 2010).

The effects of elevated temperature may be further exacerbated by increases in diel temperature variability, additional warming due to future climate change, and stream fragmentation by dams. Impoundments act as physical barriers that alter the flow of water, sediments, and organisms within headwater streams. Although the majority of the literature on dams focuses on large hydroelectric dams, several studies indicate that small dams can have a large cumulative impact on stream ecosystems. Small, surface-release dams alter physical habitat by increasing water temperature (Cumming, 2004); changing current velocity, water volume, and depth above and below reservoirs (Alexandre & Almeida, 2010, Lehner *et al.*, 2011); and trapping fine sediments as a result of the slackwater created behind reservoirs (Walter & Merritts, 2008). When coupled with agricultural land uses, which often increase the supply of sediments and pollutants to streams, small impoundments have the potential to fundamentally alter the geomorphology and quality of habitats within stream networks.

Despite the importance of temperature in structuring stream ecosystems and the increasing pace of anthropogenic changes in many tropical regions, few studies exist on the influence of LCLUC on the temperature regimes of tropical streams. The vast majority of information on stream temperatures is widely scattered in the literature, having been collected as routine background information during site-specific ecological studies. Rarely is temperature the focus of study and, as a result, temperature measurements are usually not collected at sufficient temporal or spatial resolution to provide insight into diel and annual patterns, nor how human activity may alter these patterns. Here we combine satellite-based observations of LCLUC and management (i.e., forest cover, riparian buffers and impoundments) with field-based data collected over 16 months in twelve headwater streams draining pasture, forest, and soybean watersheds. This integrated approach allows us to explore the influence of agricultural land management on stream thermal regimes, both at the catchment and landscape scales. The study addresses four central questions:

(1) What is the spatial distribution of soybeans and cattle ranching within the Xingu Basin and how has this changed over the last decade?

(2) How is land use history associated with the distribution of farm impoundments in the landscape?

(3) What is the relationship between land management (i.e., forest cover, riparian buffers, and impoundments) and stream temperature at the catchment scale?

(4) How might current management strategies be modified to mitigate the impacts of agricultural expansion on headwater streams?

Methods

Study area and general approach

The Xingu River drains the Brazilian Shield, an ancient upland region where erosion processes occurring over millennia have left little unconsolidated material to wash into streams (Goulding *et al.*, 2003). For this reason, the Xingu is considered a clear water river system, characterized by relatively small amounts of suspended sediments where watershed forest cover is maintained. The headwaters of the Xingu (Fig. 3.1) occur on the Mato Grosso plateau (~ 600 m above sea level), along the transition between the Cerrado and Amazon biomes. The plateau is characterized by low topographic relief, sloping gently from the southern headwaters towards the

Amazon River in the north. Upland areas in the southern part of the study region are dominated by cerrado vegetation, a mixed savannah and woodland ecosystem, with gallery forests occurring in the wetter areas along stream corridors. The remainder of the study area is dominated by tropical forests more typical of the Amazon biome. The rainfall gradient is consistent with this vegetation transition, with lower rainfall in the south and higher rainfall in the north. Average annual precipitation in the region ranges from 1500 to 2400 mm and is highly seasonal, with a pronounced dry season from May to August, a pronounced rainy season from November to February, and intermediate levels of rainfall in the interim months (Hijmans *et al.*, 2005). Soils in the region are dominated by oxisols (ferralitic soils) and have good structure for cultivation.

We sampled twelve headwater streams within the forest biome to the east of the Xingu Indigenous Park (Fig. 3.1). The watershed for each sample location was predominantly in a single land cover type, with some variation in the following landscape variables: riparian forest cover, number of impoundments upstream, and percent forest cover in the watershed (Table 3.1). Of these, there were four reference streams, with watershed forest cover ranging from 95 to 100%; three streams with watershed pasture grass cover ranging from 96 to 100%; three streams with watershed soybean cover ranging from 82 to 94%; and two additional streams with soybean cover ranging from 53 to 57%. To the extent possible, sites were selected to encompass the gradient of land management conditions (i.e., riparian forest cover, watershed forest cover, and impoundments). All soybean watersheds in this study were converted from pasture at some point in the last 10-15 years, a land use history that is typical of soybean areas in the region. Because sampling took place on private properties in remote areas, the selection of final sampling locations was constrained by the accessibility of candidate streams. Data collected at each sampling location was linked to landscape variables (Table 3.1) derived from satellite analyses, allowing us to scale up the analysis from the catchment to the landscape scale.

LCLUC analysis

We characterized LCLUC in the upper Xingu Basin by combining data from several satellite sensors with field-based inventories. Land use history (2001-2010) was derived from the Moderate Resolution Imaging Spectroradiometer (MODIS) enhanced vegetation index time series (MOD13Q1), which was used to perform annual land-use/cover classifications (250-m resolution), based on differences in vegetation phenology. The high temporal resolution of the MODIS time series offers advantages over sensors with higher spatial resolution, such as Landsat TM (30-m), because it allows for greater cloud-free coverage during the rainy season, when agricultural land uses (e.g., pasture and cropland) are spectrally distinct. The methods used in producing this dataset have been described in previous studies (Macedo *et al.*, 2012, Morton *et al.*, 2006) and resulted in annual land-use/cover maps of soybean, forest, and pasture/cerrado classes.

We used existing data on the extent of cerrado in 2002 (Sano *et al.*, 2007) and cerrado deforestation from 2003 to 2010 (Ferreira *et al.*, 2007) to improve on this analysis by separating the pasture and cerrado classes. First we developed a consistent time series of cerrado extent by subtracting deforestation polygons for each year from remaining cerrado vegetation in the previous year. We used this time series to separate the original pasture/cerrado class into its component land covers (pasture grass and cerrado) for 2001-2010 (Fig. B.1). This correction affected about 20% of the study area and enabled a more accurate representation of agricultural land uses within the Cerrado biome. We used the final classification to determine the proportion of soy, pasture and native vegetation (forest or cerrado) for each microbasin in the upper Xingu (as defined by the Brazilian Water Agency; ANA, 2010), as well as changes in these proportions from 2001 to 2010.

Impoundment mapping

The distribution of farm impoundments in the Xingu Basin was mapped using a high resolution (15-m) image mosaic acquired by the Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER). The mosaic was constructed from 89 image tiles, most of which were acquired during the 2007 dry season. In the few cases where 2007 images were unavailable, we used the next best image available from the Land Processes Distributed Active Archive Center (LPDAAC). Once the image mosaic was assembled, we calculated the normalized difference vegetation index (NDVI) for the study area and used an object-oriented approach to extract spectrally homogeneous and spatially distinct "image objects" or segments (Walker *et al.*, 2010). This segmentation and subsequent extraction of object-level attributes was performed using the Definiens eCognition software package. A total of 401 reference points were used for the calibration and validation of the classification models. Of these, 85 points were GPS locations of impoundments collected on the ground. These field points were supplemented by 26 additional impoundments identified by visual interpretation of the ASTER imagery. In order to get a sample of "non-impoundment" points, we randomly generated 300 additional points and super-imposed them on the ASTER image mosaic. Based on visual interpretation, nine of these points were discarded because they occurred in impoundments by chance.

The classification of impoundment and non-impoundment segments was implemented using the randomForest algorithm in the R statistical programming environment (Breiman, 2001). RandomForest uses the training data (segments) to construct an ensemble ("forest") of independent decision trees using a bootstrap sample of the data in a process called *bagging* (derived from *b*ootstrap *agg*regation). About one third of the reference cases are left out of each bootstrap sample. These "out-of-bag" (OOB) samples are predicted during each iteration and subsequently aggregated to produce an OOB error estimate. Previous work indicates that OOB error rates are robust when compared to error rates calculated using independent validation data (Walker *et al.*, 2007a).

Because impoundments have similar spectral and morphological attributes to other objects in the landscape (e.g., small lakes or cloud shadows), the final classification was masked to exclude areas outside a 500-m buffer from the stream network (SEMA, 2010) and areas inside forested protected areas, which are unlikely to contain impoundments. This eliminated the majority of commission errors. As a final step, the entire image mosaic was visually inspected at a scale of 1:50,000. Based on visual interpretation of the surrounding land-use/cover classes and proximity to rivers and roads, additional objects that were misidentified as impoundments were eliminated. About 900 impoundments that were omitted in the original image classification (~9,000 impoundments) were digitized and added to the dataset.

As a final step in the analysis, we estimated the prevalence of impoundments in the landscape and assessed how their distribution was influenced by land-use history. First, we combined our impoundment map with the stream vector layer from Mato Grosso's state environmental agency (SEMA, 2010). After calculating stream length in ArcGIS 10, we divided the stream length outside protected areas by the number of impoundments, yielding an estimate of the proportion of the stream network affected by small dams. Finally, we combined the impoundment map with the MODIS-based land-use/cover time series to estimate the concentration of impoundments in each of the following land-use history classes: forest; cerrado; natural vegetation to pasture; natural vegetation to soy cropland; and natural vegetation to pasture to soy cropland. The final result was normalized by the area (km^2) in each class.

Catchment-scale classification

We derived watershed boundaries from vegetation-corrected Shuttle Radar Topography Mission (SRTM) data using previously published methods (Hayhoe *et al.*, 2011). Briefly, the raw SRTM image mosaic was corrected using a Landsat-based vegetation classification to remove bias due to vegetation height. Given the low relief of the study area, forested areas adjacent to agricultural land lead to errors in flow direction and, thus, watershed delineation (Hayhoe *et al.*, 2011, Kellndorfer *et al.*, 2004, Walker *et al.*, 2007b). Once this bias was removed, we derived stream basins from the SRTM using the standard Hydrology Tools in ArcGIS 10.0. We determined flow direction and flow accumulation for each SRTM pixel, used these to define the stream channel, and delineated the watershed for each stream monitoring point by identifying all upstream pixels draining to that point. In two cases, the drainage basin had such low relief that an automated delineation was not possible. These were hand digitized based on the flow direction grid to correct for errors identified in the automated delineation.

For the 12 study catchments, we used a 2009 Landsat 5 (LS) image mosaic (Appendix B – Supplemental Text) to create a finer-scale (30-m) analysis of riparian cover and agricultural cover in the watershed. This would not have been possible with the MODIS-based classification because of its coarse scale (250-m), as well as its limited reliability in classifying edge pixels (i.e., riparian buffers) and areas smaller than 25 ha (Morton *et al.*, 2005). The Landsat-based classification consisted of four classes – agriculture, forest, water, and wetland. First, we calculated the normalized difference vegetation index (NDVI) and Tasseled Cap (TC) transformations, resulting in a total of 9 bands (LS 1-5, NDVI, TC1-3). We extracted pixel values in all bands for each of 500 training points. Of these, 302 were collected in the field during the summer of 2010 (234 in pasture or soy, 68 in forest). In order to improve the classification, we selected 87 additional water and 25 additional wetland points based on on-screen visual interpretation of the image mosaic. Classification of these training pixels was implemented using randomForest, as described above. Finally, we used the randomForest ensemble of models to predict the land use/cover in each pixel of the image mosaic. Once completed, this classification was used to summarize percent agriculture in each watershed, as well as the percent forest cover in riparian buffers of varying length: 30-m surrounding the sample point, 100-m upstream of the point, and 500-m upstream of the point. All stream buffers were defined as being 30-m wide, in keeping with current requirements under the Brazilian Forest Code (Stickler, 2009).

Catchment-scale field sampling

At each sampling site, we deployed two Onset HOBO Pendant Temperature and Light data loggers to measure water temperature and light (lux) every 30 minutes. Light was measured using a data logger attached to a flotation device to maintain the logger at the water surface, while stream temperature was measured 25 cm below the surface to minimize the influence of direct sunlight on logged temperature measurements. The light loggers measure illuminance and estimate the relative amount of light reaching the stream surface at each site. Hemispheric (fisheye) photographs were also used to characterize the riparian forest cover (percent light transmitted) at each sampling site. Each sample point was paired with one of nine weather stations that logged precipitation events and hourly air temperature. The present analysis focuses on sixteen months of data collected by the data loggers at each site. In order to verify that the temperature patterns observed in this study are representative of a typical year, we also examined a longer time series, consisting of hourly temperature data collected in a subset of the study area from 2007 to 2010. This dataset was collected as part of a study on discharge (Hayhoe *et al.*, 2011) in 9 first order streams (6 soy and 3 forest watersheds).

In order to directly test the effects of impoundments on stream temperature we placed pairs of synchronized data loggers above and below six impoundments in the study area during the month of July, 2010. The temperatures above (upstream) and below (downstream) the impoundments were compared using an analysis of variance. We then selected a single impoundment (~0.63 km long) to examine the rate of temperature recovery downstream. To do this, we created a transect using a total of 11 temperature loggers over the course of 6 weeks (September to November, 2010). One logger was placed above the impoundment (the baseline); two loggers were placed at the two outlets of the impoundment and averaged to yield the mean temperature at 0-m below the impoundment; 7 loggers were placed from 0-m to 1050-m at 150-m intervals; and the last logger was placed near a bridge approximately 2350-m downstream. These field observations were then combined with the stream layer and the ASTER-based impoundment maps to estimate the proportion of the Xingu stream network with warmer water temperatures as a result of impoundments.

Modeling temperature at the catchment scale

In addition to land use/cover, we selected the following variables as potential predictors of stream temperature: precipitation, percent light transmitted at the sample location (derived from hemispheric photographs), light at the stream surface (lux), air temperature, the number and density (per stream km) of upstream impoundments, distance to the nearest impoundment, percent forest cover in riparian buffers, and percent forest cover in the watershed. Because extreme temperatures and diel variability are more likely to limit aquatic organisms than mean temperature, we used the upper quantile (75% probability) daytime temperature as the dependent variable. Light and rainfall data were log-transformed and all covariates were standardized to facilitate interpretability across variables (Gelman & Hill, 2007, Schielzeth, 2010).

As a first step, we conducted exploratory analyses using ordinary least square (OLS) regression models to examine the relative importance of each variable. The OLS regressions were used as a basis for choosing the lag (temporal or spatial) for each variable that optimized its ability to predict water temperature. For air temperature, we considered both the daily and upper quantile mean air temperatures with lags from 0 to 2 days. For rainfall, we considered the total daily rainfall with lags from 0 to 2 days, as well as the total weekly rainfall at 1 and 2-week lags. Finally, we evaluated several nested spatial arrangements for riparian buffers of 30-m width (30-m, 100-m, and 500-m long buffers upstream of the sampling location). Comparison of regression results, based on R^2 and the Akaike Information Criterion (AIC), indicated that (log) weekly rainfall with a two-week lag; mean daily air temperature with no lag; and a 500-m long upstream riparian buffer were the best predictors of stream temperature.

Once the full set of independent variables was assembled, we eliminated collinear variables based on examination of the correlation matrix and variable inflation factors (VIF < 4, Zuur *et al.*, 2009). Notably, percent forest cover in the watershed was highly collinear with impoundment density and was excluded from the models. Instead, we elected to focus on riparian forest buffers and impoundment density, which are more readily managed in agricultural landscapes. We used the final set of independent variables to develop linear mixed models in order to understand the relative importance of different land management variables in predicting observed stream temperature (Table 3.2). Our model took the form:

$$\mathbf{Y_T = \beta_0 + \beta_1 x_{rf} + \beta_2 x_{at} + \beta_3 x_{rf} * x_{at} + \beta_4 x_{imp} + \beta_5 x_{lt} + \beta_6 x_{ppt} + \mathcal{E}}$$

where Y_T is the predicted upper quantile daytime temperature; x_{rf} is percent forest in the riparian buffer 500-m upstream; x_{at} is the air temperature; x_{lt} is the (log) light at the stream surface; x_{ppt} is the (log) weekly rainfall with a two-week lag; x_{imp} is the density of impoundments upstream of the sampling location; and $\mathcal{E}$ is a random effect term including the month of the year and sampling location. The term $x_{rf}*x_{at}$ represents the interaction between riparian forest cover and air temperature as predictors of stream temperature. This interaction was included in the model because of the known relationship between forest cover and land surface temperature. As forest cover decreases we would expect an increase in surface temperature in the watershed, which directly affects air temperature (Loarie *et al.*, 2011b). Because of this interdependency, these two factors in the model are likely to interact. All models included the month of year and sampling location as a random effect in order to account for seasonality and repeated measures at each site (Gelman & Hill, 2007, Pinheiro & Bates, 2000). Including location of the sample sites as a random effect reduces some of the noise created by random variation at each individual site, without overfitting the model. Models were evaluated using the corrected AIC (AICc; Burnham & Anderson, 2002).

Modeling temperature at the landscape scale

We used the catchment-level temperature model to scale up our understanding of the effect of impoundments and riparian forests to the entire Xingu landscape. Based on the impoundment map and stream layer, we calculated the impoundment density for each microbasin (ANA, 2010) in the Xingu Basin. Gridded climatological datasets for air temperature (Fan & van den Dool, 2008) and rainfall (Huffman *et al.*, 2007) were used to calculate the long-term mean values of these environmental variables for each microbasin. We estimated headwater stream temperature for the following three scenarios of riparian buffer conservation: 100%, 50% and 0% conservation of riparian forest buffers within each microbasin. For each scenario, we estimated the average (log) light reaching the stream surface based on field measurements from the sample locations that most closely approximated those scenarios. Microbasins within protected areas were assumed to have 100% riparian forest conservation in all three scenarios. The final results for each scenario were expressed as the deviation from a hypothetical reference scenario that assumed 100% riparian buffer conservation and no impoundments for all microbasins.

Results

The overall accuracies of the MODIS, Landsat 5, and ASTER classifications were 92%, 97%, and 99%, respectively (Table B.1). More specifically, the user's accuracy for the Landsat-based riparian forest class was 96% and that of the ASTER-based impoundment class was 98%. As previously reported (Macedo *et al.*, 2012), the user's accuracies for the MODIS-based forest, pasture, and cropland classes were 94%, 94%, and 89%, respectively.

Landscape-scale trends in land use/cover

MODIS-based analyses of land use/cover in the upper Xingu indicate that the number of catchments dominated by agricultural land uses doubled during the last decade. Whereas in

2000, 14% of watersheds outside protected areas had more than 50% of their area occupied by agriculture (i.e., pasture and soybeans), by 2010 this number had increased to 28%. In 2001, the average catchment had 25% of its area occupied by agriculture, nearly all of it in pasture. By 2010, the average catchment had 40% of its area occupied by agriculture, with over 15% in soybeans and the remainder in pasture (Fig B.2). The areas of most rapid soybean expansion were the watersheds close to the BR-163 highway in the western Xingu and the municipality of Querência, to the east of the Xingu Indigenous Park (Fig. 3.3b). Cattle ranching expanded throughout the upper Xingu and came to dominate many of the watersheds south and east of the park by 2010 (Fig. 3.3a).

Based on our ASTER classification, we estimated that there were approximately 10,000 impoundments in the upper Xingu Basin as of 2007 (Fig. 3.4). With few exceptions, all of these occurred in headwater streams outside of protected areas. On average, there was one impoundment per 7.4 km of stream outside of protected areas. In 2007, the concentration of impoundments in pasture areas (0.14/km^2) was twice as high as that in soybeans (0.06/km^2), while the concentration in cerrado (0.05/km^2) and forest (0.02/km^2) areas was lower than that in either agricultural land use (Fig. 3.5). Areas that were converted from pasture to soybean production had a higher concentration of impoundments than those that were converted directly from native vegetation to soybeans.

Catchment-scale trends in stream temperature

The landscape-scale changes described above have direct implications for water temperature at the catchment scale. In general, streams in reference (forested) watersheds were cooler and less variable than those in watersheds dominated by pasture and soybeans (Fig. 3.6a).

The mean daily maximum temperature in forested streams (25.8^{o}C) was more than 4^{o}C cooler than pasture (30.1^{o}C) streams and 3^{o}C cooler than soy (29.0^{o}C) streams (Fig. 3.6b). This difference is even more pronounced if we consider the upper quantile, with forested streams 5.7^{o}C and 3.7^{o}C cooler than pasture and soy streams, respectively. These patterns suggest that small streams in forested watersheds are relatively buffered against extreme temperatures when compared to those in agricultural watersheds. This buffering capacity is illustrated by the fact that stream temperatures in pasture watersheds frequently exceeded 30^{o}C during the hottest part of the day, reaching a daily maximum of 35^{o}C - 36^{o}C on several occasions, whereas forested streams had an absolute maximum of 27^{o}C during the entire time series. A comparison of long-term data in soybean and forest watersheds indicated that stream temperature was highly seasonal and follows a sinusoidal pattern, with higher temperatures during the peak of the rainy season (southern hemisphere summer) and lower temperatures during the dry season. This pattern was consistent with seasonal variations in air temperature, and differences among forested and soybean watersheds were consistent across years (Fig. 3.7).

Stream temperature was significantly correlated with the amount of light reaching the stream surface ($p<0.001$), which explained approximately 30% of the observed variation in daytime temperature (R^2 = 0.28). The mean amount of light reaching the stream surface was positively correlated with canopy openness and negatively correlated with percent forest cover in a 30-m buffer around the sample point. Of the three riparian buffer lengths considered (30-m, 100-m, and 500-m), the 500-m buffer had substantially more explanatory power than the other candidates, indicating a lag in the recovery of stream temperatures downstream of a clearing. The mean upper quantile temperature was negatively correlated with percent forest cover (Fig. 3.8a) upstream ($p<0.01$, $R^2 = 0.65$), suggesting that maintenance of riparian areas is a key factor in maintaining stream temperature.

Water temperature was positively correlated with the density of impoundments (Fig. 3.8b) upstream of the sampling locations ($p<0.01$, $R^2 = 0.64$). Direct measurement of temperatures above and below six impoundments indicated that these water bodies have a pronounced warming affect. Both mean and upper quantile temperatures downstream of the impoundments were significantly warmer than temperatures upstream (Fig. 3.9a; ANOVA, $p<0.001$), with a mean temperature increase (ΔT) of 1.7 ^{o}C. Results from one transect below an impoundment suggest a linear pattern of recovery downstream, as riparian shading and groundwater inputs bring the temperature back to equilibrium. The temperature had not completely recovered to the baseline level 2.4 km downstream of the impoundment (Fig. 3.9b).

Catchment-scale temperature model

Although soybean and pasture catchments were both associated with warmer stream temperatures, they exhibited different patterns in management-related covariates such as riparian forest cover and impoundment density (Table 3.1). We developed linear mixed models (Table 3.2) to understand the relative importance of each of these predictors of stream temperature, while controlling for environmental variables (rainfall, air temperature, and month of year). Comparison of parameter estimates (Fig. B.3) indicates that upstream riparian cover and impoundment density were the most important predictors of stream temperature, followed by air temperature. Riparian forest cover showed a significant interaction with air temperature in predicting stream temperature ($p<0.001$), with the former having a strong cooling effect and the latter a strong warming effect. The interaction parameter indicated that (at average stream temperatures) each additional unit of upstream riparian forest cover resulted in a 17.5% reduction in the potential warming effect of air temperature, supporting the notion that riparian forests serve an important buffering function.

To estimate the proportion of the stream network altered by impoundment-induced warming, we combined field-based measurements of downstream temperature recovery with our satellite-based map of impoundments. Based on the downstream temperature transect, we estimated a cooling rate of 0.63 ^{o}C per km. Given this cooling rate and the mean ΔT after impoundments, the average recovery distance would be 2.74 km downstream of the outlet. Assuming that these measured relationships were representative of the average impoundment in the Basin, we estimate that 27,380 km (37%) of the stream network outside protected areas were potentially under the thermal influence of impoundments in 2007. Including the stream length occupied by the impoundments themselves (mean ~0.6 km) would increase the estimate to 45% of the stream network.

Landscape-scale temperature model

As riparian buffer conservation decreased, the predicted change in stream temperature (ΔT) increased, and the relative importance of impoundments as a driver of warming decreased (Fig. 3.10). Under the conservation scenario (100% RF), the mean ΔT was 0.23^{o}C, 5% of microbasins had a $\Delta T > 1^{o}$C, and 0.5% of microbasins had a $\Delta T > 2^{o}$C. Under the scenario with 50% RF conservation, the mean ΔT was 1.17 ^{o}C and 9% of microbasins had a $\Delta T > 2^{o}$C. Under the scenario with no RF conservation, the mean ΔT was 2.26 ^{o}C and 75% of microbasins – or all microbasins outside of protected areas – had a $\Delta T > 2^{o}$C.

Discussion and Conclusions

Global demand for agricultural products is expected to increase by as much as 70% by 2050 (Bruinsma, 2009) and tropical regions are the only remaining areas with land available to meet these demands (DeFries & Rosenzweig, 2010). Although socially and economically important, the expansion of industrial agriculture in the tropics involves a number of potential tradeoffs, including the fragmentation and degradation of freshwater ecosystems and associated changes in stream temperature. Meeting the food demands of a growing population while minimizing the negative impacts of agricultural expansion will be one of the greatest challenges of the coming decades. Achieving it will require thoughtful management of agricultural lands at the landscape scale and the development of region-specific mitigation strategies based on solid science. This study documents the extent to which agricultural expansion has already impacted headwater streams in the southern Amazon and expands our understanding of how to manage these impacts in this and other agricultural frontiers.

Despite environmental legislation to protect forests and riparian areas on private lands (Azevedo, 2009, Stickler, 2009), soybeans and cattle ranching are expanding and intensifying rapidly in upland areas in the Xingu Basin. Within the last decade, small watersheds outside of protected areas have seen a steady decline in the proportion of upland forest cover and an increase in the proportion of agricultural land uses (Fig. 3.3). In addition to decreased forest cover, cattle pastures are strongly associated with the installation of small farm impoundments to provide drinking water for cattle. Previous research suggests that cattle ranching is also associated with degradation of riparian areas, due to the direct effects of grazing and the encroachment of pasture grasses into the stream channel (Deegan *et al.*, 2011). Our analysis suggests that the fingerprint of this pasture legacy may be evident even after conversion to soybeans. Furthermore, although impoundments are associated with the legacy of cattle ranching in the region, it appears that many more are being installed as a result of a growing network of private and public roads in the landscape. This explains the presence of impoundments in areas classified as cerrado and forest, as well as those converted directly from natural vegetation to soybeans. Due to limited data on informal roads in the region, we were unable to evaluate the relative importance of roads for the proliferation of impoundments in the landscape. This is an important area for future research and will complement the results presented here.

At the catchment scale, our results indicate that land management can play an important role in reducing land use-related increases in stream temperature. As expected, the amount of forest cover in upstream riparian buffers is a key predictor of stream temperature, suggesting that existing legislation to conserve and restore riparian buffers is appropriate. In this regard, temperature can serve as a simple and relatively inexpensive measure of the long-term impact of these efforts on the ground. The presence of instream impoundments also had a measurable impact on downstream temperature and, given the density of impoundments in the landscape, is likely an important factor in the overall integrity of the stream network. Although our sample locations captured a wide range in the variation of riparian forest cover, they were limited to "end member" watersheds that were almost exclusively in one land use. Future research efforts could expand on this work by including a gradient of land-use/cover configurations that would lend more insight into the thresholds beyond which declines in stream integrity become evident.

Our results confirm the importance of riparian buffers for mitigating the thermal impacts of land use and thereby buffering streams against land-use related degradation. Currently, the Brazilian Forest Code requires the maintenance of 30-m buffers around small streams and even wider buffers around springs and impoundments. Historically, compliance with these requirements was low due to a mixture of unclear land tenure, lack of capacity for monitoring and enforcement, and poor dissemination of the requirements to property managers and owners. In recent years the state and federal governments have made much progress in clarifying land tenure, educating landowners, and improving enforcement through satellite-based monitoring and environmental licensing (Azevedo, 2009). Extension activities through non-profit organizations and government programs have raised awareness and built capacity for the restoration of riparian buffers in the region, although restoration is often challenging, particularly in areas with a pasture legacy. Recently proposed changes to Brazilian environmental legislation (Tollefson, 2011) threaten to weaken these efforts by reducing riparian buffer requirements and delegating enforcement activities to the state or municipal level, where there may be little capacity or political will to do so.

Our results also highlight impoundments as a pervasive and previously undocumented threat to the upper Xingu network. These water bodies are installed in an *ad hoc* manner in the landscape, primarily as a result of conversion for pasture and expansion of the road network. At least one impoundment is present in nearly all first and second order streams in agricultural watersheds and each is associated with measurable increases in stream temperature. Our landscape-scale analysis suggests that these impoundments may have a large cumulative impact on headwater streams, fundamentally altering the thermal regimes, hydrology, and connectivity of the stream network. The Brazilian government’s decision to proceed with installation of the Belo Monte hydroelectric dam in the Xingu Basin has received a great deal of national and international media attention, particularly for its impacts on indigenous communities upstream. While the focus on large hydroelectric dams is warranted, this study suggests that small farm impoundments upstream of the Xingu Indigenous Park may already be exerting a profound impact on the temperature regime and, by extension, stream metabolism, connectivity, and overall function. Such changes have the potential to negatively impact water quality, as well as fisheries that are important for reserve residents. Mitigating these impacts, as well as those of future agricultural expansion and intensification in the region, would require new regulations that limit the number of new farm impoundments in emerging agricultural landscapes.

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Table 3.1: Landscape attributes of each sample stream.

Watershed	IMP (#)	IMP (per km)	% RF (500-m)	Area (km^2)	% FOR
Forest 1	0	0	100	9.8	97
Forest 2	0	0	100	14.6	96
Forest 3	0	0	98	17.7	95
Forest 4	0	0	100	18.1	100
Pasture 1	4	0.75	100	15	04
Pasture 2	3	0.47	29	22.8	03
Pasture 3	1	0.93	35	4.6	00
Soy 1	5	0.7	70	20.9	18
Soy 2	1	0.37	88	10	43
Soy 3	1	0.45	00	16.2	47
Soy 4	1	0.38	98	9.6	06
Soy 5	1	0.48	100	5.5	08

IMP (#), number of impoundments; IMP (per km), number of impoundments per km of stream length upstream; RF, % riparian forest cover (500-m upstream); Area, watershed area; FOR, % forest cover in the watershed.

Table 3.2: Model comparison for daily stream temperature using the corrected Akaike Information Criterion (AICC). Also reported are the degrees of freedom and the AICc weight for each model.

Model	df	ΔAICc	Weight
RF + LT + PPT + AT + IMP + RF*AT	11	0	0.950
RF + LT + PPT + AT + RF*AT	10	6	0.050
RF + PPT + AT + IMP + RF*AT	10	61	< 0.001
RF + LT + PPT + AT + IMP	10	66	< 0.001
RF + LT + AT + IMP + RF*AT	10	2187	< 0.001

RF, percent riparian forest cover (500-m upstream); LT, log of light at stream surface; PPT, log of precipitation (2-week lag); AT, air temperature; IMP, density of impoundments (number/km of stream).

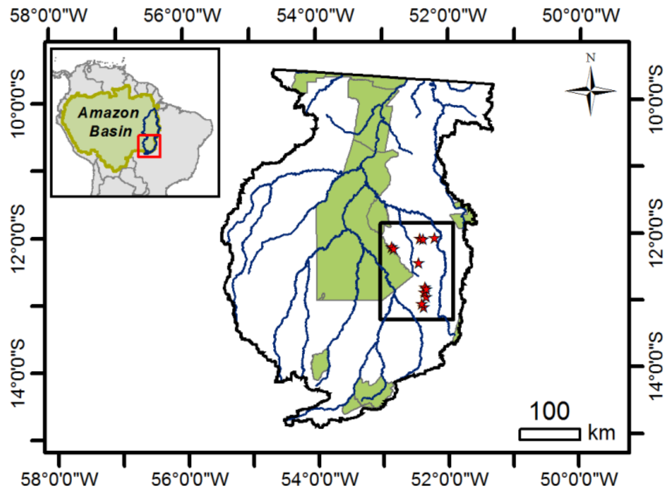

Figure 3.1: Map of the study area, with major rivers of the upper Xingu Basin. Stars indicate the long-term sampling locations (black box). The upper Xingu Basin is located in the southeastern Amazon Basin (*Upper left*).

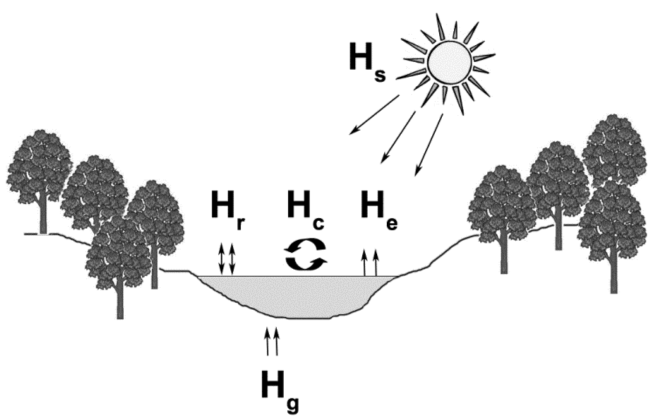

Figure 3.2: River heat exchange processes. Energy exchange at the air/water interface occurs as a result of: (i) solar (net shortwave) radiation (Hs); (ii) net long-wave radiation (Hr); (iii) latent heat flux due to evaporation (He); and (iv) sensible heat flux (Hc) due to conduction and convection as a result of river-atmosphere temperature differences. Adapted from Caissie, 2006.

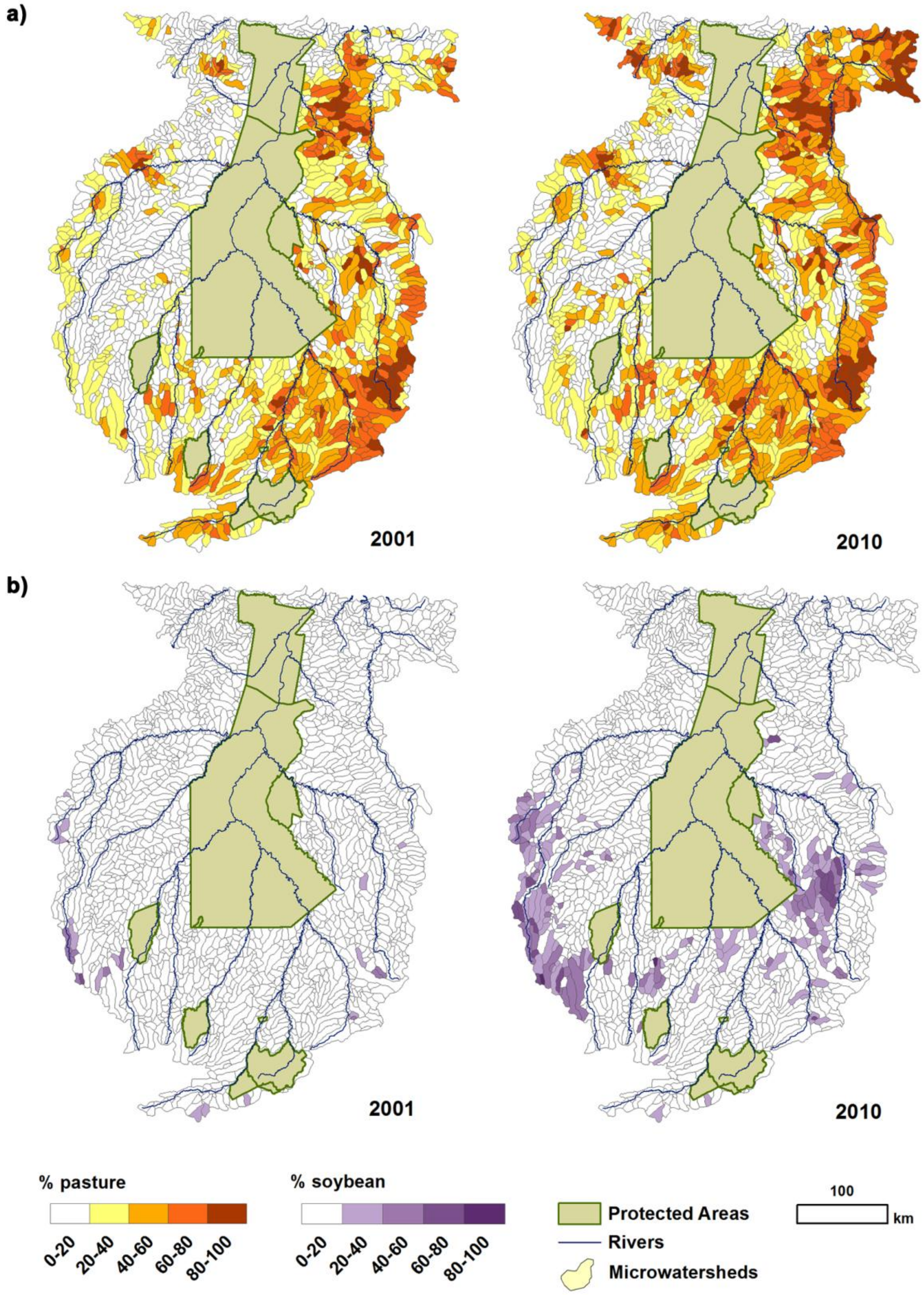

Figure 3.3: Proportion of Xingu microbasins occupied by cattle ranching (top) and soybeans (bottom) from 2001 to 2010. Watershed boundaries were acquired from the Brazilian Water Agency (ANA, 2010) and land use from our MODIS-based classification.

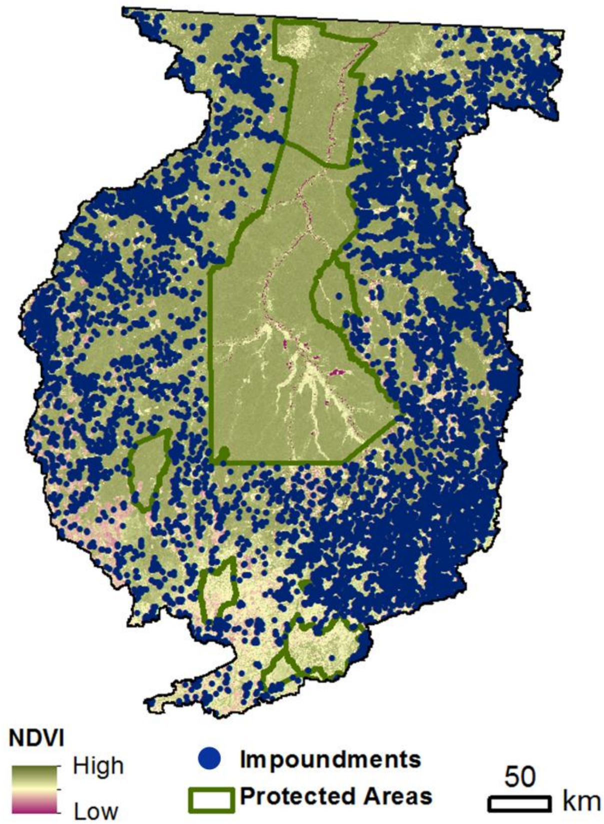

Figure 3.4: Impoundments in the upper Xingu Basin. The map is based on classification of an ASTER image mosaic and indicates the presence of nearly 10,000 small farm impoundments in the upper watershed as of 2007.

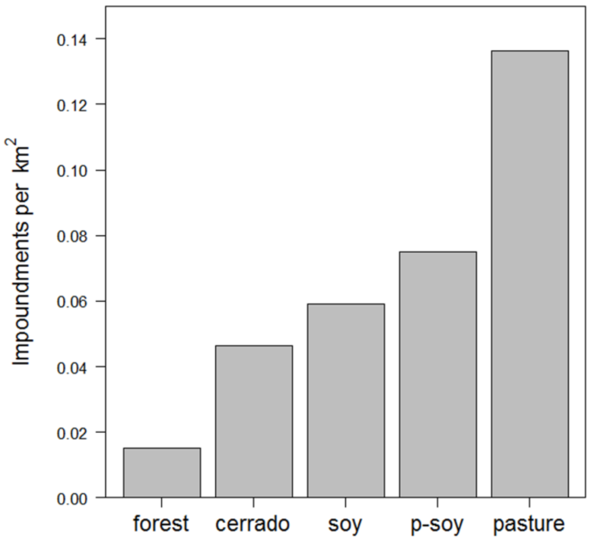

Figure 3.5: Distribution of impoundments in each land-use history, normalized by area. Pasture and soy areas were converted directly from native vegetation (forest or cerrado) for that land use. Areas converted from native vegetation to pasture and subsequently to soybeans are designated as p-soy. The distribution of impoundments was mapped using an ASTER image mosaic, whereas land-use history (2001-2010) was derived from MODIS-based analyses (Ferreira *et al.*, 2007, Macedo *et al.*, 2012).

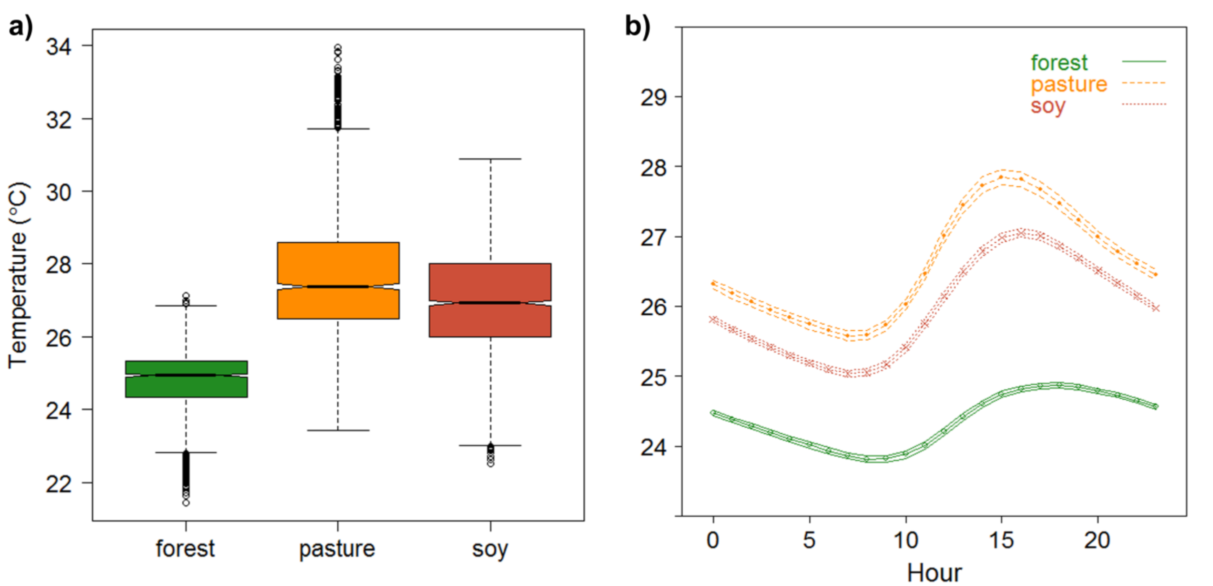

Figure 3.6: Relationship between land use and stream temperature. (a) Summary of daytime stream temperature (upper quantile) by land use. Land use is a significant predictor of stream temperature ($p < 0.001$, $R^2 = 0.38$). (b) Mean diel temperature patterns, with bootstrapped non-parametric confidence intervals. The x-axis represents the hour of the day (0-24) and the y-axis the mean hourly temperature (˚C).

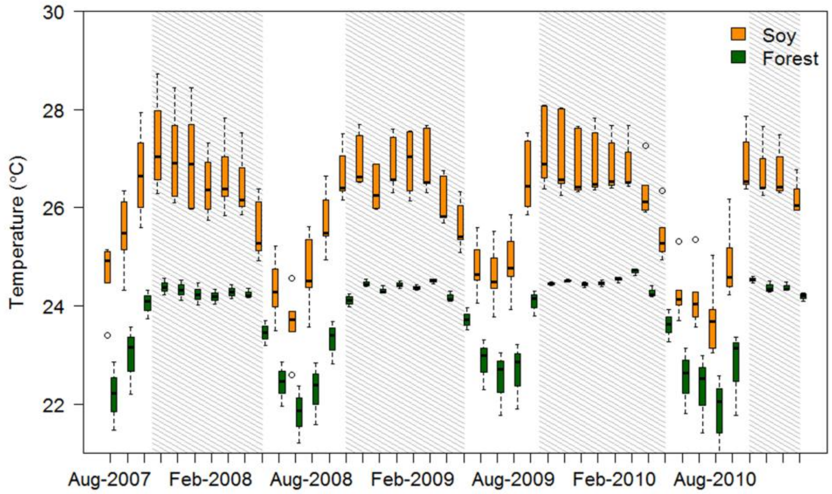

Figure 3.7: Monthly stream temperature in first order streams in soy (N=6) and forest (N=3) watersheds from August 2007 to December 2010. The hatched background depicts wet season months.

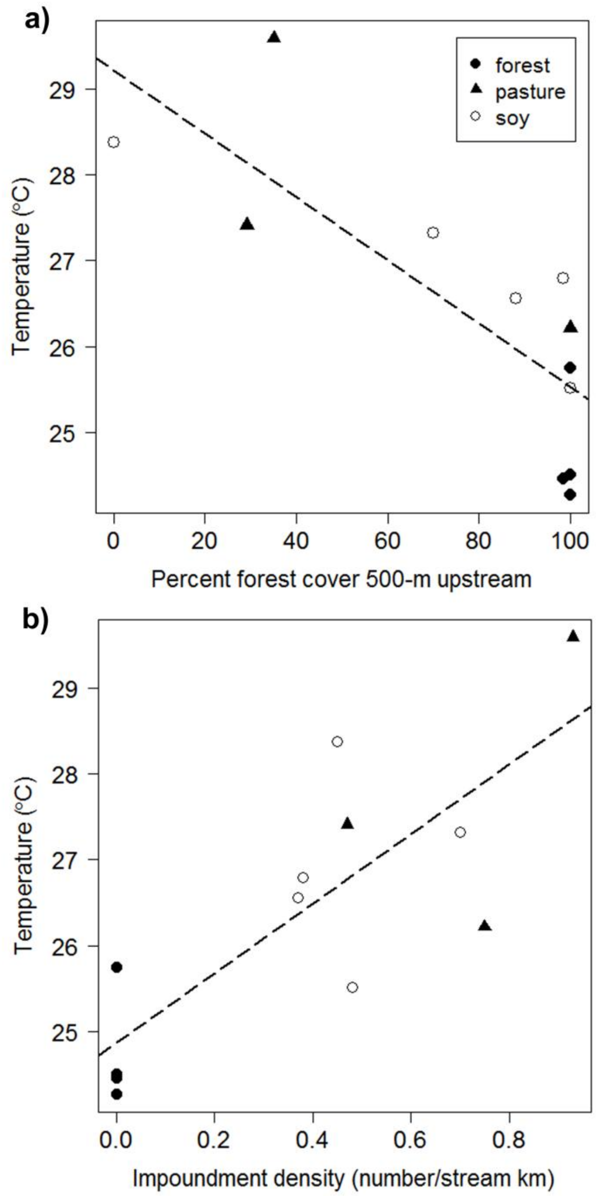

Figure 3.8: Relationship between mean daytime stream temperature (upper quantile) and covariates related to land management. (a) Stream temperature was negatively correlated with percent forest cover in the riparian buffer 500-m upstream ($p<0.01$, $R^2=0.65$). (b) Stream temperature was positively correlated with the density of impoundments upstream of the sampling location ($p<0.01$, $R^2=0.64$).

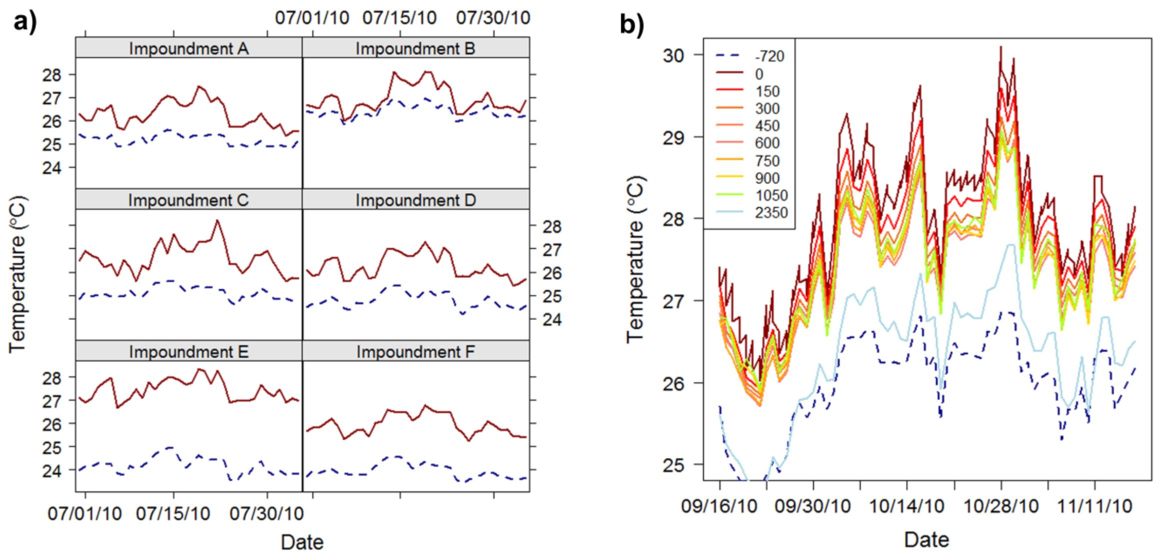

Figure 3.9: Influence of impoundments on stream temperature. (a) Upper quantile temperature upstream (dashed blue line) and downstream (solid red line) of six impoundments in the study area. (b) Transect showing recovery of stream temperature downstream of impoundment D. Loggers were placed every 150-m below the outlet (0-m) of the impoundment from 0- to 1050-m and one logger was placed at 2350-m downstream.

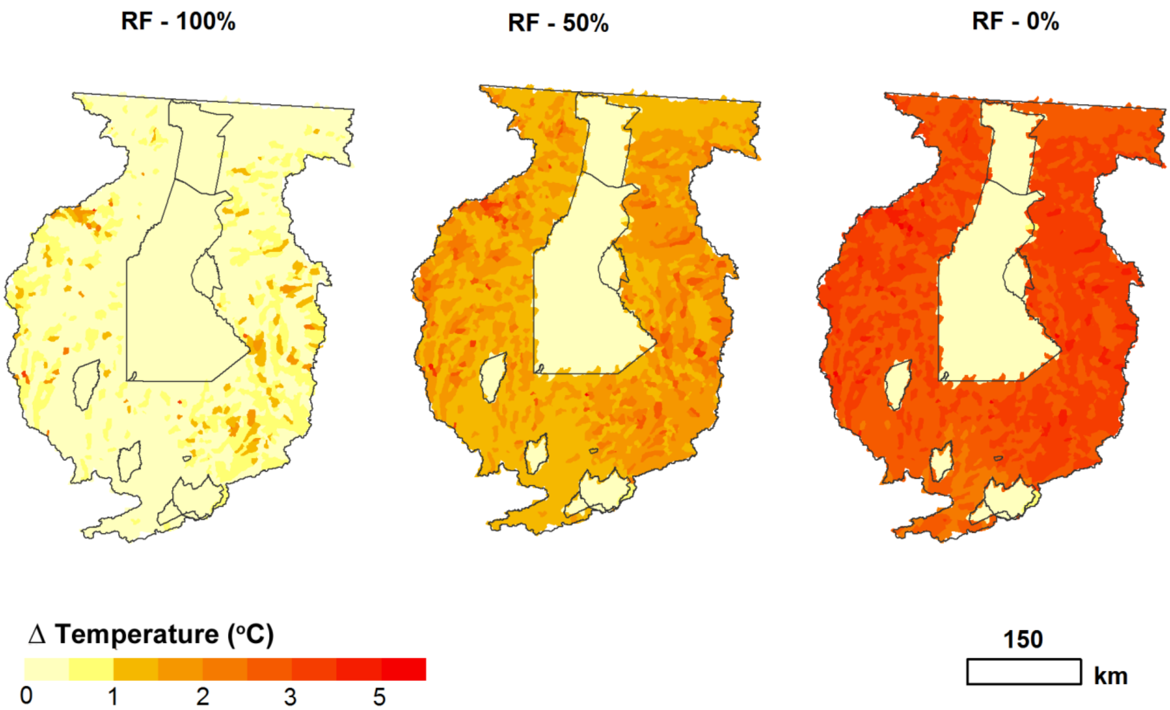

Figure 3.10: Predicted increases in headwater stream temperature under different management scenarios. Temperature was calculated for each microbasin (ANA, 2010) in the upper Xingu Basin, and is expressed as the deviation from a hypothetical reference scenario that assumed no impoundments and 100% riparian forest (RF) conservation. The three modeled scenarios use the impoundment density mapped for 2007 and varied riparian forest cover such that forest buffers were 100%, 50%, and 0% conserved within each microbasin.

References

Alexandre C. M., Almeida P. R. (2010) The impact of small physical obstacles on the structure of freshwater fish assemblages. *River Research and Applications,* **26**, 977-994.

ANA (2010) National Hydrographic Division - Ottobacias. Available at http://www.ana.gov.br/bibliotecavirtual/solicitacaoBaseDados.asp. Accessed on June 15, 2010. Brazilian National Water Agency.

Azevedo A. A. (2009) Legitimating Unsustainability? Analysis of the Environmental Licensing System for Rural Properties - SLAPR (Mato Grosso). Ph.D. in Sustainable Development, Center for Sustainable Development (CDS). University of Brasília, Brasília, Brasil, 325 pp.

Batista J. D. (2010) Sazonalidade, impacto ambiental e o padrão de deiversidade beta de *Odonata* em riachos tropicais no Brasil Central. Ph.D., Dept. of Entomology. Federal University of Viçosa, Viçosa, Minas Gerais, Brasil, 98 pp.

Benstead J. P., Douglas M. M., Pringle C. M. (2003) Relationships of stream invertebrate communities to deforestation in eastern Madagascar. *Ecological Applications,* **13**, 1473-1490.

Breiman L. (2001) Random forests. *Machine Learning,* **45**, 5-32.

Brett J. R. (1956) Some principles in the thermal requirements of fishes. *Quarterly Review of Biology,* **31**, 75-87.

Bruijnzeel L. A. (2004) Hydrological functions of tropical forests: not seeing the soil for the trees? *Agriculture Ecosystems & Environment,* **104**, 185-228.

Bruinsma J. (2009) The resource outlook to 2050. By how much do land, water use and crop yields need to increase by 2050? In: *Expert Meeting on How to Feed the World in 2050.* Rome, Italy, Food and Agriculture Organization of the United Nations Economic and Social Development Dept.

Buisson L., Blanc L., Grenouillet G. (2008) Modelling stream fish species distribution in a river network: the relative effects of temperature versus physical factors. *Ecology of Freshwater Fish,* **17**, 244-257.

Burnham K. P., Anderson D. R. (2002) Information theory and log-likelihood models: a basis for model selection and inference. In: *Model selection and multimodel inference: a practical information-theoretic approach.* pp 33-74. New York, Springer.

Caissie D. (2006) The thermal regime of rivers: a review. *Freshwater Biology,* **51**, 1389-1406.

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. *J Hydrology,* **369**, 165-174.

Coe M. T., Latrubesse E. M., Ferreira M. E., Amsler M. L. (2011) The effects of deforestation and climate variability on the streamflow of the Araguaia River, Brazil. *Biogeochemistry,* **105**, 119-131.

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.

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**, 23973-23989.

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**.

Cumming G. S. (2004) The impact of low-head dams on fish species richness in Wisconsin, USA. *Ecological Applications,* **14**, 1495-1506.

Deegan L., Neill C., Haupert C., Ballester M., Krusche A., Victoria R., Thomas S., de Moor E. (2011) Amazon deforestation alters small stream structure, nitrogen biogeochemistry and connectivity to larger rivers. *Biogeochemistry,* **105**, 53-74.

DeFries R., Rosenzweig C. (2010) Toward a whole-landscape approach for sustainable land use in the tropics. *Proceedings of the National Academy of Sciences of the United States of America*, **107**, 19627-19632.
Durance I., Ormerod S. J. (2007) Climate change effects on upland stream macroinvertebrates over a 25-year period. *Global Change Biology,* **13**, 942-957.
Eaton J. G., Scheller R. M. (1996) Effects of climate warming on fish thermal habitat in streams of the United States. *Limnology and Oceanography,* **41**, 1109-1115.
Evans E. C., McGregor G. R., Petts G. E. (1998) River energy budgets with special reference to river bed processes. *Hydrological Processes,* **12**, 575-595.
Fan Y., van den Dool H. (2008) A global monthly land surface air temperature analysis for 1948–present. *J. Geophys. Res.,* **113**, D01103.
Ferreira N., Ferreira L., Huete A., Ferreira M. (2007) An operational deforestation mapping system using MODIS data and spatial context analysis. *International Journal of Remote Sensing,* **28**, 47-62.
Franco E. A. D., Budy P. (2005) Effects of biotic and abiotic factors on the distribution of trout and salmon along a longitudinal stream gradient. *Environmental Biology of Fishes,* **72**, 379-391.
Freeman M. C., Pringle C. M., Jackson C. R. (2007) Hydrologic connectivity and the contribution of stream headwaters to ecological integrity at regional scales. *Journal of the American Water Resources Association,* **43**, 5-14.
Gelman A., Hill J. (2007) *Data analysis using regression and multilevel/hierarchical models,* Cambridge ; New York, Cambridge University Press.
Gergel S. E., Turner M. G., Miller J. R., Melack J. M., Stanley E. H. (2002) Landscape indicators of human impacts to riverine systems. *Aquatic Sciences,* **64**, 118-128.
Gillooly J. F., Brown J. H., West G. B., Savage V. M., Charnov E. L. (2001) Effects of size and temperature on metabolic rate. *Science,* **293**, 2248-2251.
Gillooly J. F., Charnov E. L., West G. B., Savage V. M., Brown J. H. (2002) Effects of size and temperature on developmental time. *Nature,* **417**, 70-73.
Gillooly J. F., Londono G. A., Allen A. P. (2008) Energetic constraints on an early developmental stage: a comparative view. *Biology Letters,* **4**, 123-126.
Goulding M., Barthem R., Ferreira E. J. G. (2003) *The Smithsonian atlas of the Amazon,* Washington, DC, Smithsonian Institution Press.
Hayhoe S. J., Neill C., Porder S., McHorney R., Lefebvre P., Coe M. T., Elsenbeer H., Krusche A. V. (2011) Conversion to soy on the Amazonian agricultural frontier increases streamflow without affecting stormflow dynamics. *Global Change Biology,* **17**, 1821-1833.
Hijmans R. J., Cameron S. E., Parra J. L., Jones P. G., Jarvis A. (2005) Very high resolution interpolated climate surfaces for global land areas. *International Journal of Climatology,* **25**, 1965-1978.
Hochachka P. W., Somero G. N. (2002) *Biochemical adaptation: mechanism and process in physiological evolution,* New York, Oxford University Press.
Huffman G. J., Adler R. F., Bolvin D. T., Gu G., Nelkin E. J., Bowman K. P., Hong Y., Stocker E. F. *et al.* (2007) The TRMM Multi-satellite Precipitation Analysis: Quasi-Global, Multi-Year, Combined-Sensor Precipitation Estimates at Fine Scale. *J. Hydrometeor.,* **8**, 38-55.
Huston M. A., Brown J. H., Allen A. P., Gillooly J. F. (2003) Heat and biodiversity. *Science,* **299**, 512-512.
Kaushal S. S., Likens G. E., Jaworski N. A., Pace M. L., Sides A. M., Seekell D., Belt K. T., Secor D. H. *et al.* (2010) Rising stream and river temperatures in the United States. *Frontiers in Ecology and the Environment,* **0**.
Kellndorfer J., Walker W., Pierce L., Dobson C., Fites J. A., Hunsaker C., Vona J., Clutter M. (2004) Vegetation height estimation from shuttle radar topography mission and national elevation datasets. *Remote Sensing of Environment,* **93**, 339-358.

Lehner B., Liermann C. R., Revenga C., Vörösmarty C., Fekete B., Crouzet P., Döll P., Endejan M. *et al.* (2011) High-resolution mapping of the world's reservoirs and dams for sustainable river-flow management. *Frontiers in Ecology and the Environment,* **9**, 494-502.

Loarie S. R., Lobell D. B., Asner G. P., Field C. B. (2011a) Land-cover and surface water change drive large albedo increases in South America. *Earth Interactions,* **15**.

Loarie S. R., Lobell D. B., Asner G. P., Mu Q. Z., Field C. B. (2011b) Direct impacts on local climate of sugar-cane expansion in Brazil. *Nature Climate Change,* **1**, 105-109.

Lorion C. M., Kennedy B. P. (2009a) Relationships between deforestation, riparian forest buffers and benthic macroinvertebrates in neotropical headwater streams. *Freshwater Biology,* **54**, 165-180.

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

Macedo M., DeFries R., Morton D., Stickler C., Galford G., Shimabukuro Y. (2012) Decoupling of deforestation and soy production in the southern Amazon during the late 2000s. *PNAS,* **109**, 1341-1346.

Malcolm I. A., Hannah D. M., Donaghy M. J., Soulsby C., Youngson A. F. (2004) The influence of riparian woodland on the spatial and temporal variability of stream water temperatures in an upland salmon stream. *Hydrology and Earth System Sciences,* **8**, 449-459.

Morton D. C., DeFries R. S., Shimabukuro Y. E., Anderson L. O., Arai E., Espirito-Santo F. D., Freitas R., Morisette J. (2006) Cropland expansion changes deforestation dynamics in the southern Brazilian Amazon. *Proceedings of the National Academy of Sciences of the United States of America,* **103**, 14637-14641.

Morton D. C., DeFries R. S., Shimabukuro Y. E., Anderson L. O., Espirito-Santo F. D. B., Hansen M., Carroll M. (2005) Rapid assessment of annual deforestaion in the Brazilian Amazon using MODIS data. *Earth Interactions,* **9**.

Myrick C. A., Cech J. J. (2004) Temperature effects on juvenile anadromous salmonids in California's central valley: what don't we know? *Reviews in Fish Biology and Fisheries,* **14**, 113-123.

Neill C., Elsenbeer H., Krusche A. V., Lehmann J., Markewitz D., Figueiredo R. D. (2006) Hydrological and biogeochemical processes in a changing Amazon: results from small watershed studies and the large-scale biosphere-atmosphere experiment. *Hydrological Processes,* **20**, 2467-2476.

Neuheimer A. B., Thresher R. E., Lyle J. M., Semmens J. M. (2011) Tolerance limit for fish growth exceeded by warming waters. *Nature Clim. Change,* **1**, 110-113.

Pinheiro J. C., Bates D. M. (2000) *Mixed-effects models in S and S-PLUS,* New York, Springer.

Pringle C. (2003) What is hydrologic connectivity and why is it ecologically important? *Hydrological Processes,* **17**, 2685-2689.

Rehwoldt R., Menapace L. W., Alessand.D, Nerrie B. (1972) Effect of increased temperature upon acute toxicity of some heavy-metal ions. *Bulletin of Environmental Contamination and Toxicology,* **8**, 91-&.

Roth T. R., Westhoff M. C., Huwald H., Huff J. A., Rubin J. F., Barrenetxea G., Vetterli M., Parriaux A. *et al.* (2010) Stream temperature response to three riparian vegetation scenarios by use of a distributed temperature validated model. *Environmental Science & Technology,* **44**, 2072-2078.

Sano E. E., Rosa R., Brito J. L. S. B., Ferreira L. G. (2007) Mapeamento de cobertura vegetal do bioma Cerrado: estratégias e resultados. In: *Pesquisa Agropecuaria Brasileira.* pp 153-156, Planaltina, DF, Brazil, EMBRAPA (Brazilian Agricultural Research Corporation) - Cerrado division.

Schielzeth H. (2010) Simple means to improve the interpretability of regression coefficients. *Methods in Ecology and Evolution,* **1**, 103-113.

SEMA (2010) Digital map database - MT stream network (hidrografia): Available at http://basesig.hd1.com.br/. Accessed on June 15, 2010. Mato Grosso State Environmental Agency.

Stickler C. M. (2009) Defending public interests in private forests: land use policy alternatives for the Xingu River headwaters region of southeastern Amazônia. Ph.D., Geography. University of Florida, Gainesville, FL, 199 pp.

Theurer F. D., Lines I., Nelson T. (1985) Interaction between riparian vegetation, water temperature, and salmonid habitat in the Tucannon River. *Water Resources Bulletin*, **21**, 53-64.

Tollefson J. (2011) Brazil revisits forest code. *Nature*, **476**, 259-260.

Uriarte M., Yackulic C. B., Lim Y., Arce-Nazario J. A. (2011) Influence of land use on water quality in a tropical landscape: a multi-scale analysis. *Landscape Ecology*, **26**, 1151–1164.

Vannote R. L., Minshall G. W., Cummins K. W., Sedell J. R., Cushing C. E. (1980) The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences*, **37**, 130-137.

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

Walker W. S., Kellndorfer J. M., Pierce L. E. (2007b) Quality assessment of SRTM C- and X-band interferometric data: Implications for the retrieval of vegetation canopy height. *Remote Sensing of Environment*, **106**, 428-448.

Walker W. S., Stickler C. M., Kellndorfer J. M., Kirsch K. M., Nepstad D. C. (2010) Large-area classification and mapping of forest and land cover in the Brazilian Amazon: A comparative analysis of ALOS/PALSAR and Landsat data sources. *Ieee Journal of Selected Topics in Applied Earth Observations and Remote Sensing*, **3**, 594-604.

Walter R. C., Merritts D. J. (2008) Natural streams and the legacy of water-powered Mills. *Science*, **319**, 299-304.

Ward J. V. (1985) Thermal characteristics of running waters. *Hydrobiologia*, **125**, 31-46.

Ward J. V., Stanford J. A. (1982) Thermal responses in the evolutionary ecology of aquatic insects. *Annual Review of Entomology*, **27**, 97-117.

Webb B. W., Nobilis F. (2007) Long-term changes in river temperature and the influence of climatic and hydrological factors. *Hydrological Sciences Journal-Journal Des Sciences Hydrologiques*, **52**, 74-85.

Zuur A. F., Ieno E. N., Walker N., Saveliev A. A., Smith G. M. (2009) *Mixed effects models and extensions in ecology with R*, New York, NY, Springer.

Chapter 4

Hydrologic connectivity and the future of Amazon protected areas

Abstract

The Amazon Basin contains the world's greatest diversity of freshwater fish, as well as productive fisheries that are a vital source of protein for local people. The aquatic ecosystems that support these resources are increasingly under threat, primarily due to land-cover and land-use changes (LCLUC) in upland forest areas. Large-scale deforestation may degrade stream ecosystems through a variety of mechanisms, including fragmentation due to roads and impoundments; changes in the amount of light, sediments, and nutrient inputs from riparian areas; and alterations to both the quality and quantity of water flowing within streams. Although indigenous lands and protected areas (ILPAs) are the primary tools being used to safeguard forests, their design has often overlooked the hydrological connections linking them to surrounding landscapes. This study examines the vulnerability of the Amazon ILPA network to future losses in hydrologic connectivity at two scales. First, we examine the case of the Xingu Indigenous Park (PIX), a 2.6 million ha indigenous reserve located in the heart of the Amazon's 'arc of deforestation'. The PIX has an upstream zone of influence (ZOI) roughly four times its size, with approximately 40% of its area in agriculture and an estimated 7,500 existing farm impoundments. Future projections indicate that deforestation in the landscape will increase to between 49 and 79% of the ZOI by 2050. Scaling up, we find that 30% of existing Amazon ILPAs are highly vulnerable to potential future reductions in hydrologic connectivity, simply because of their location within their watersheds. Of these, between 26 and 50% are likely to experience substantial deforestation (>40%) within their ZOIs by 2050. The long-term success of

Amazon ILPAs in conserving freshwater resources will require looking well beyond their boundaries to mitigate the influence of LCLUC upstream. By combining information on vulnerability and the likely timing of future deforestation, this study provides a simple framework for prioritizing these landscape management efforts.

Introduction

The Amazon Basin contains the greatest diversity of freshwater fish species of any watershed in the world (Abell *et al.*, 2008, Revenga *et al.*, 1998, Thieme *et al.*, 2007). Inland fisheries yield 450,000 tons of fish each year and are a vital source of protein for local people (Junk *et al.*, 2007), yet the aquatic ecosystems that support these resources are under increasing pressure. Today, large-scale deforestation is among the most pervasive threats to freshwater ecosystems in the Amazon and is increasingly driven by global demand for agricultural commodities such as soybeans, biofuels, and beef (Macedo *et al.*, 2012, Nepstad *et al.*, 2006b). Watershed forest cover is a key determinant of fundamental hydrological processes, including evapotranspiration (ET), rainfall, stream flow, and downstream fluxes of sediments and nutrients. In addition to altering these processes, the large-scale conversion of forests to pasture grasses and croplands may further degrade stream ecosystems by changing the amount of light and nutrient inputs from riparian areas, increasing stream temperature, increasing fragmentation due to roads and impoundments, and altering water quality due to increases in sediment and pesticide loading. Hence, large-scale agricultural expansion has the potential to fundamentally change the quality and distribution of freshwater habitats within a stream network, with direct consequences for stream biota and the overall integrity of fluvial ecosystems (Greenwood *et al.*, 2011, Lorion & Kennedy, 2009a, Lorion & Kennedy, 2009b).

Today, indigenous lands and protected areas[13] (ILPAs) are the primary tool being employed to conserve forested areas in the face of growing threats. A great deal of research effort has been invested in evaluating the effectiveness of the Amazon protected area network and its potential to provide lasting environmental services, including carbon storage and reduction of greenhouse gas emissions (Fearnside, 2009, Ricketts *et al.*, 2010, Soares *et al.*, 2010); inhibition of land-use related fires and deforestation (Adeney *et al.*, 2009, Bruner *et al.*, 2001, Ewers & Rodrigues, 2008, Joppa *et al.*, 2008, Nelson & Chomitz, 2011, Nepstad *et al.*, 2006a); conservation of representative biodiversity (Schulman *et al.*, 2007); and buffering against potential climatic "tipping points" that threaten to desiccate Amazonian forests and push them towards a more savannah-like physiognomy (Nepstad *et al.*, 2008, Walker *et al.*, 2009). On balance, these studies indicate that the Amazon network of ILPAs effectively conserves standing forests and, by proxy, terrestrial biodiversity, but there has been little emphasis on assessing its effectiveness in conserving freshwater ecosystems and maintaining hydrological ecosystem services.

The conservation of aquatic biodiversity poses unique challenges to protected area managers, who are often charged with managing freshwater resources within park boundaries in the face of threats that may be hundreds of kilometers upstream and well outside their jurisdiction. Part of the difficulty stems from the fact that most tropical protected areas have been designed with terrestrial conservation criteria in mind, often ignoring the connectivity of freshwater systems by excluding critical parts of a watershed (Abell *et al.*, 2007, Nel *et al.*, 2011, Pringle, 2001). The flow of water within river networks is thus an important mechanism linking

[13] The term "protected area" refers to any area of land or sea managed for the persistence of biological diversity and other natural processes through constraints on incompatible land uses (Possingham *et al.*, 2006). In this study, the "ILPA network" includes both strict nature reserves and managed use areas, such as indigenous lands and sustainable use reserves, which serve the dual purpose of conserving biological diversity and sustaining human livelihoods.

protected areas to surrounding landscapes, making them vulnerable to land use changes occurring outside their boundaries – changes that may alter the ecological flows required to maintain disturbance regimes, nutrient flows, organism movement, and population processes within the reserve (Hansen & DeFries, 2007, Herbert *et al.*, 2010). As protected areas become increasingly isolated (DeFries *et al.*, 2005) and embedded in human-dominated landscapes (Wittemyer *et al.*, 2008), there is a growing awareness that effective management must consider the surrounding landscape (DeFries *et al.*, 2007, Ewers & Rodrigues, 2008, Hansen & DeFries, 2007). Recent studies have suggested methodologies for delineating the zones of influence (ZOI) that may impact a protected area as an important first step in developing effective management plans, identifying potential threats in the surrounding landscape, and setting realistic conservation targets (DeFries *et al.*, 2010a, DeFries *et al.*, 2010b, Hansen *et al.*, 2011). Here, we use the term ZOI to refer only to the *hydrological* zone of influence upstream of protected areas.

The concept of connectivity has been used extensively to describe spatial connections in riverine landscapes (Amoros & Bornette, 2002, Pringle, 2003, Ward, 1989, Ward *et al.*, 2002). Ward (1989) describes rivers as having interactive pathways in four dimensions, one temporal and three spatial, consisting of longitudinal (headwater-estuarine), lateral (riverine-riparian), and vertical (riverine-groundwater) connections. The location of a protected area within this hydrological landscape (Fig. 4.1) plays a key role in determining how it will be affected by alterations in hydrologic connectivity, defined as the “water-mediated transport of matter, energy and organisms within and between elements of the hydrological cycle” (Freeman *et al.*, 2007, Pringle, 2003, Pringle, 2001). Human development activities may reduce hydrologic connectivity directly via the expansion of infrastructure (e.g., dams, roads, and farm impoundments), water diversion, groundwater extraction, and irrigation. The resulting fragmentation may be exacerbated by indirect impacts associated with the removal of native vegetation, including changes in patterns of discharge (Coe *et al.*, 2009, Coe *et al.*, 2011, Hayhoe *et al.*, 2011), local and regional precipitation (Da Silva *et al.*, 2008, Werth & Avissar, 2002) and ET, stream temperature (Macedo *et al.*, in prep.,Caissie, 2006), and nutrient and sediment fluxes.

Habitat fragmentation in dendritic landscapes (e.g., stream networks) has different – and arguably more severe – consequences for fragment size than in linear or two-dimensional systems (Grant *et al.*, 2007). The result is smaller fragments and higher variance in fragment size (Freeman *et al.*, 2007), which can lead to pronounced mismatches between the geometry of dispersal and the geometry of disturbance. This disparity can have critical implications for population persistence (Fagan, 2002, Fagan *et al.*, 2002). Even in areas where stream reaches remain structurally connected, fluvial species may experience a functional decrease in connectivity. For example, the removal of riparian forests has been shown to increase stream temperatures (Macedo *et al.*, in prep.), decrease shading, and reduce the inputs of leaf litter and large woody debris (Wright & Flecker, 2004), all of which alter the quality and distribution of habitat for fish and other aquatic organisms (Lorion & Kennedy, 2009a, Lorion & Kennedy, 2009b). Small impoundments not only increase local stream temperature but also create a lentic (lake-like) environment in place of a lotic (stream) environment. Despite being physically connected by water, stream reaches with impoundments and degraded riparian areas may require fluvial species to move through suboptimal habitat conditions (Schlosser *et al.*, 2000), thus increasing the resistance to movement within the stream network. This loss of connectivity negatively impacts fish dispersal and recolonization after an extreme event (Hess, 1996) and increases the likelihood of local extinction (Fagan, 2002).

Although there is growing recognition of the need to incorporate freshwater criteria into conservation planning and protected area design (Abell *et al.*, 2007, Herbert *et al.*, 2010, Nel *et al.*, 2009, Thieme *et al.*, 2007), the development and implementation of these principles is years behind that in the terrestrial – or even the marine – realm (Barmuta *et al.*, 2011). Few freshwater protected areas exist today, even though the loss of freshwater biodiversity is occurring at a rapid pace (Abell *et al.*, 2007, Dudgeon *et al.*, 2006, Johnson *et al.*, 2008, Vorosmarty *et al.*, 2010). This study examines the potential for the existing network of Amazon protected areas to fill this gap through management of the surrounding landscape. In the context or freshwater resources, specific management strategies might include: the development of integrated watershed land use plans; conservation and restoration of riparian forest buffers in agricultural watersheds; regulation to minimize the negative impacts of hydroelectric dams, roads, and farm impoundments; monitoring and enforcement of existing environmental legislation; watershed fisheries management; and the creation of policy incentives that encourage environmentally sound land management (e.g., no-till agriculture, erosion control, livestock fencing) on private properties.

This study examines the status of the current Amazon ILPA network from the perspective of freshwater conservation, focusing on two scales. At the landscape scale, we examine the case of the Xingu Indigenous Park (PIX), a large indigenous reserve in the heart of the Amazon's agricultural frontier. At the scale of the Amazon Basin, we evaluate the vulnerability of ILPAs to future losses in hydrologic connectivity due to deforestation and development within their zones of influence. We focus on four central questions:

(1) To what extent has land use change in the headwaters of the Xingu Basin altered hydrologic connectivity in the PIXs zone of influence?

(2) To what extent does the existing network of Amazon ILPAs contribute to ecological services through hydro-climatic regulation (i.e., evapotranspiration) and the protection of critical habitats (i.e., wetlands)?

(3) How vulnerable are Amazon Basin ILPAs to hydrologic fragmentation due to future land-use/cover changes within their watersheds?

(4) Given limited resources, how can we prioritize management of ILPAs based on the likely location and timing of development?

Data and Methods

Study area

The Xingu Indigenous Park and the subwatersheds that drain into it occupy an area of approximately 14 million ha in the state of Mato Grosso, Brazil. The 2.6 million ha indigenous reserve was created in 1964 for the subsistence of the 14 ethnic groups living within its boundaries. At the time, the region was largely forested and the park boundaries were demarcated in such a way that they excluded most of the headwaters region, which flow directly into the park via major tributaries of the Xingu River. Today, the PIX is in the heart of the Amazon's agricultural frontier and immediately downstream of a region that has undergone rapid deforestation over the last two decades and is now one of Brazil's major cattle and soy producing regions (Morton *et al.*, 2006, Nepstad *et al.*, 2006b, Stickler *et al.*, 2009).

Moving beyond the agricultural frontier, the Amazon ILPA system is comprised of over 500 conservation units, including strict protected areas (i.e., national parks and ecological reserves), indigenous lands, and sustainable use areas that are managed for timber or non-timber forest products. Our assessment considers the drainage area of the Amazon Basin, which encompasses approximately 6.9 million km^2 and spans seven countries (Fig. 4.1). Because our focus is on hydrologic connectivity, we did not consider the easternmost regions of the Brazilian Legal Amazon, which occur in the Cerrado biome and form the headwaters of major rivers in eastern South America (Coe *et al.*, 2011), nor did we consider the adjacent forested regions of the Orinoco Basin and Guiana shield, which drain northward towards the Atlantic Ocean.

Data sources

In order to characterize the degree of freshwater protection afforded by Amazon ILPAs, as well as their upstream zones of influence, we combined several datasets derived from satellite-based sensors with existing databases of protected areas and watershed boundaries. Boundaries for the Amazon protected area network come from a database put together from various sources by the Amazon Scenarios Project, as did modeling results projecting deforestation until 2050, based on different development scenarios (Nepstad *et al.*, 2009, Soares *et al.*, 2010, Soares *et al.*, 2006). Boundaries for indigenous lands in the Brazilian Amazon came from Brazil's National Indian Foundation (FUNAI, 2011). Watershed boundaries came from the Brazilian water agency (ANA, 2010) and a digitized stream network for the upper Xingu from the Mato Grosso State Environmental Agency (SEMA, 2010). For watershed delineation, we used a hydrologically conditioned digital elevation model (DEM) from the 15 arc-second Hydrosheds product (Lehner *et al.*, 2006).

Land use data for the upper Xingu Basin was derived from the Moderate Resolution Imaging Spectroradiometer (MODIS) enhanced vegetation index (MOD13Q1; Macedo *et al.*,

2012) time series (2001-2010), combined with datasets on the distribution (Sano *et al.*, 2007) and deforestation (Ferreira *et al.*, 2007) of native cerrado vegetation. Spatial data on the distribution of wetlands was derived from Japanese Earth Resources Satellite Synthetic Aperture Radar (JERS SAR) images (Hess *et al.*, 2009) and data on ET in the Amazon Basin came from the MODIS ET data product (MOD16). The distribution of impoundments was derived from a 2007 mosaic of the Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER; Macedo *at al.*, in prep.).

Delineating the hydrological zone of influence

For the purposes of this analysis we defined the zone of influence as the drainage area flowing into a protected area via the stream network. We excluded rivers that form the boundary of a protected area, but never flow through it. This is a relatively common design in the Amazon ILPA network, but we assume that managers have little jurisdiction over the management of these rivers. Our analysis treated each protected area as an independent unit, even when it occurred within a large corridor of protected areas. Although having adjacent protected areas in the ZOI reduces the likelihood of land-use related disruptions to hydrologic connectivity, the effective conservation of aquatic resources in an individual protected area will require, at a minimum, coordination with ILPA managers upstream to harmonize management activities and potentially competing uses of freshwater resources.

We used the ArcHydro tools in ArcGIS 10 for all watershed delineation and stream generation. First, we defined the stream network using the Hydrosheds flow direction grid at 15 arc-second resolution (Lehner *et al.*, 2006). Next, we generated a point layer that represents the intersection of the stream vector with any protected area boundary. This layer was edited to remove spurious drainage points generated along the boundaries where rivers and protected areas coincided, as well as points that represented outlets where rivers drained out of the protected area. The final layer of drainage points represented inlets that connect the protected area interior to the zone of influence upstream. These inlets were used to delineate subwatersheds comprising the zone of influence for each protected area. The final ZOI layer was generated by merging all subwatersheds contributing to a given conservation unit.

Assessing hydrological function and vulnerability of ILPAs

For the case of the Xingu Indigenous Park, we used existing land-use and land-cover information to assess the degree to which past LCLUC in the region has decreased hydrologic connectivity upstream. Specifically, we assess the spatial distribution of agriculture (soy croplands and cattle pastures) and the density of upstream impoundments in the ZOI. To do this, we summarized recent data on land use (Macedo *et al.*, 2012) and impoundments (Macedo *et al.*, in prep.) using the Brazilian Water Agency's smallest watershed unit, which we refer to here as a microbasin. We then compared 2010 land cover in this agricultural frontier with modeled scenarios of deforestation in 2050 under business as usual (BAU) and governance (GOV) assumptions (Fig. 4.3; Soares *et al.*, 2006). As described by Soares et al. (2006), the BAU scenario assumed that deforestation rates during the period from 1997 to 2002 would continue unabated. The governance scenario assumed improved governance and stricter limits on the amount and location of deforestation. Both scenarios assumed that the paving of new roads in the region would go forward as planned. Because the Amazon Scenarios model masked out the non-forest (cerrado) areas in the southern portion of the Xingu Basin, our analysis only considered the forested portion of the ZOI, where land cover information was available for both current and future scenarios. Finally, we compared the distribution of forest cover (%) in the ZOI and the 1166 microbasins it encompasses for present-day (2010) and predicted future (2050) scenarios.

As a preliminary assessment of the contribution of the ILPA network to hydro-climatic cycling, we estimated the proportion of mean annual ET in the Amazon Basin attributable to forests within protected areas. Similarly, we examined the location of Amazon ILPAs relative to the distribution of wetland and floodplain areas in order to estimate the proportion of wetland ecosystems that is legally protected. To assess the vulnerability of these ILPAs to future hydrologic fragmentation, we developed a hydrologic connectivity index (HCI) to summarize information on the zone of influence for each protected area. First we converted all data layers to an equal area projection (South America Albers Equal Area) and calculated the area of each ILPA and its ZOI. The HCI is simply the ratio of the ZOI to the ILPA area, allowing a rapid assessment of the relative location of each PA within its watershed and its vulnerability to upstream land-use changes and losses in connectivity. We used ordinary least squares regression (OLS) models to examine the relationship between the (log) area of ILPAs and both the (log) area of the ZOI and the (log) HCI. We also examined the effect of protected area type (i.e., indigenous, sustainable use, or strict) on both HCI and area, using a combination of analyses of variance (ANOVAs) and Tukey's HSD tests.

In a separate analysis, we considered areas with an HCI greater than one to be at high risk, because in these cases the ZOI exceeds the area under protection by the conservation unit and, by extension, a large proportion of the headwaters region is potentially vulnerable to land-use/cover changes. Using a similar logic, we classified areas with an HCI between 0.25 and 1 as medium risk and those with an HCI between 0 and 0.25 as low risk (Table 4.1). In the case of Jaú National Park, we elected to exclude one of the contributing subwatersheds, which would have resulted in a ZOI encompassing a large fraction of the Rio Negro Basin based on a short stretch of river passing through a corner of the park. This change switched Jaú from a high-risk to a low-risk status, which we deemed reasonable, since it is a large (>2 million ha) PA that encompasses most of the watershed of the Jaú River.

Results

Xingu Indigenous Park

The Xingu Indigenous Park has a hydrological zone of influence more than four times its size, with 14 subwatersheds and several major rivers draining into the reserve from the agricultural landscape outside its borders. This drainage area contains over 7,500 impoundments (Macedo *et al.*, in prep.) and an estimated 53,700 km of streams (SEMA, 2010). On average, there is at least one impoundment for every 7 km of stream length, although these are not evenly distributed in the landscape (Fig. 4.4b). Impoundments are most common in microbasins dominated by cattle pastures or soy croplands and more likely to occur in small headwater streams and along roads. As of the 2009-2010 growing year, 39% of the forested portion of the ZOI for the PIX was occupied by agriculture (Fig. 4.4a). Approximately one quarter of the area in agricultural production was under soybean cultivation and the remainder was in cattle ranching (Macedo *et al.*, 2012). If development in the region were to follow a BAU trajectory (Soares *et al.*, 2006), the deforested area in the ZOI is predicted to reach 79% by 2050. Under a governance scenario, the deforested area would increase to an estimated 48%. At the microbasin scale, this translates to half of all microbasins with less than 50% forest cover under BAU, compared to approximately one-third of microbasins in 2010 and 40% under GOV (Fig. 4.5).

Protecting hydrological services

As noted by Soares and coauthors (2010), the Amazon network of ILPAs already protects 54% of the remaining forests of the Brazilian Amazon and 56% of its forest carbon. These same forests also serve important hydrological functions by maintaining evapotranspiration (ET) and, thus, regulating regional water fluxes. Based on a combination of the MODIS ET time series and the protected area boundaries, we estimate that the network of Brazilian protected areas is also responsible for 54% of the annual ET flux in the Brazilian Amazon. The forests protected within indigenous reserves alone contribute approximately 26% of the annual ET flux for the Brazilian Amazon.

Given limited information about the distribution and diversity of freshwater species across the Amazon Basin, spatially explicit data on the distribution of ecologically important freshwater habitats may serve as a useful proxy. Wetland ecosystems and floodplain habitats are particularly rich and have been mapped successfully using a combination of optical and microwave remote sensing (Hess *et al.*, 2009). Although these wetland areas occupy only 20% of the Amazon Basin, they are among the most important in sustaining hydrological and biogeochemical processes, biodiversity, economically important fisheries, and local livelihoods. Despite their importance for freshwater conservation, we estimate that only 24% of Amazon wetlands are under formal protection today.

Amazon ILPAs

Our database contained a total of 539 ILPAs, of which 53% were indigenous reserves, 23% were sustainable use areas, and 24% were strict protected areas. In total, the network covers 208 million ha, roughly one-third of the Amazon Basin. The mean (log) area of indigenous reserves was significantly smaller than that of either strict or sustainable use areas (Tukey's HSD, $p<0.001$), even though they accounted for 47% of the area under protection. There was no statistical difference in the average size of strict and sustainable use areas.

A total of 252 ILPAs (47%) had an HCI value of zero, corresponding to an area of 50 million ha (24% of the total area under protection). Although small ILPAs (<250,000 ha, median area ~ 14,000 ha) accounted for 79% of the protected areas in this class, large ILPAs (median area ~ 500,000 ha) accounted for 84% of the area. These areas had no hydrological zone of influence and fell into two categories: 1) areas that encompassed entire watersheds or were situated in the headwaters, and 2) areas that had no streams flowing through them. The remaining 287 ILPAs had an HCI value greater than zero, indicating that they had a hydrological ZOI and that streams inside their borders were potentially vulnerable to land-use changes upstream. The (log) area of ILPAs was negatively correlated with (log) HCI (Fig. 4.6; $p<0.001$, $R^2=0.16$). Protected area type was not a significant predictor of the HCI and adding it as a factor in the regression model did not improve the fit ($\Delta AIC=1$).

Of the 253 protected areas (strict and sustainable use PAs) considered in this analysis, a total of 71 were classified as high risk (HCI>1) with respect to potential losses in hydrologic connectivity, 146 as low risk (HCI<0.25), and the remaining areas as medium risk (Table 4.1). Of the high-risk PAs, 27 were strict protected areas and 44 were sustainable use areas. Of the 286 indigenous lands, 58 were identified as high risk, 190 as low risk, and the remainder as medium risk (Table 4.1; Fig. 4.6).

Although the HCI may be an indicator of the vulnerability of ILPAs to potential development threats upstream, it does not capture the actual likelihood of threat in the coming decades, which may vary considerably depending on the PA's location within the Amazon Basin

(Fig. 4.3). In general, ILPAs in the major subbasins of the western Amazon (e.g., the Solimões and Negro) are less likely to face large-scale land-use changes in the coming decades than those in the eastern subbasins (e.g., the Xingu and Tocantins), where agricultural expansion and infrastructure development is already underway (Fig. 4.1 and Fig. 4.6). To better understand the connection between potential risk (HCI) and the likelihood of land-use related threats, we examined the zones of influence of all high risk ILPAs under the BAU and GOV scenarios. By 2050, approximately half of these ILPAs were predicted to experience greater than 40% deforestation within their ZOI under the business as usual scenario, compared to just over one-quarter under the governance scenario (Fig. 4.8).

Discussion

Headwater streams represent between two-thirds and three-fourths of total stream length in a typical drainage basin and directly connect the upland and riparian landscape to the rest of the stream ecosystem (Freeman *et al.*, 2007, Goulding *et al.*, 2003). They not only contribute to the overall biodiversity of a stream network but also provide important spawning and rearing areas, as well as a source of nutrients in the form of organic matter (Deegan *et al.*, 2011). The degradation of headwater streams and loss of connectivity to ecosystems downstream can, therefore, affect the biological integrity of entire river networks (Meyer *et al.*, 2007). This is of particular concern in the upper Xingu Basin where the area downstream of the headwaters is designated for the protection of biodiversity and subsistence of indigenous populations. This study documents the extent to which agricultural expansion and intensification has already impacted headwater streams in the southern Amazon and suggests that integrated management of the landscape may help mitigate the downstream effects on protected areas in this and other frontier regions.

In the case of the Xingu Indigenous Park, our results highlight the importance of considering hydrologic connectivity in the design and management of protected areas. Agricultural development in the Xingu headwaters had removed 40% of upland forest cover in the ZOI by 2010, with concomitant reductions in riparian forest buffers and increased fragmentation of the stream network by roads and small farm impoundments (Macedo *et al.*, in prep.). This case study underscores the challenges of managing freshwater resources within protected areas in the face of large-scale changes in the surrounding landscape. Our spatially (and hydrologically) explicit approach provides a straightforward methodology for identifying the subwatersheds that most threaten streams inside the PIX and other Amazon ILPAs. This information can be used as an objective way of prioritizing the location and timing of mitigation activities (e.g., watershed land use planning, riparian buffer restoration), allowing for more efficient coordination and use of resources.

The fact that the current ILPA network occupies over half of the Amazon Basin's remaining forests, while protecting less than one-quarter of its wetland areas, is a testament to the inherent bias of protected areas towards terrestrial conservation targets. Nevertheless, the existing ILPA network contributes significantly to regional ET and, with strategic management, may make an important contribution to the conservation of freshwater ecosystems. Over 60% of the ILPAs examined in this study were at low risk of hydrologic fragmentation, meaning they encompassed entire watersheds or protected substantial portions of sensitive headwater regions. Of the protected areas identified as high risk, only half were under threat of substantial development upstream over the next 40 years. Proactive management of the landscape surrounding the remaining ILPAs – including land use zoning and planning, protection of riparian and wetland areas on private properties, and regulation of impoundments – has the potential to make a large difference in the conservation outcome of high-risk reserves. This research has practical applications for the selection and design of new protected areas and improved management of existing areas, both within the Amazon Basin and in other tropical watersheds facing similar development pressures.

Limitations of the study and directions for future research

Our assessment of the threats to freshwater ecosystems within protected areas is limited to large-scale deforestation and subsequent land-use changes. We believe this is a good starting point because it links satellite-derived landscape metrics with freely available data on protected area locations and hydrology, providing a simple framework that is applicable to conservation planning in data-poor regions (Thieme *et al.*, 2007). It is worth noting, however, that the HCI index presented here is simplistic and captures only one aspect of ILPA vulnerability to hydrologic fragmentation. It does not, for example, consider the current and proposed distribution of hydroelectric dams in the Amazon, which directly impact hydrologic connectivity and, in some cases, may involve dam construction or large-scale flooding within protected areas. Furthermore, the HCI focuses solely on the potential influence of water flowing into protected areas from upstream, saying nothing of the freshwater conservation value of these areas. For example, a protected area with an HCI of zero may well have no streams or wetlands within its boundaries, making it of little use to freshwater conservation goals. Future research might focus on the development of a comprehensive index that incorporates more complex measures of freshwater conservation value, as well as other potential threats within the watershed.

Watershed vegetation is a key driver of fundamental hydrological processes, such as regional precipitation, discharge, and downstream fluxes of sediments and nutrients. In this regard, our analysis of ET considers only one aspect of hydrological cycling in ILPAs. Previous research indicates that large-scale deforestation alters the amount of ET, as well as the amount and timing of water flowing within rivers. By maintaining forest cover, ILPAs make an important contribution to this process, but large-scale deforestation outside protected areas has the potential to cause permanent reductions in regional rainfall and, thus, completely alter the capacity of ILPAs to provide hydrological services over the long term. Identifying these theoretical tipping points is an important area of on-going research.

In addition to deforestation, there are a number of disturbances that may not respect the one-way flow of water or the designated boundaries of protected areas. These include, but are not limited to: water diversion, hydroelectric dams, commercial fisheries, multi-national investments in oil extraction and large infrastructure projects, non-point source pollution, changing global demands for agricultural commodities, and global climate change. Any one of these mechanisms can substantially alter the scenarios discussed here, increasing the vulnerability of remote ILPAs and presenting additional threats to freshwater resources within protected areas. Effective management and planning will require an adaptive approach anchored in periodic reassessments of land-use trajectories and infrastructure plans, and the development of new models that take these into account.

A great deal of research effort has gone into bringing the principles of landscape ecology to bear on riverine ecosystems (Allan, 2004, Herbert *et al.*, 2010, Nel *et al.*, 2009), with many advances in the quantification of hydrologic connectivity (Calabrese & Fagan, 2004, Cote *et al.*, 2009, Erős *et al.*, 2011), assessment of critical watershed areas (Abell *et al.*, 2007, Barmuta *et* *al.*, 2011), and improved design of freshwater conservation areas. As suggested by Barmuta et al. (2011), it's time to bridge the gap between 'planning' and 'doing' in freshwater conservation. We suggest that a good way to start bringing these concepts to scale and testing them in the real world is by simply evaluating the existing ILPA network, identifying areas that are vulnerable to anthropogenic disturbance, and prioritizing management in areas that are under threat of development. This is a critical first step towards maximizing the potential of existing ILPAs to conserve freshwater ecosystems.

Table 4.1: Vulnerability of indigenous lands and protected areas to hydrologic fragmentation.

	Indigenous Lands			Strict PAs			Sustainable Use PAs		
Risk	**ILs (#)**	**Area (km²)**	**HCI**	**PAs (#)**	**Area (km²)**	**HCI**	**PAs (#)**	**Area (km²)**	**HCI**
High	58	186462	5.8	27	110287	3.2	44	118407	2.5
medium	38	284510	0.6	16	102091	0.5	20	167344	0.6
low	190	516375	0	85	368179	0	61	226112	0
Total	**286**	**987347**		**128**	**580557**		**125**	**511863**	

HCI, median Hydrologic Connectivity Index for each risk category; PA, protected area; IL, indigenous land

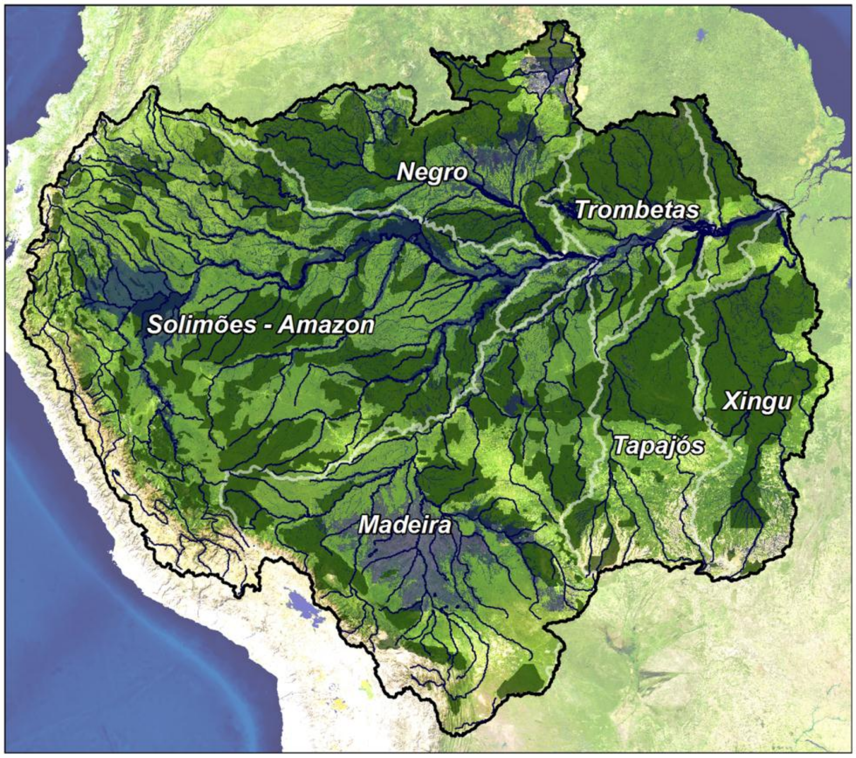

Figure 4.1: Overview of the Amazon Basin and its major sub-basins. Areas shaded in dark green include protected areas, indigenous reserves, and sustainable use areas. Areas shaded in blue represent wetland and seasonally flooded regions of the Basin (Hess *et al.*, 2009). *Map courtesy of Paul Lefebvre* (2012).

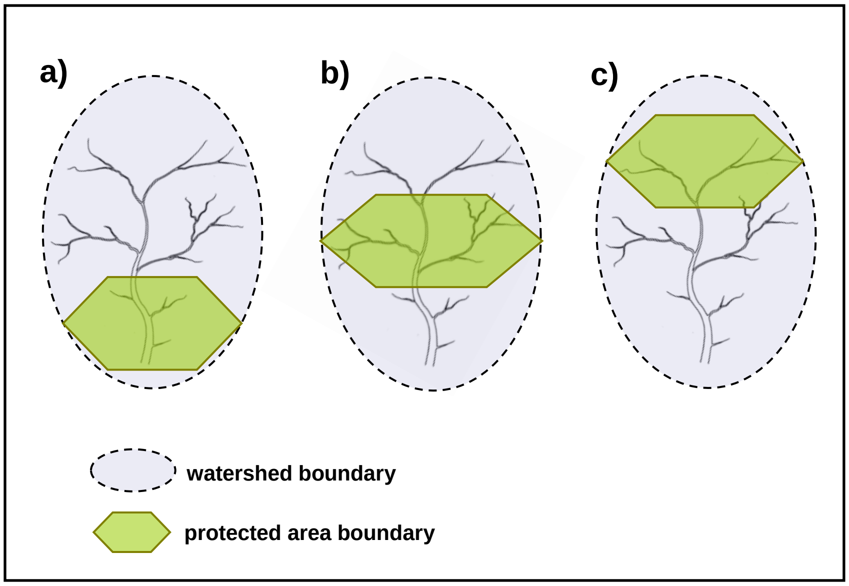

Figure 4.2: Three hypothetical locations of protected areas within the hydrological landscape: a) protection of the lower watershed (delta, estuary); b) protection of the middle watershed; and c) protection of the upper watershed (headwaters). Regardless of location, protected areas that occupy only a fraction of a watershed are vulnerable to hydrologic fragmentation and land cover change occurring outside their boundaries (adapted from Pringle, 2001).

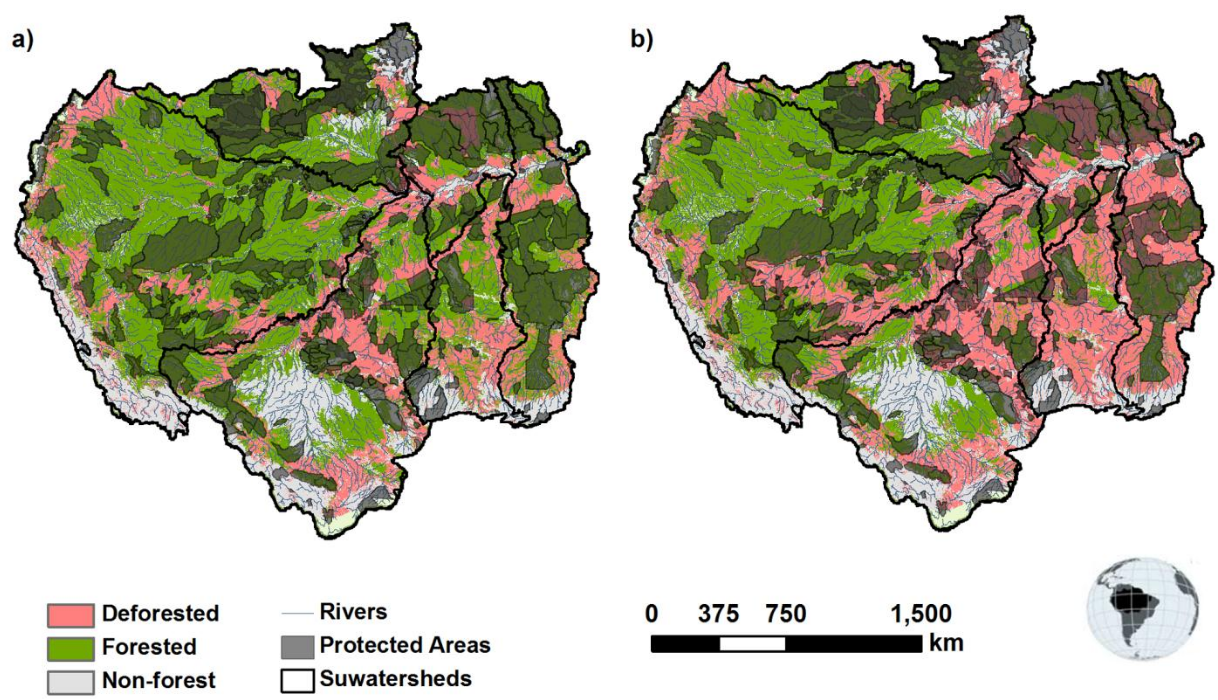

Figure 4.3: Modeled results for two development scenarios in the year 2050 (from Soares *et al.*, 2006) for major subwatersheds of the Amazon Basin. (a) Area deforested under improved governance (GOV). (b) Area deforested assuming business as usual (BAU).

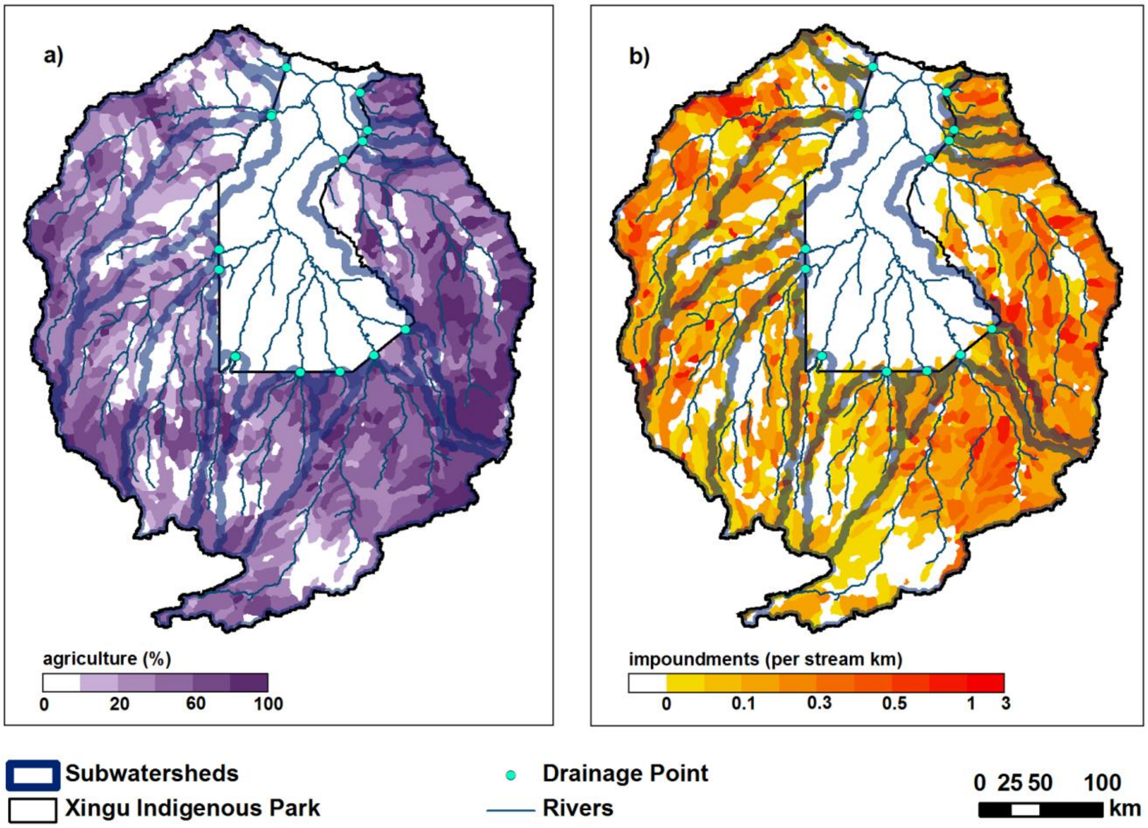

Figure 4.4: Agricultural development in the zone of influence outside the Xingu Indigenous Park, summarized by microbasin (ANA, 2010). (a) Density of agriculture (pasture and soy croplands) in 2010, based on a 250-m resolution land use classification (Macedo *et al.*, 2012). (b) Density of impoundments in 2007, based on a 15m resolution classification of impoundments (Macedo *et al.*, in prep.).

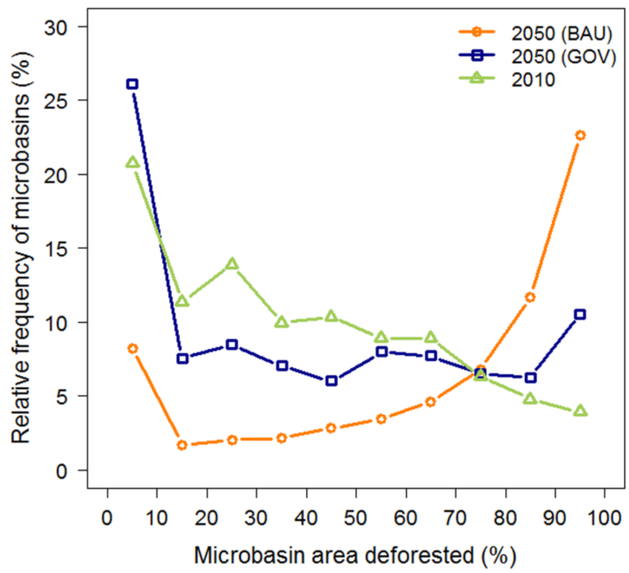

Figure 4.5: Relative frequency distribution of deforestation levels (%) in the 1166 microbasins comprising the zone of influence for the Xingu Indigenous Park. The actual distribution of area deforested in 2010 is compared to modeling results for deforestation under governance (GOV) and business as usual (BAU) scenarios (Soares *et al.*, 2006).

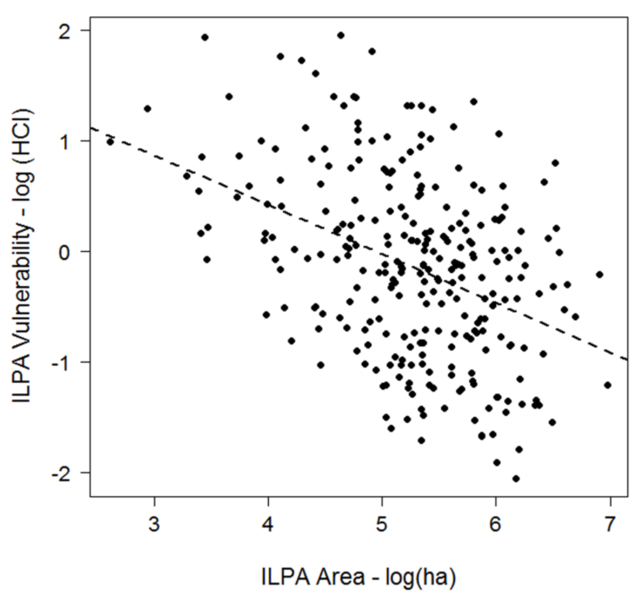

Figure 4.6: Relationship between the area of Indigenous Lands and Protected Areas (ILPAs) and the Hydrologic Connectivity Index (HCI). The HCI is a measure of protected area vulnerability to potential land use changes upstream.

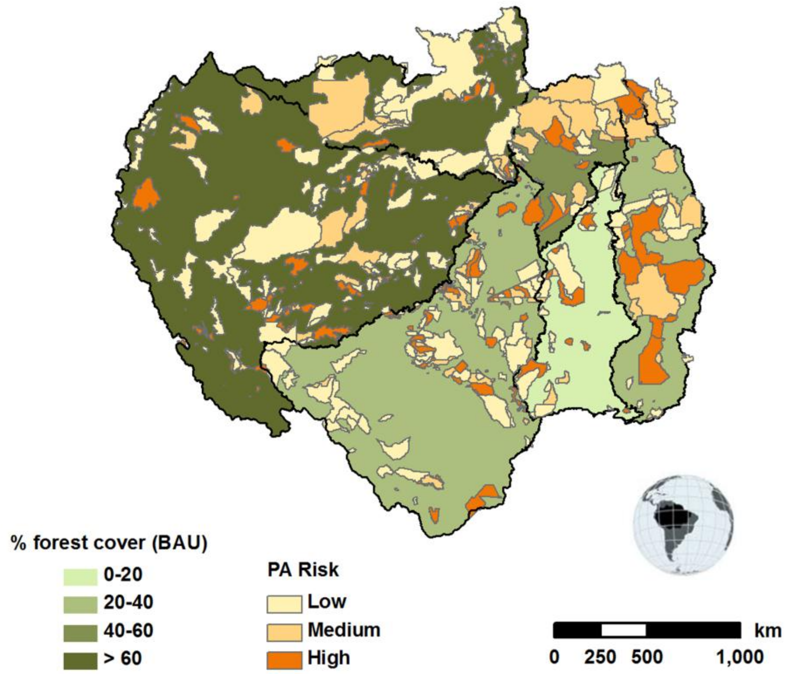

Figure 4.7: The Amazon network of indigenous lands and protected areas, categorized according to the hydrologic connectivity index. ILPAs with an HCI value greater than one are highly vulnerable to losses in hydrologic connectivity within the upstream zone of influence. Remote ILPAs in the western Amazon are unlikely to experience land-use related threats by 2050, particularly compared to ILPAs in the eastern Amazon.

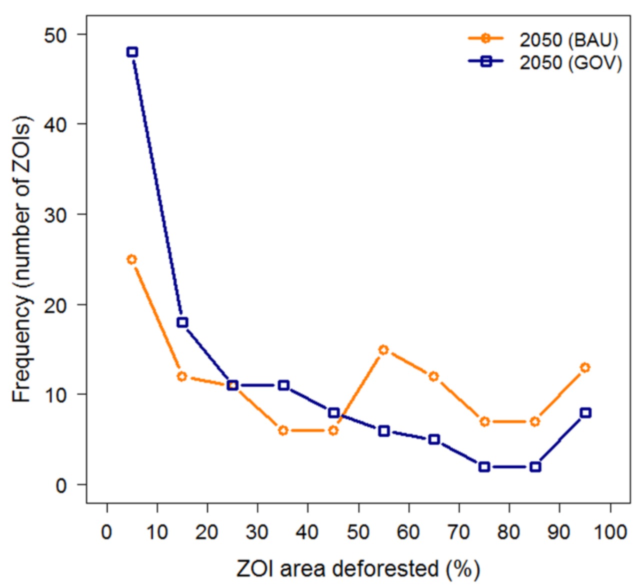

Figure 4.8: Predicted threat to ILPAs classified as high risk (HCI > 1) under BAU and GOV scenarios for 2050. The frequency distributions summarize predicted future deforestation levels (%) in the zones of influence of high-risk ILPAs.

References

Abell R., Allan J. D., Lehner B. (2007) Unlocking the potential of protected areas for freshwaters. *Biological Conservation*, **134**, 48-63.

Abell R., Thieme M. L., Revenga C., Bryer M., Kottelat M., Bogutskaya N., Coad B., Mandrak N. *et al.* (2008) Freshwater ecoregions of the world: A new map of biogeographic units for freshwater biodiversity conservation. *Bioscience*, **58**, 403-414.

Adeney J. M., Christensen N. L., Pimm S. L. (2009) Reserves protect against deforestation fires in the Amazon. *PLoS ONE*, **4**.

Allan J. D. (2004) Landscapes and riverscapes: The influence of land use on stream ecosystems. *Annual Review of Ecology Evolution and Systematics*, **35**, 257-284.

Amoros C., Bornette G. (2002) Connectivity and biocomplexity in waterbodies of riverine floodplains. *Freshwater Biology*, **47**, 761-776.

ANA (2010) National Hydrographic Division - Ottobacias. Available at http://www.ana.gov.br/bibliotecavirtual/solicitacaoBaseDados.asp. Accessed on June 15, 2010. Brazilian National Water Agency.

Barmuta L. A., Linke S., Turak E. (2011) Bridging the gap between 'planning' and 'doing' for biodiversity conservation in freshwaters. *Freshwater Biology*, **56**, 180-195.

Bruner A. G., Gullison R. E., Rice R. E., da Fonseca G. A. B. (2001) Effectiveness of parks in protecting tropical biodiversity. *Science*, **291**, 125-128.

Caissie D. (2006) The thermal regime of rivers: a review. *Freshwater Biology*, **51**, 1389-1406.

Calabrese J. M., Fagan W. F. (2004) A comparison-shopper's guide to connectivity metrics. *Frontiers in Ecology and the Environment*, **2**, 529-536.

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. *J Hydrology*, **369**, 165-174.

Coe M. T., Latrubesse E. M., Ferreira M. E., Amsler M. L. (2011) The effects of deforestation and climate variability on the streamflow of the Araguaia River, Brazil. *Biogeochemistry*, **105**, 119-131.

Cote D., Kehler D., Bourne C., Wiersma Y. (2009) A new measure of longitudinal connectivity for stream networks. *Landscape Ecology*, **24**, 101-113.

Da Silva R. R., Werth D., Avissar R. (2008) Regional impacts of future land-cover changes on the amazon basin wet-season climate. *Journal of Climate*, **21**, 1153-1170.

Deegan L., Neill C., Haupert C., Ballester M., Krusche A., Victoria R., Thomas S., de Moor E. (2011) Amazon deforestation alters small stream structure, nitrogen biogeochemistry and connectivity to larger rivers. *Biogeochemistry*, **105**, 53-74.

DeFries R., Hansen A., Newton A. C., Hansen M. C. (2005) Increasing isolation of protected areas in tropical forests over the past twenty years. *Ecological Applications*, **15**, 19-26.

DeFries R., Hansen A., Turner B. L., Reid R., Liu J. G. (2007) Land use change around protected areas: Management to balance human needs and ecological function. *Ecological Applications*, **17**, 1031-1038.

DeFries R., Karanth K. K., Pareeth S. (2010a) Interactions between protected areas and their surroundings in human-dominated tropical landscapes. *Biological Conservation*, **143**, 2870-2880.

DeFries R., Rovero F., Wright P., Ahumada J., Andelman S., Brandon K., Dempewolf J., Hansen A. *et al.* (2010b) From plot to landscape scale: linking tropical biodiversity measurements across spatial scales. *Frontiers in Ecology and the Environment*, **8**, 153-160.

Dudgeon D., Arthington A. H., Gessner M. O., Kawabata Z. I., Knowler D. J., Leveque C., Naiman R. J., Prieur-Richard A. H. *et al.* (2006) Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews*, **81**, 163-182.

Erős T., Schmera D., Schick R. S. (2011) Network thinking in riverscape conservation – A graph-based approach. *Biological Conservation*, **144**, 184-192.

Ewers R. M., Rodrigues A. S. L. (2008) Estimates of reserve effectiveness are confounded by leakage. *Trends in Ecology & Evolution*, **23**, 113-116.

Fagan W. F. (2002) Connectivity, fragmentation, and extinction risk in dendritic metapopulations. *Ecology*, **83**, 3243-3249.

Fagan W. F., Unmack P. J., Burgess C., Minckley W. L. (2002) Rarity, fragmentation, and extinction risk in desert fishes. *Ecology*, **83**, 3250-3256.

Fearnside P. M. (2009) Carbon benefits from Amazonian forest reserves: leakage accounting and the value of time. *Mitigation and Adaptation Strategies for Global Change*, **14**, 557-567.

Ferreira N., Ferreira L., Huete A., Ferreira M. (2007) An operational deforestation mapping system using MODIS data and spatial context analysis. *International Journal of Remote Sensing*, **28**, 47-62.

Freeman M. C., Pringle C. M., Jackson C. R. (2007) Hydrologic connectivity and the contribution of stream headwaters to ecological integrity at regional scales. *Journal of the American Water Resources Association*, **43**, 5-14.

FUNAI (2011) Indigenous lands of Brazil. Availabe at http://mapas.funai.gov.br/. Accessed on June 15, 2011. National Indian Foundation of Brazil.

Goulding M., Barthem R., Ferreira E. J. G. (2003) *The Smithsonian atlas of the Amazon*, Washington, DC, Smithsonian Institution Press.

Grant E. H. C., Lowe W. H., Fagan W. F. (2007) Living in the branches: population dynamics and ecological processes in dendritic networks. *Ecology Letters*, **10**, 165-175.

Greenwood M. J., Harding J. S., Niyogi D. K., McIntosh A. R. (2011) Improving the effectiveness of riparian management for aquatic invertebrates in a degraded agricultural landscape: stream size and land-use legacies. *Journal of Applied Ecology*, no-no.

Hansen A. J., Davis C. R., Piekielek N., Gross J., Theobald D. M., Goetz S., Melton F., DeFries R. (2011) Delineating the ecosystems containing protected areas for monitoring and management. *Bioscience*, **61**, 363-373.

Hansen A. J., DeFries R. (2007) Ecological mechanisms linking protected areas to surrounding lands. *Ecological Applications*, **17**, 974-988.

Hayhoe S. J., Neill C., Porder S., McHorney R., Lefebvre P., Coe M. T., Elsenbeer H., Krusche A. V. (2011) Conversion to soy on the Amazonian agricultural frontier increases streamflow without affecting stormflow dynamics. *Global Change Biology*, **17**, 1821-1833.

Herbert M. E., McIntyre P. B., Doran P. J., Allan J. D., Abell R. (2010) Terrestrial reserve networks do not adequately represent aquatic ecosystems. *Conservation Biology*, **24**, 1002-1011.

Hess G. R. (1996) Linking extinction to connectivity and habitat destruction in metapopulation models. *The American Naturalist*, **148**, 226-236.

Hess L. L., A.A. Affonso, C. Barbosa, M. Gastil-Buhl, J.M. Melack, Novo E. M. L. M. (2009) LBA-ECO LC-07 Basinwide Amazon Wetlands 100m Mask based on JERS SAR Images. Dataset. Available on-line [http://lba.cptec.inpe.br/] from LBA Data and Information System, National Institute for Space Research (INPE/CPTEC), Cachoeira Paulista, Sao Paulo, Brazil.

Johnson P. T. J., Olden J. D., Zanden M. J. V. (2008) Dam invaders: impoundments facilitate biological invasions into freshwaters. *Frontiers in Ecology and the Environment*, **6**, 359-365.

Joppa L. N., Loarie S. R., Pimm S. L. (2008) On the protection of "protected areas". *Proceedings of the National Academy of Sciences of the United States of America*, **105**, 6673-6678.

Junk W. J., Soares M. G. M., Bayley P. B. (2007) Freshwater fishes of the Amazon River basin: their biodiversity, fisheries, and habitats. *Aquatic Ecosystem Health & Management*, **10**, 153-173.

Lehner B., Verdin K., Jarvis A. (2006) HydroSHEDS Technical Documentation. Washington, DC. Available at http://hydrosheds.cr.usgs.gov., World Wildlife Fund US.

Lorion C. M., Kennedy B. P. (2009a) Relationships between deforestation, riparian forest buffers and benthic macroinvertebrates in neotropical headwater streams. *Freshwater Biology*, **54**, 165-180.

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

Macedo M., DeFries R., Morton D., Stickler C., Galford G., Shimabukuro Y. (2012) Decoupling of deforestation and soy production in the southern Amazon during the late 2000s. *PNAS,* **109**, 1341-1346.

Meyer J. L., Strayer D. L., Wallace J. B., Eggert S. L., Helfman G. S., Leonard N. E. (2007) The contribution of headwater streams to biodiversity in river networks. *Journal of the American Water Resources Association,* **43**, 86-103.

Morton D. C., DeFries R. S., Shimabukuro Y. E., Anderson L. O., Arai E., Espirito-Santo F. D., Freitas R., Morisette J. (2006) Cropland expansion changes deforestation dynamics in the southern Brazilian Amazon. *Proceedings of the National Academy of Sciences of the United States of America,* **103**, 14637-14641.

Nel J. L., Reyers B., Roux D. J., Impson N. D., Cowling R. M. (2011) Designing a conservation area network that supports the representation and persistence of freshwater biodiversity. *Freshwater Biology,* **56**, 106-124.

Nel J. L., Roux D. J., Abell R., Ashton P. J., Cowling R. M., Higgins J. V., Thieme M., Viers J. H. (2009) Progress and challenges in freshwater conservation planning. *Aquatic Conservation-Marine and Freshwater Ecosystems,* **19**, 474-485.

Nelson A., Chomitz K. M. (2011) Effectiveness of strict vs. multiple use protected areas in reducing tropical forest fires: A global analysis using matching methods. *PLoS ONE,* **6**.

Nepstad D., Schwartzman S., Bamberger B., Santilli M., Ray D., Schlesinger P., Lefebvre P., Alencar A. *et al.* (2006a) Inhibition of Amazon deforestation and fire by parks and indigenous lands. *Conservation Biology,* **20**, 65-73.

Nepstad D., Soares B. S., Merry F., Lima A., Moutinho P., Carter J., Bowman M., Cattaneo A. *et al.* (2009) The end of deforestation in the Brazilian Amazon. *Science,* **326**, 1350-1351.

Nepstad D. C., Stickler C. M., Almeida O. T. (2006b) Globalization of the Amazon soy and beef industries: Opportunities for conservation. *Conservation Biology,* **20**, 1595-1603.

Nepstad D. C., Stickler C. M., Soares B., Merry F. (2008) Interactions among Amazon land use, forests and climate: prospects for a near-term forest tipping point. *Philosophical Transactions of the Royal Society B-Biological Sciences,* **363**, 1737-1746.

Possingham H., Wilson K., Aldeman S., Vynne C. (2006) Protected areas: goals, limitations, and design. In: *Principles of conservation biology.* (eds Groom M, Meffe G, Carroll C) pp 509-551. Sunderland, Massachusetts, USA, Sinauer Associates.

Pringle C. (2003) What is hydrologic connectivity and why is it ecologically important? *Hydrological Processes,* **17**, 2685-2689.

Pringle C. M. (2001) Hydrologic connectivity and the management of biological reserves: A global perspective. *Ecological Applications,* **11**, 981-998.

Revenga C., Murray S., Abramovitz J., Hammond A. (1998) *Watersheds of the World: Ecological Value and Vulnerability.,* Washington, D.C., World Resources Institute.

Ricketts T. H., Soares-Filho B., da Fonseca G. A. B., Nepstad D., Pfaff A., Petsonk A., Anderson A., Boucher D. *et al.* (2010) Indigenous lands, protected areas, and slowing climate change. *PLoS Biology,* **8**.

Sano E. E., Rosa R., Brito J. L. S. B., Ferreira L. G. (2007) Mapeamento de cobertura vegetal do bioma Cerrado: estratégias e resultados. In: *Pesquisa Agropecuaria Brasileira.* pp 153-156, Planaltina, DF, Brazil, EMBRAPA (Brazilian Agricultural Research Corporation) - Cerrado division.

Schlosser I. J., Johnson J. D., Knotek W. L., Lapinska M. (2000) Climate variability and size-structured interactions among juvenile fish along a lake-stream gradient. *Ecology,* **81**, 1046-1057.

Schulman L., Ruokolainen K., Junikka L., Saaksjarvi I. E., Salo M., Juvonen S. K., Salo J., Higgins M. (2007) Amazonian biodiversity and protected areas: Do they meet? *Biodiversity and Conservation*, **16**, 3011-3051.

SEMA (2010) Digital map database - MT stream network (hidrografia): Available at http://basesig.hd1.com.br/. Accessed on June 15, 2010. Mato Grosso State Environmental Agency.

Soares B., Moutinho P., Nepstad D., Anderson A., Rodrigues H., Garcia R., Dietzsch L., Merry F. *et al.* (2010) Role of Brazilian Amazon protected areas in climate change mitigation. *Proceedings of the National Academy of Sciences of the United States of America,* **107**, 10821-10826.

Soares B. S., Nepstad D. C., Curran L. M., Cerqueira G. C., Garcia R. A., Ramos C. A., Voll E., McDonald A. *et al.* (2006) Modelling conservation in the Amazon basin. *Nature*, **440**, 520-523.

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

Thieme M., Lehner B., Abell R., Hamilton S. K., Kellndorfer J., Powell G., Riveros J. C. (2007) Freshwater conservation planning in data-poor areas: An example from a remote Amazonian basin (Madre de Dios River, Peru and Bolivia). *Biological Conservation,* **135**, 484-501.

Vorosmarty C. J., McIntyre P. B., Gessner M. O., Dudgeon D., Prusevich A., Green P., Glidden S., Bunn S. E. *et al.* (2010) Global threats to human water security and river biodiversity. *Nature*, **467**, 555-561.

Walker R., Moore N. J., Arima E., Perz S., Simmons C., Caldas M., Vergara D., Bohrer C. (2009) Protecting the Amazon with protected areas. *Proceedings of the National Academy of Sciences of the United States of America,* **106**, 10582-10586.

Ward J. V. (1989) The 4-Dimensional Nature of Lotic Ecosystems. *Journal of the North American Benthological Society,* **8**, 2-8.

Ward J. V., Tockner K., Arscott D. B., Claret C. (2002) Riverine landscape diversity. *Freshwater Biology,* **47**, 517-539.

Werth D., Avissar R. (2002) The local and global effects of Amazon deforestation. *J. Geophys. Res.,* **107**, 8087.

Wittemyer G., Elsen P., Bean W. T., Burton A. C. O., Brashares J. S. (2008) Accelerated human population growth at protected area edges. *Science*, **321**, 123-126.

Wright J. P., Flecker A. S. (2004) Deforesting the riverscape: the effects of wood on fish diversity in a Venezuelan piedmont stream. *Biological Conservation,* **120**, 439-447.

Chapter 5

Land-use change in the Amazon: A multiscale assessment of the challenges and opportunities for the management of freshwater ecosystems

Introduction

The forces driving changes in tropical forests – whether towards deforestation or forest conservation – operate across global, regional, and local scales. At the global scale, growing demand for agricultural commodities, such as soybeans, biofuels, and beef is a powerful driver of deforestation in tropical regions. At the same time, international efforts to reduce greenhouse gas (GHG) emissions and protect environmental services aim to create financial incentives and funding mechanisms for Reducing Emissions from Deforestation and Forest Degradation (REDD+), with the goal of conserving tropical forests. At the national scale, some policies fund large infrastructure projects that lead to deforestation, while others create extensive protected area systems to conserve biodiversity and protect the rights of forest-dependent peoples. Nowhere is this tug-of-war more evident today than in the Brazilian Amazon, at once a rapidly growing frontier of agriculture and infrastructure development and, in the last five years, a world leader in committing to and achieving GHG reductions through improved monitoring and decreased deforestation (Moutinho *et al.*, 2011, Nepstad *et al.*, 2009).

The net outcome of these complex and often opposing forces in the Amazon Basin has implications for freshwater ecosystems at multiple scales. The large-scale conversion of forests to croplands and pasture grasses alters surface roughness, albedo, and the partitioning between latent and sensible heat fluxes, with consequences for regional and even global hydro-climatic cycles (Jackson *et al.*, 2008). Research in Amazonia indicates that large-scale deforestation triggers significant decreases in regional evapotranspiration and precipitation, potentially altering atmospheric circulation and rainfall patterns in distant regions. Within the Amazon, deforestation may inhibit and redistribute rainfall (Medvigy *et al.*, 2011), increase surface temperatures (Loarie *et al.*, 2011), and alter stream flow (Coe *et al.*, 2011), all important factors that structure stream ecosystems. In agricultural landscapes, these hydrological changes may be further exacerbated by land use practices that lead to the degradation of riparian buffers, removal of watershed forest cover, installation of impoundments, soil compaction, and the use of chemical fertilizers and pesticides. This dissertation takes an incremental step towards understanding the multiscale causes and consequences of these changes for stream ecosystems by:

(1) Identifying the spatial-temporal dynamics of deforestation and subsequent land use transitions in the southern Amazon (Chapter 2);

(2) Analyzing the spatial distribution of impoundments, watershed forest cover, and riparian buffers (i.e., land management), and their implications for stream connectivity in the upper Xingu landscape (Chapter 3);

(3) Quantifying the impact of land management on stream temperature at the catchment scale (Chapter 3); and

(4) Assessing the vulnerability of the Amazon network of indigenous lands and protected areas to current and projected future deforestation in surrounding landscapes (Chapter 4).

This chapter summarizes the major findings, introduces some of the relevant policies and institutions, and discusses the management opportunities that exist at each scale of study.

Amazon Basin scale – Managing forest cover for multiple benefits

Despite recent reductions in deforestation, agricultural expansion in the Amazon's frontier has been and will likely continue to be the biggest driver of deforestation in the region (Chapter 2; Macedo *et al.*, 2012, Nepstad *et al.*, 2006b). While the Brazilian government's

National Policy for Climate Change has committed to an 80% decrease in deforestation by 2020, other national and agroindustry policies will make significant investments to increase agricultural production and the area under cultivation during the same time frame. The most plausible path to reconciling these two goals is a combination of policy incentives and enforcement mechanisms that direct the expansion of sugar cane, soybeans and other intensive crops onto the 72.6 million hectares (ha) of forest land that have already been cleared, of which 15.2 million ha have been abandoned (Moutinho *et al.*, 2011) and the remainder is used for extensive cattle ranching (Bowman *et al.*, 2012). Results from Chapter 2 provide evidence that this shift from cropland expansion into forests to expansion onto already cleared lands is not only possible, but has already occurred in the forested region of Mato Grosso during the period from 2000 to 2010 (Chapter 2; Macedo *et al.*, 2012). Maintaining these gains while preventing indirect land use changes associated with the displacement of cattle ranching (Arima *et al.*, 2011) to other regions will require investments to modernize and intensify the cattle sector, coupled with redoubled efforts to monitor and enforce anti-deforestation policies as local profits and commodity prices continue to climb (Angelsen, 2010, Bowman *et al.*, 2012).

The Amazon network of indigenous lands and protected areas (ILPA) is the cornerstone of forest conservation in the region, protecting 54% of remaining forests in the Brazilian Amazon today (Soares *et al.*, 2010). The ILPA network has proven effective in containing deforestation and reducing anthropogenic fires, with clear benefits for the conservation of forests, their carbon stocks, and the many socio-ecological functions that they support (Nepstad *et al.*, 2006a, Ricketts *et al.*, 2010, Soares *et al.*, 2010). These same ILPAs could be more effectively managed for the benefit of freshwater resources by evaluating upstream land-use threats (i.e., agricultural expansion) and taking actions to mitigate their impacts within protected areas (i.e., conservation or restoration of riparian buffers). Chapter 4 provides a simple framework for examining the hydrological context of protected areas and evaluating their vulnerability to potential losses in hydrologic connectivity due to deforestation upstream (Pringle, 2001). By using existing datasets and modeled scenarios of future development (Soares *et al.*, 2006), it provides a practical method for rapid assessment of the ILPA network, allowing for periodic reevaluation of risks (i.e., deforestation, infrastructure projects) and the strategic allocation of resources to proactively manage these threats. Mitigation activities might include development of watershed land use plans, building the capacity of landowners to implement best practices on their properties, and the development of mechanisms to compensate landowners for avoided deforestation. In the near term, some of these mitigation efforts could be supported by existing funding mechanisms, including the Amazon Region Protected Area Program (ARPA) and the Amazon Fund (Moutinho *et al.*, 2011). Potential future mechanisms under the U.N. Framework Convention on Climate Change (UNFCCC) REDD+ program could bring such efforts to scale, offering unprecedented opportunities for forest conservation on private and public lands in the Amazon, as well as tremendous challenges for monitoring, reporting, and verification (Moutinho *et al.*, 2011, Nepstad *et al.*, 2009, Stickler *et al.*, 2009).

Xingu Basin scale – Mitigating the impacts of agricultural expansion

While the national and international policies mentioned above aim to change forest outcomes at the Amazon scale, their success or failure may ultimately be determined at the landscape scale. The long-term effectiveness of top-down rules for resource management may be contingent on the development of nested governance structures that operate at multiple scales and can account for the realities of implementation on the ground (Chapter 1; Brondizio *et al.*, 2009, Ostrom *et al.*, 1999). The case of the Xingu Indigenous Park (PIX), located in the heart of the Amazon's agricultural frontier, illustrates the substantial challenges associated with achieving this multiscale coordination on the ground (Brondizio *et al.*, 2009, Stickler, 2009, Stickler *et al.*, 2009). Created in 1964, the park is designated for the subsistence of indigenous communities and conservation of the forests and freshwater resources they depend on. Despite effectively conserving forest cover within its borders, the 2.6 million ha reserve drains an area more than four times its size (Chapter 4), which today is increasingly dominated by cattle ranching and industrial soybean production (Chapter 2). Chapter 3 indicates that the removal of riparian forest buffers and installation of over 7,500 upstream impoundments (1 per 7 km of stream length) have had a large cumulative impact on headwater stream temperature and connectivity at the landscape scale, potentially compromising freshwater resources within the PIX.

Results from this dissertation underscore the importance of managing agricultural landscapes to mitigate the negative impacts (e.g., increased temperature) of production on stream ecosystems. Specifically, they confirm the importance of conserving riparian buffers in agricultural watersheds and highlight impoundments as a widespread – and currently unregulated – threat to hydrologic connectivity. The Brazilian Forest Code is the central piece of legislation governing the conservation and use of forests on rural properties in the Amazon forest biome. It not only mandates the conservation of riparian buffers, but also requires landowners to protect forests on 80% of their property. If fully implemented and enforced, the current Forest Code would facilitate coordination at the landscape scale and mitigate many of the potential impacts of agricultural development on freshwater ecosystems. The case of the Xingu headwaters illustrates the many barriers to securing this outcome. Challenges include unclear land tenure, limited capacity for enforcement, corruption within state and local government agencies, and inefficient collection of fines even when they are levied (Arima *et al.*, 2005, Azevedo, 2009). In recent years, the state and federal governments have made much progress in improving enforcement by clarifying land tenure, developing satellite-based monitoring and environmental licensing at the property level, and prosecuting corrupt officials at several levels of government (Azevedo, 2009). While these actions appear to have had an impact in deterring deforestation (Chapter 2), legislation currently under debate in the Brazilian Congress threatens to weaken the Forest Code by reducing riparian buffer requirements, providing amnesty for many producers who are not in compliance, and eliminating federal powers to prosecute environmental crimes at the state level.

Microbasin scale – Managing rural properties

Ultimately, the decisions that determine the distribution and configuration of forest cover in the landscape – and their influence on freshwater ecosystems – are made at the scale of individual properties. At this level, land use decisions are governed by very practical considerations, including (but not limited to) fluctuations in commodity prices, the monetary costs of adopting best practices, knowledge of the legal requirements, and the perceived costs and benefits of compliance. For example, results from Chapter 3 indicate that intact riparian buffers effectively regulate stream temperature by shading small streams. While this supports legal requirements of riparian buffers as a strategy for mitigating the impacts of agriculture, it says nothing of the logistical difficulty of making it happen on the ground.

Many cattle ranches have large tracts of degraded riparian areas, which have been trampled by cattle and invaded by non-native pasture grasses that out-compete tree seedlings, increase the likelihood of escaped fires, and make restoration very difficult. In this context, land managers, who are legally required to restore their riparian areas, may face several challenges. First, there is generally no dependable supply of native seedlings in the region, much less at the scale needed for restoration. Second, the technical capacity for successful restoration of these landscapes is still being developed. Even where landowners endeavor to grow their own seedlings, they have to find native seeds, test methods to get them to germinate, and develop techniques to manage pasture grasses, which otherwise prevent seedling establishment. Finally, the costs of implementing recommended management is often prohibitive. One option for restoring these landscapes is to provide conditions that allow them to recover on their own. For example, fencing livestock out of riparian areas is a direct way to reduce degradation, with immediate benefits for water quality (e.g., reduced sedimentation). For some cattle ranchers, even this option may be too expensive, as it requires substantial investment in fencing materials.

Despite the challenges mentioned above, many opportunities exist for improved governance and coordination of landowners on the ground. In the early 2000s, escalating deforestation in Mato Grosso prompted the development of several grassroots efforts aimed at facilitating change at the property level. Indigenous groups within the PIX have organized into an association (ATIX) to advocate for their rights; municipal programs and environmental NGOs have created native seed banks and worked with landowners to develop locally appropriate techniques for riparian restoration; and state institutions have developed registries that require documentation of property boundaries and development of management plans to bring them into compliance. These actions have been supported by restrictions on credit for illegal deforesters and industry-led moratoria focused on excluding soy and beef from the supply chains of major exporters. One of the most promising initiatives is the development of a voluntary land registry (Cadastro de Compromisso Ambiental – CCS), which establishes guidelines for socio-environmental responsibility at the farm scale. The initial effort has been supported by non-profit organizations (Aliança da Terra and the Amazon Environmental Research Institute) that provide extension services to help producers identify and map environmental problems on their properties; develop targets for improving their performance; and audit their progress towards those targets. Today, the registry has hundreds of registered farms and is working towards developing a market-based certification scheme to provide a financial incentive to producers.

Governance – Challenges and opportunities for achieving cross-scale coordination

Evidence from around the world suggests that an important characteristic of lasting management systems is that local forest users gain participation in rulemaking and forest management (Gibson *et al.*, 2005, Persha *et al.*, 2011). In this regard, the Amazon's agricultural frontier poses both challenges and opportunities. A major challenge is the fact that landowners and land managers are generally newcomers to the region and, thus, may have very different perceptions of the value of forests and freshwater resources than do the indigenous groups who have used them for centuries (Brondizio *et al.*, 2009, Cash *et al.*, 2006). Furthermore, while management actions are executed at the farm level, they are often motivated by decisions at other scales and in other regions. A typical large landowner may live in a distant urban area and make decisions based on a complex set of factors, including global commodities markets, land prices, the availability of credit, and evolving perceptions about the relative risks and rewards of land use decisions. This mismatch between the scale of decision-making and the scale of management on the ground can lead to scenarios like the case of the PIX, where distant agribusiness interests benefit from land use decisions while local indigenous populations bear most of the costs.

Results from this dissertation indicate that the land use choices made locally can have a large cumulative effect on freshwater ecosystems at the landscape scale, with downstream consequences that extend well beyond individual property boundaries. Managing these complex, cross-scale interactions will require the development of equally complex management systems that can operate at multiple levels and communicate across scales (see Chapter 1; Brondizio *et al.*, 2009, Ostrom, 2009). As noted here, many of the institutional building blocks are already in place at each scale. The long-term management of Amazon forests and the connectivity of freshwater ecosystems they protect will likely depend on finding creative new ways to link these institutions and improve their effectiveness in an increasingly complex world.

Summary and next steps

This dissertation lends new insights into the multiscale consequences of agricultural expansion for tropical stream ecosystems and leads to the following major conclusions:

(1) Given the large supply of degraded pasture lands in the Amazon Basin, an opportunity exists to conserve forests while increasing agricultural expansion. Achieving this is contingent on developing effective policies that both contain deforestation and encourage more efficient use of already cleared lands. Results from Chapter 2 provide preliminary evidence that this transition is possible, but maintaining these gains will require investments to intensify the cattle sector.

(2) In the Xingu Basin, large-scale agricultural expansion has decreased stream connectivity through the degradation of riparian buffers and the widespread installation of farm impoundments in the landscape. Results from Chapter 3 indicate that impoundments are a legacy of the region's history of cattle ranching and provide the first documentation of this widespread (and unregulated) phenomenon in the landscape.

(3) At the catchment scale, land management has a direct impact on the temperature of headwater streams. Results from Chapter 3 indicate that the removal of riparian forest buffers, installation of impoundments, and large-scale removal of watershed vegetation significantly increase stream temperature. Given how widespread these factors are in the Xingu landscape, they likely have a large cumulative impact on the stream network including downstream indigenous lands.

(4) The Amazon network of indigenous lands and protected areas is already serving an important function by conserving standing forests and the freshwater ecosystems they support. Results from Chapter 4 indicate that at least 30% of these areas are vulnerable to current or future deforestation within their watersheds. Maximizing the potential of these areas to conserve freshwater resources will be contingent on managing future anthropogenic threats in surrounding landscapes.

Future research will focus on understanding the biotic implications of stream degradation and fragmentation. Specifically, I am interested in examining the influence of water quality (e.g., stream temperature, turbidity, and dissolved oxygen, among others) and stream fragmentation (e.g., impoundments, habitat degradation) on large-bodied fish species. Furthermore, I am interested in refining our understanding of the thresholds beyond which riparian degradation, watershed deforestation, and impoundments have a measurable impact on streams. These are critical next steps towards developing appropriate management criteria for mitigating the impacts of agricultural production on freshwater ecosystems and the people that depend on them.

References

Angelsen A. (2010) Policies for reduced deforestation and their impact on agricultural production. *Proceedings of the National Academy of Sciences of the United States of America,* **107**, 19639-19644.

Arima E., Barreto P., Brito M. (2005) *Pecuária na Amazônia: Tendências e implicações para a conservação ambiental,* Belém, Pará, Instituto do Homem e Meio Ambiente da Amazônia.

Arima E. Y., Richards P., Walker R., Caldas M. M. (2011) Statistical confirmation of indirect land use change in the Brazilian Amazon. *Environmental Research Letters,* **6**, 024010.

Azevedo A. A. (2009) Legitimating Unsustainability? Analysis of the Environmental Licensing System for Rural Properties - SLAPR (Mato Grosso). Ph.D. in Sustainable Development, Center for Sustainable Development (CDS). University of Brasília, Brasília, Brasil, 325 pp.

Bowman M. S., Soares-Filho B. S., Merry F. D., Nepstad D. C., Rodrigues H., Almeida O. T. (2012) Persistence of cattle ranching in the Brazilian Amazon: A spatial analysis of the rationale for beef production. *Land Use Policy,* **29**, 558-568.

Brondizio E. S., Ostrom E., Young O. R. (2009) Connectivity and the governance of multilevel social-ecological systems: The role of social capital. *Annual Review of Environment and Resources,* **34**, 253-278.

Cash D. W., Adger W. N., Berkes F., Garden P., Lebel L., Olsson P., Pritchard L., Young O. (2006) Scale and cross-scale dynamics: Governance and information in a multilevel world. *Ecology and Society,* **11**.

Coe M. T., Latrubesse E. M., Ferreira M. E., Amsler M. L. (2011) The effects of deforestation and climate variability on the streamflow of the Araguaia River, Brazil. *Biogeochemistry,* **105**, 119-131.

Gibson C. C., Williams J. T., Ostrom E. (2005) Local Enforcement and Better Forests. *World Development,* **33**, 273-284.

Jackson R. B., Randerson J. T., Canadell J. G., Anderson R. G., Avissar R., Baldocchi D. D., Bonan G. B., Caldeira K. *et al.* (2008) Protecting climate with forests. *Environmental Research Letters,* **3**.

Loarie S. R., Lobell D. B., Asner G. P., Mu Q. Z., Field C. B. (2011) Direct impacts on local climate of sugar-cane expansion in Brazil. *Nature Climate Change,* **1**, 105-109.

Macedo M., DeFries R., Morton D., Stickler C., Galford G., Shimabukuro Y. (2012) Decoupling of deforestation and soy production in the southern Amazon during the late 2000s. *PNAS,* **109**, 1341-1346.

Medvigy D., Walko R. L., Avissar R. (2011) Effects of Deforestation on Spatiotemporal Distributions of Precipitation in South America. *Journal of Climate,* **24**, 2147-2163.

Moutinho P., Martings O. S., Christovam M., Lima A., Nepstad D., Crisostomo A. C. (2011) The emerging REDD+ regime of Brazil. *Carbon Management,* **2**.

Nepstad D., Schwartzman S., Bamberger B., Santilli M., Ray D., Schlesinger P., Lefebvre P., Alencar A. *et al.* (2006a) Inhibition of Amazon deforestation and fire by parks and indigenous lands. *Conservation Biology,* **20**, 65-73.

Nepstad D., Soares B. S., Merry F., Lima A., Moutinho P., Carter J., Bowman M., Cattaneo A. *et al.* (2009) The end of deforestation in the Brazilian Amazon. *Science,* **326**, 1350-1351.

Nepstad D. C., Stickler C. M., Almeida O. T. (2006b) Globalization of the Amazon soy and beef industries: Opportunities for conservation. *Conservation Biology,* **20**, 1595-1603.

Ostrom E. (2009) A General Framework for Analyzing Sustainability of Social-Ecological Systems. *Science,* **325**, 419-422.

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

Persha L., Agrawal A., Chhatre A. (2011) Social and Ecological Synergy: Local Rulemaking, Forest Livelihoods, and Biodiversity Conservation. *Science,* **331**, 1606-1608.

Pringle C. M. (2001) Hydrologic connectivity and the management of biological reserves: A global perspective. *Ecological Applications*, **11**, 981-998.

Ricketts T. H., Soares-Filho B., da Fonseca G. A. B., Nepstad D., Pfaff A., Petsonk A., Anderson A., Boucher D. *et al.* (2010) Indigenous lands, protected areas, and slowing climate change. *PLoS Biology*, **8**.

Soares B., Moutinho P., Nepstad D., Anderson A., Rodrigues H., Garcia R., Dietzsch L., Merry F. *et al.* (2010) Role of Brazilian Amazon protected areas in climate change mitigation. *Proceedings of the National Academy of Sciences of the United States of America*, **107**, 10821-10826.

Soares B. S., Nepstad D. C., Curran L. M., Cerqueira G. C., Garcia R. A., Ramos C. A., Voll E., McDonald A. *et al.* (2006) Modelling conservation in the Amazon basin. *Nature*, **440**, 520-523.

Stickler C. M. (2009) Defending public interests in private forests: land use policy alternatives for the Xingu River headwaters region of southeastern Amazônia. Ph.D., Geography. University of Florida, Gainesville, FL, 199 pp.

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

Appendix A

Supplemental Figures and Tables for Chapter 2 – Decoupling of deforestation and soy production in the southern Amazon during the late 2000s

Table A.1: Validation of decision tree using field data collected in July and August of 2010

	Forest	Pasture	Cropland	Total	User's accuracy
Forest	61	2	2	65	93.85%
Pasture	5	145	5	155	93.55%
Cropland	2	7	73	82	89.02%
Total	68	154	80	302	
Producer's accuracy	89.71%	94.16%	91.25%		
Overall accuracy:	92.38%				
Cohen's kappa:	0.8767				

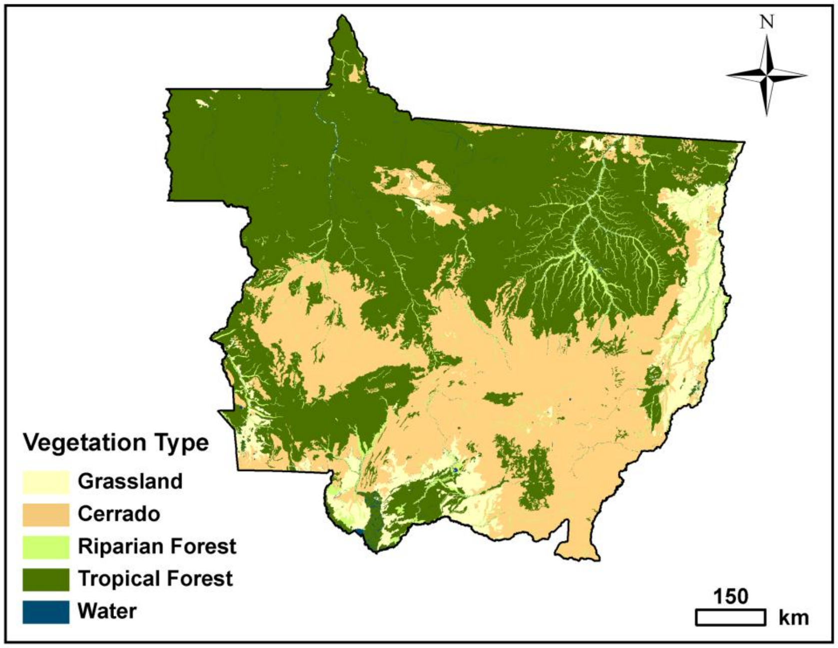

Figure A.1: Potential vegetation in the state of Mato Grosso (MT). The state is divided between Cerrado (savannah woodlands and grasslands) and Amazon (tropical forest) ecosystems (Mello, 2007).

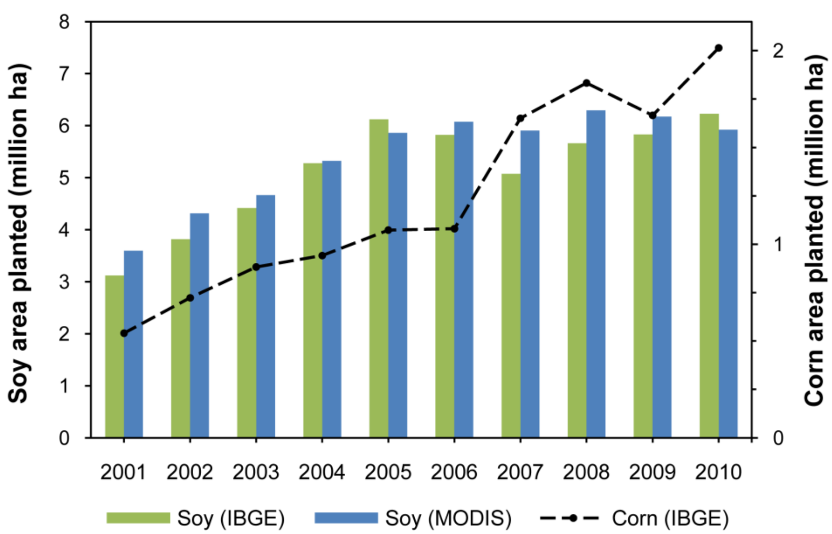

Figure A.2: Area planted in soy in Mato Grosso (bars) from Moderate Resolution Imaging Spectroradiometer (MODIS)-based estimates in this study and Brazilian government data (IBGE, 2011). The datasets show relatively good agreement (r = 0.94, R^2 = 0.88, RMSE=0.44). Although soy is the most prevalent cash crop in the state, secondary row crops such as corn (dashed line) are also economically important (IBGE, 2011).

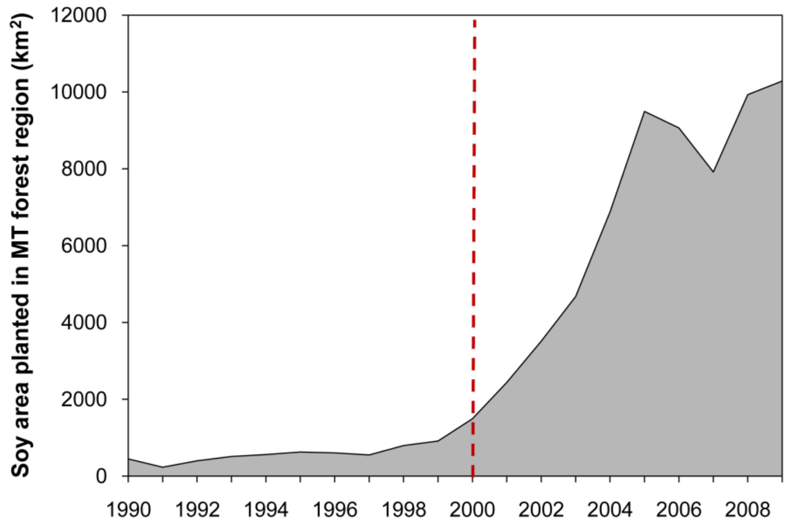

Figure A.3: Soybean area planted in Mato Grosso's forested municipalities from 1990-2008 (IBGE, 2011). Little mechanized soy production existed in the forested region of the state prior to 2000 (red dashed line).

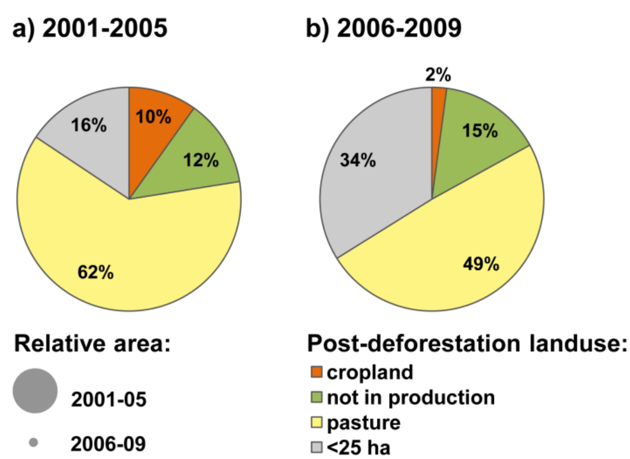

Figure A.4: Postdeforestation land uses in Mato Grosso for large-scale (> 25 ha) deforestation during the periods from 2001 to 2005 (a) and 2006 to 2009 (b). Total deforestation during the two time periods was 1.2 million ha and 0.23 million ha, respectively. Total soy production in forested municipalities was 16.5 million tons and 20.3 million tons, respectively (IBGE, 2011). Data were derived from the PRODES dataset (INPE, 2011) and the MODIS enhanced vegetation index (EVI) time series.

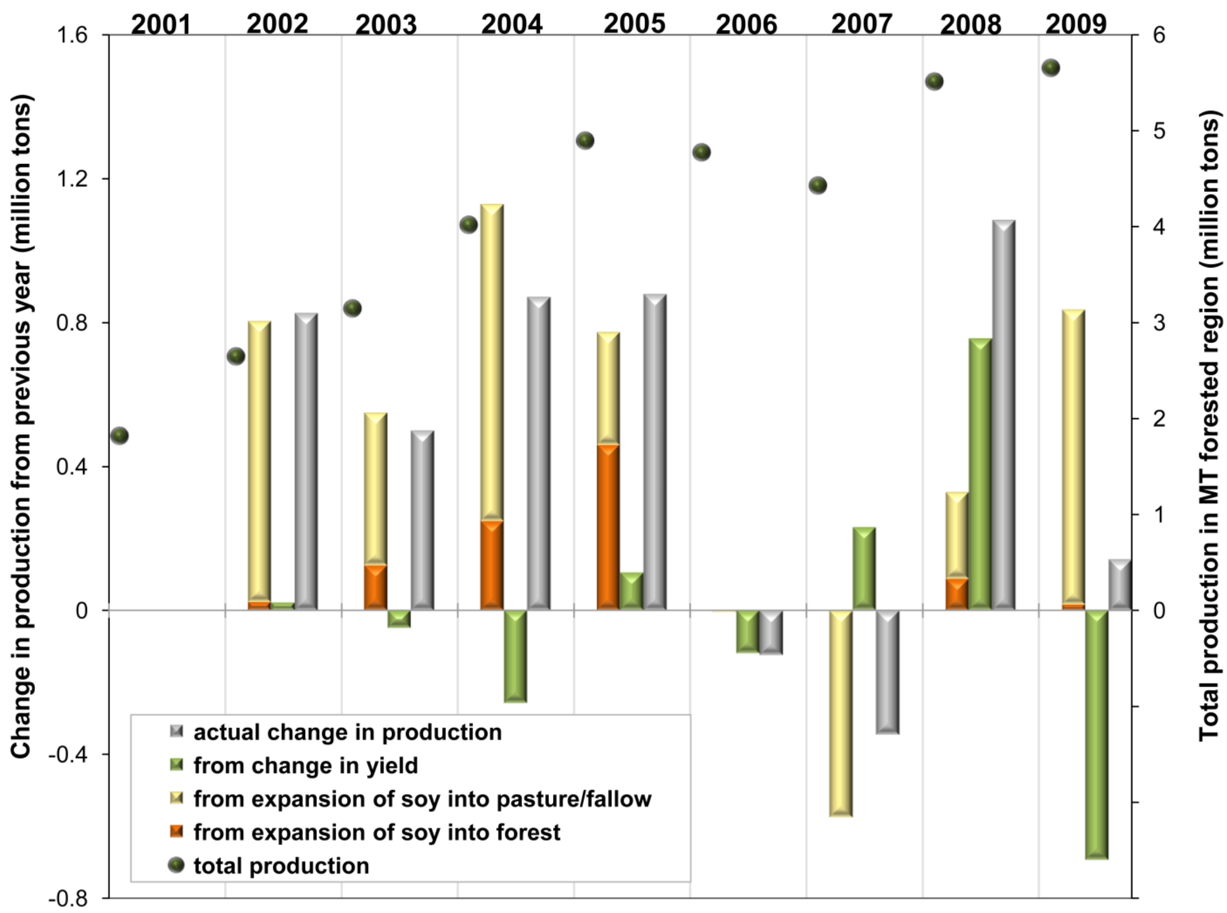

Figure A.5: Allocation of annual changes in soy production to yield, expansion into forest, and expansion into already-cleared land in the forested region of Mato Grosso. Production and area data from the IBGE (IBGE, 2011) were allocated to the forested region of the state using the MODIS time series.

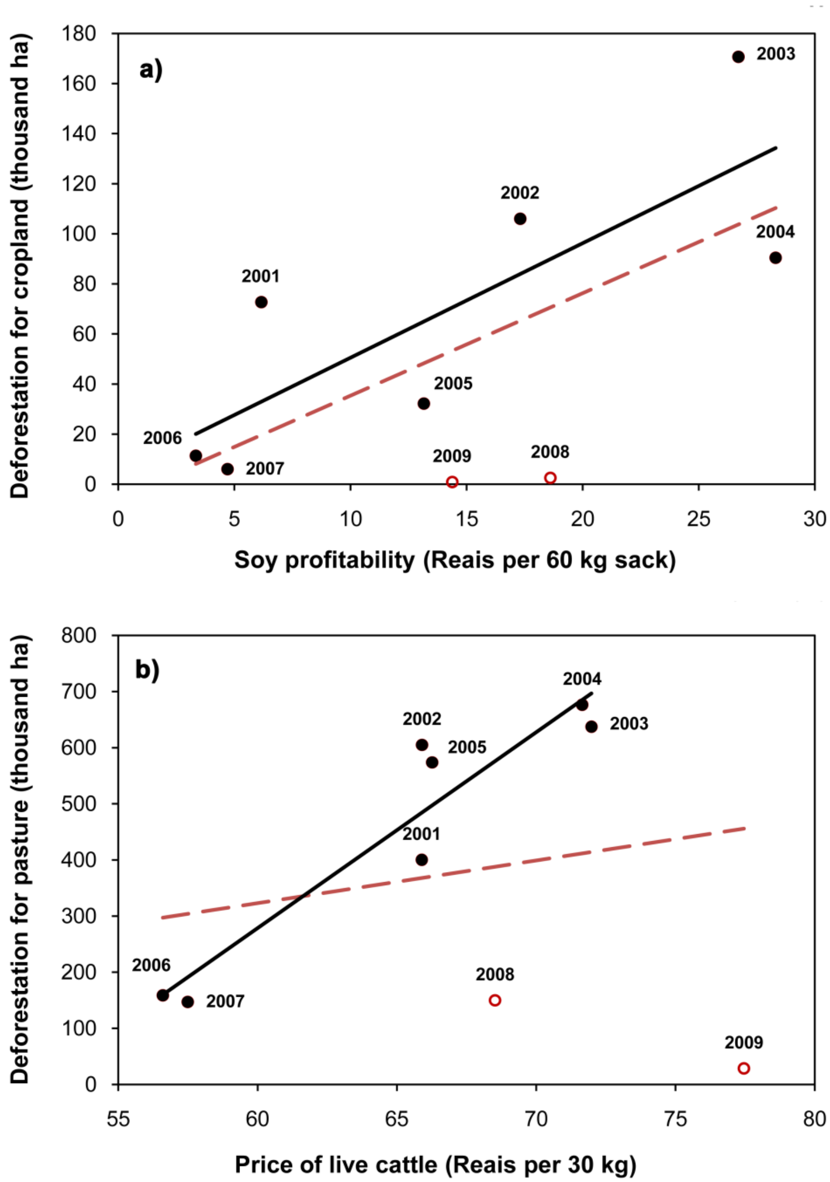

Figure A.6: Relationship between market indicators and deforestation for agriculture in Mato Grosso. (a) Correlation between profitability (CONAB, 2011, FGV, 2011b) and deforestation for cropland from 2001 to 2009 (red dashed line; R^2=0.39, n=9) and 2001 to 2007 only (black solid line; R^2=0.64, n=7). (b) Correlation between the farm gate price of cattle (FGV, 2011a) and deforestation for pasture from 2001 to 2009 (red dashed line; R^2=0.04, n=9) and 2001 to 2007 only (black solid line; R^2 = 0.89, n=7).

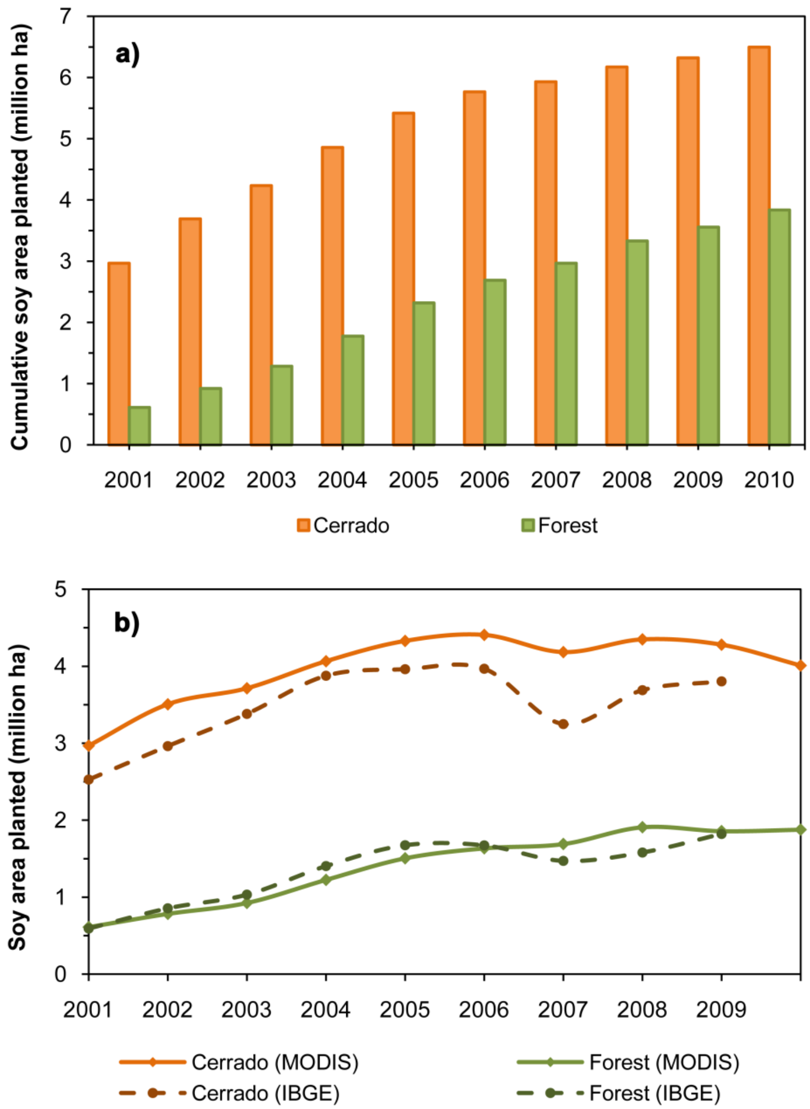

Figure A.7: Soy area planted in Mato Grosso's Cerrado (savannah woodlands and grasslands) and Amazon (tropical forest) biomes (Fig. A.1). Cumulative area planted (a) is derived from the MODIS analysis, whereas the annual area planted (b) compares IBGE municipal data (IBGE, 2011) and MODIS-based results.

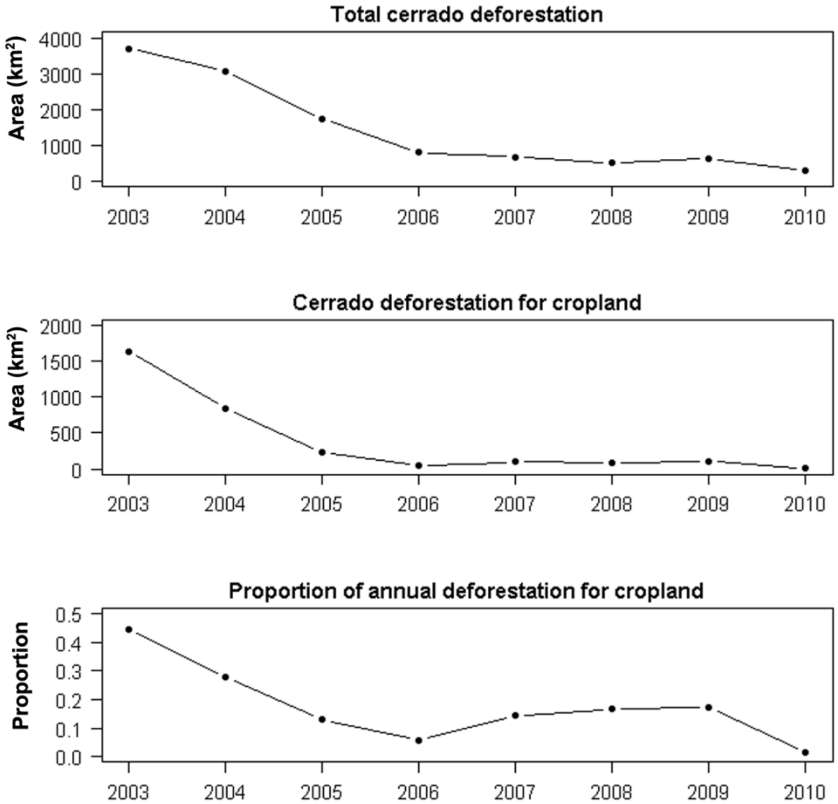

Figure A.8: Cerrado clearings for cropland in Mato Grosso from 2003 to 2010, based on published deforestation polygons (Ferreira *et al.*, 2007) and MODIS-based land use classifications. Both total cerrado deforestation and deforestation for cropland decreased in the cerrado region during this period.

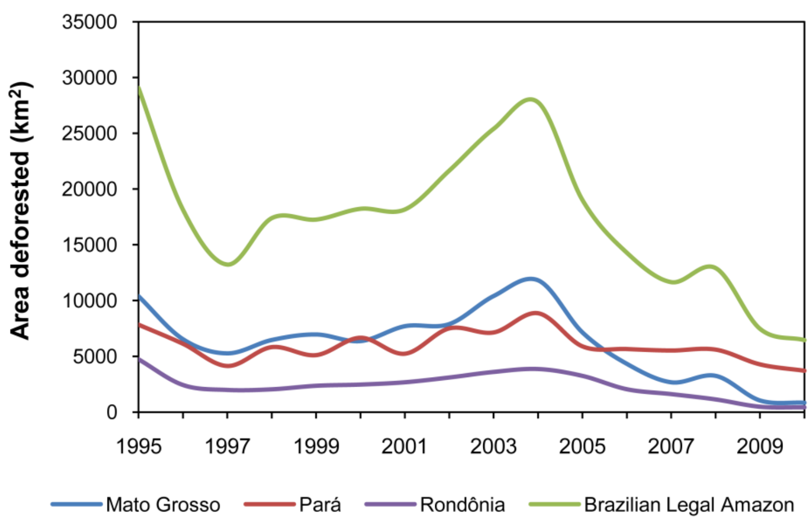

Figure A.9: Annual deforestation in the Brazilian Legal Amazon from 1995 to 2010 (INPE, 2011). The states of Pará, Rondônia, and Mato Grosso accounted for the majority of deforestation during this time period.

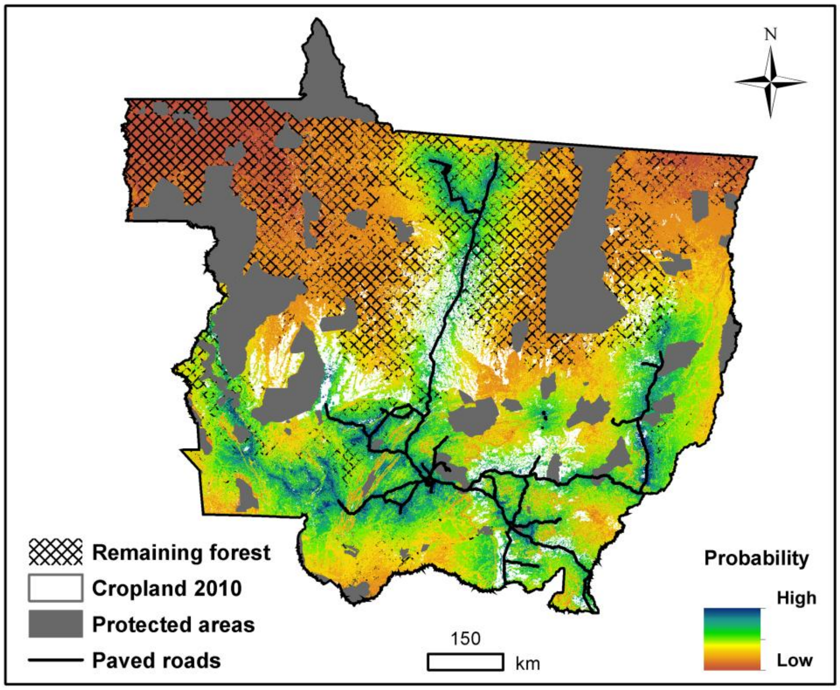

Figure A.10: Relative probability of conversion to cropland. The probability is determined by slope, climate, soil properties, road infrastructure, and other factors (Jasinski *et al.*, 2005). State and federal protected areas (gray) are masked out and remaining forest areas are cross-hatched.

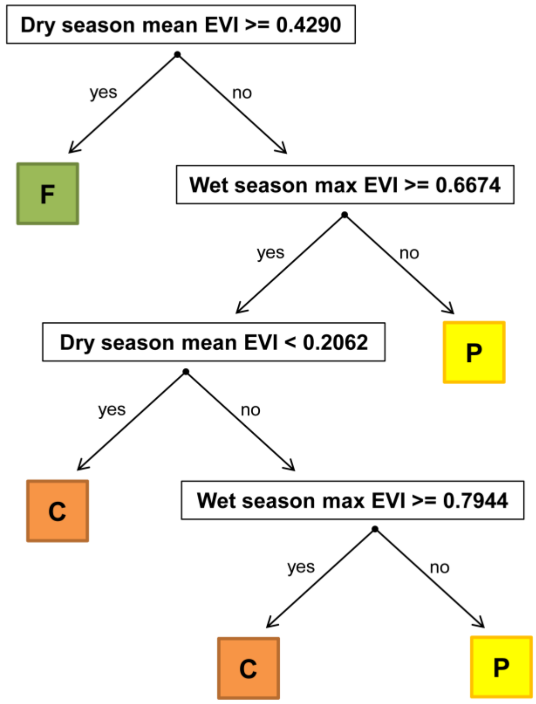

Figure A.11: Decision tree classifier based on the MODIS EVI. The resulting land use classes are cropland (C), pasture/cerrado (P), and forest (F). Thresholds were determined using field training data collected in July and August of 2006.

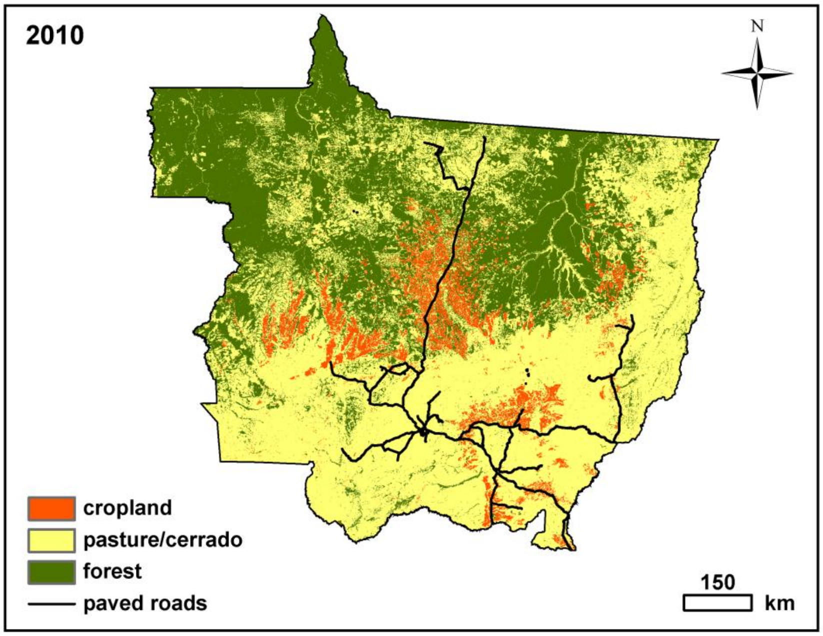

Figure A.12: Classification output for Mato Grosso in 2010. The resulting land use classes are cropland (C), pasture/cerrado (P), and forest (F). The decision tree was trained using 2006 field data. Similar maps were derived for each year from 2001 to 2010.

References

CONAB (2011) Cost of production of summer crops - time series. Available at http://www.conab.gov.br. Accessed on March 15, 2011., Companhia Nacional do Abastecimento.

Ferreira N., Ferreira L., Huete A., Ferreira M. (2007) An operational deforestation mapping system using MODIS data and spatial context analysis. *International Journal of Remote Sensing*, 28, 47-62.

FGV (2011a) Index of Prices Received (IPR) for Cattle - Mato Grosso Getúlio Vargas Foundation - Brazilian Institute of Economics.

FGV (2011b) Index of Soy Prices Received (IPR) - Mato Grosso Getúlio Vargas Foundation - Brazilian Institute of Economics.

IBGE (2011) Municipal Agricultural Production. Available at http://seriesestatisticas.ibge.gov.br/. Accessed on March 1, 2011. Brazilian Institute of Geography and Statistics.

INPE (2011) Program for the Estimation of Amazon Deforestation (Projeto PRODES Digital). Available at http://www.dpi.inpe.br/prodesdigital/prodes.php. Accessed on January 20, 2011. Brazilian National Agency for Space Research.

Jasinski E., Morton D., DeFries R., Shimabukuro Y., Anderson L., Hansen M. (2005) Physical landscape correlates of the expansion of mechanized agriculture in Mato Grosso, Brazil. *Earth Interactions*, 9.

Mello F. (2007) Estimates of soil carbon stocks in the states of Rondônia and Mato Grosso before human interventions (Portuguese). MS thesis Escola Superior de Agricultura “Luiz de Queiroz”, Universidade de São Paulo, Piracicaba, SP, Brazil, 89 pp.

Appendix B

Supplemental Figures and Tables for Chapter 3 – In hot water: The influence of agricultural land management on headwater stream temperature in the southern Amazon

Supplemental Text

Landsat preprocessing

We acquired 12 Landsat-5 TM (L1G) scenes (bands 1-5) from the Brazilian National Institute for Space Research (INPE) with acquisition dates ranging from July 14 to August 08, 2009. To minimize cloud cover, only dry season images were obtained. Geometric rectification was performed using nearest neighbor resampling to co-register each scene to its orthorectified GeoCover analogue obtained from the Global Land Cover Facility (GLCF), using a combination of manual and automated registration methods (Walker *et al.*, 2010). Co-registration was achieved with a root mean square (RMS) error of less than 0.5 pixels. Radiometric calibration and atmospheric correction were performed on each scene using ENVI 4.8. Scenes from the same date and path were then mosaicked together. Because seam lines were still visible across dates, we used the Iteratively Reweighted Multivariate Automated Detection (IR-MAD) algorithm (Canty & Nielsen, 2008) to normalize across dates before creating the final image mosaic.

Table B.1: Accuracy assessment of land use, land cover, and impoundment classification.

MODIS Classification - Land Use (independent validation with ground data)						
	Forest	Pasture	Cropland		Total	Error (Commission)
Forest	61	2	2		65	0.062
Pasture	5	145	5		155	0.065
Cropland	2	7	73		82	0.11
Total	68	154	80			
Error (Omission)	0.102	0.058	0.0875			
Overall kappa	0.877					
Overall accuracy	92%					
Landsat Classification - Riparian Forests (bootstrapped error estimate)						
	Agriculture	Forest	Water	Wetland	Total	Error (Commission)
Agriculture	228	5	0	1	234	0.026
Forest	3	65	0	0	68	0.044
Water	0	0	85	2	87	0.023
Wetland	1	0	1	23	25	0.08
Total	232	70	86	26		
Error (Omission)	0.017	0.071	0.012	0.115		
Overall kappa	0.948					
Overall accuracy	97%					
ASTER Classification - Impoundments (bootstrapped error estimate)						
	Impoundment	Other			Total	Error (Commission)
Impoundment	109	2			111	0.018
Other	2	289			291	0.007
Total	111	291				
Error (Omission)	0.018	0.007				
Overall kappa	0.975					
Overall accuracy	99%					

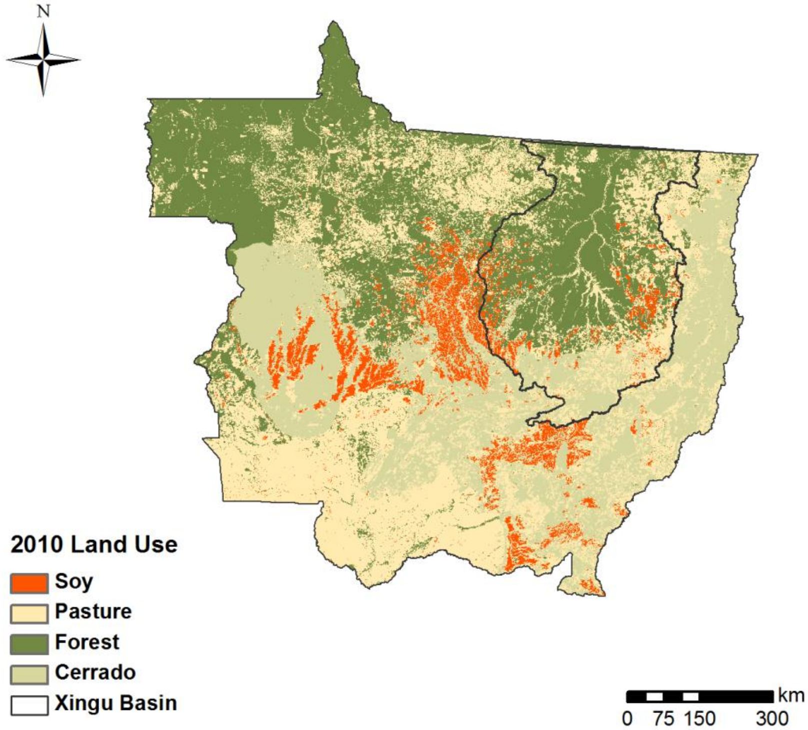

Figure B.1: Land use in Mato Grosso, Brazil in 2010. The land use/cover classification was compiled from existing MODIS-based datasets for the forest and cerrado biomes (Ferreira *et al.*, 2007, Macedo *et al.*, 2012).

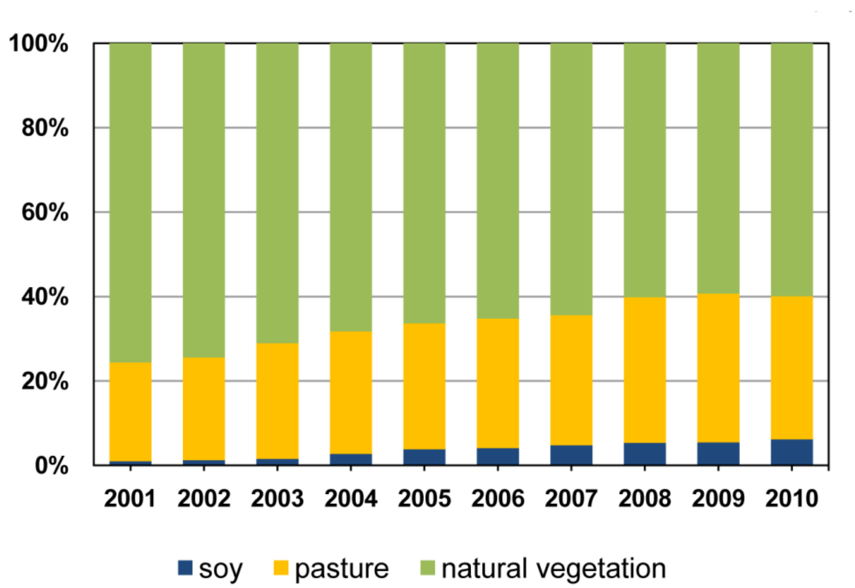

Figure B.2: Mean proportion of catchments outside protected areas in each land use. Catchment boundaries are from the Brazilian Water Agency (ANA, 2010) and land use classifications are from our MODIS classification.

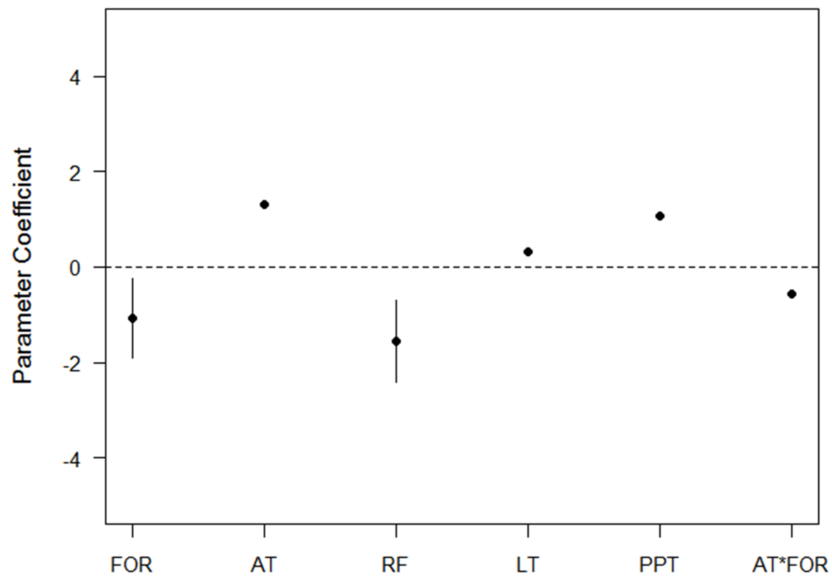

Figure B.3: Estimated parameter coefficients for fixed effects included in the stream temperature model. Fixed effects included: FOR, percent forest cover in the watershed; AT, air temperature; RF, percent forest cover in the riparian buffer; LT, (log) light at the stream surface; PPT, (log) precipitation with a 2-week lag; and AT*FOR, an interaction term.

References

ANA (2010) National Hydrographic Division - Ottobacias. Available at http://www.ana.gov.br/bibliotecavirtual/solicitacaoBaseDados.asp. Accessed on June 15, 2010. Brazilian National Water Agency.

Canty M. J., Nielsen A. A. (2008) Automatic radiometric normalization of multitemporal satellite imagery with the iteratively re-weighted MAD transformation. *Remote Sensing of Environment*, **112**, 1025-1036.

Ferreira N., Ferreira L., Huete A., Ferreira M. (2007) An operational deforestation mapping system using MODIS data and spatial context analysis. *International Journal of Remote Sensing*, **28**, 47-62.

Macedo M., DeFries R., Morton D., Stickler C., Galford G., Shimabukuro Y. (2012) Decoupling of deforestation and soy production in the southern Amazon during the late 2000s. *PNAS*, **109**, 1341-1346.

Walker W. S., Stickler C. M., Kellndorfer J. M., Kirsch K. M., Nepstad D. C. (2010) Large-area classification and mapping of forest and land cover in the Brazilian Amazon: A comparative analysis of ALOS/PALSAR and Landsat data sources. *Ieee Journal of Selected Topics in Applied Earth Observations and Remote Sensing*, **3**, 594-604.