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BROKEN ROADS AND BROKEN LAWS

HOW INFRASTRUCTURE AND LAW ENFORCEMENT SHAPE AMAZONIAN DEFORESTATION FRONTIERS

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Abstract

Deforestation of tropical rainforests contributes significantly to global climate change and accelerated biodiversity loss. To mitigate these impacts, the goal of drastically reducing tropical deforestation has been a fundamental part of Brazil's national climate and biodiversity conservation strategies, which were reaffirmed on a global scale as a key part of the 2016 Paris Climate Agreement. This thesis contributes to the science policy-debate about deforestation in the tropics by studying infrastructure access and environmental governance in the Brazilian Amazon. It uses several analytical approaches on the micro-, meso- and macro-scale, and goes beyond the scope of classical deforestation studies by considering many forms of Land use and land cover change (LULCC), including pasture to crop conversion and agricultural intensification.

The first part of this study focuses on the role of accessibility for LULCC on the macro scale. It demonstrates how different concepts and measurements of accessibility can result in considerably different LULCC predictions. It adopts a regression model to explain the geographical distribution of pasture and crop expansion in the Brazilian Amazon using different accessibility measures and a panel-dataset with land cover information. The results suggest that (1) the difference between wet and dry season accessibility (due to road quality) is an important determinant of pasture and cropland expansion, and (2) that different measures of infrastructure access (e.g., distance to markets versus distance to towns or processing facilities) can explain different aspects of LULCC. These findings suggest that LULCC research can benefit from improved and context-specific accessibility measurements.

The second part of this study operates on the meso-scale by comparing LULCC trends in different deforestation frontiers. It seeks to contribute to the LULCC-policy debate by applying frontier theory insights to map, quantify, and compare land cover dynamics in the Brazilian Amazon between 2004 and 2015. Its theoretical framework performs well in explaining broad variations in scope, context, and agents of land use and land cover change across different frontier regions. We observed two types of transformative processes at deforestation frontiers in the Amazonian context. First, contemporary frontier development is characterized by an intensification of cattle ranching and an increasing share of agricultural activities in local production portfolios, which could be the result of better access to modern technologies and markets combined with forest governance induced scarcity of land for expansion of

historically dominant extensive pasture systems. Second, the proportional share of medium and large-scale deforestation declines at first, but rebounds during the observation period in all frontier types after 2012 -casting doubts on the long-term sustainability of current conservation governance approaches in Brazil.

The third part of this study adopts a micro-scale perspective in its analysis. It shows that, between 2005 and 2014, increased environmental law enforcement became a driver of intensification decisions amongst cattle ranchers in the Brazilian state of Acre, located in the western Brazilian Amazon. It uses a choice-model based on both primary and secondary data to estimate the effect of increased law-enforcement on the likelihood of ranchers to engage in pasture restoration of cattle production systems. It finds evidence that federal law enforcement activities induced restoration efforts by way of affecting risk-perception among non-compliant cattle producers, and that pasture restoration was subsequently associated with lower deforestation rates. The findings show that large, well-endowed farms were more likely to engage in pasture restoration efforts than small-scale family agriculture, potentially marginalizing the latter and jeopardizing future conservation outcomes in the region. Standard means to boost agricultural productivity, such as credit schemes and technical assistance, had mixed effects on restoration decisions in the study region. These results indicate that the alignment of environmental and agricultural support policies are critical in ensuring increased sustainability in Acre and possibly other agricultural frontier regions in the tropics with similar socio-economic and environmental challenges.

This study's findings suggest that infrastructure access and conservation policies do not only directly affect forest conservation by enabling actors' individual ability to access the forest, but also contribute indirectly by influencing their broader production choices, which are associated with varying levels of land demand. In this context, the thesis finds evidence for two often-neglected insights: First, improved infrastructure access can reduce pressure on forests if it offers producers a broader variety of production choices that are associated with lower land-demand. Secondly, law enforcement in combination with policy induced land-scarcity creates incentives to increase agricultural productivity, which might not only help to save the forest, but also contribute to the performance and competitiveness of the agricultural sector in Brazil in the long run. However, policy makers should always keep social costs in mind and seek cross-sectoral solutions.

Resumo

O desmatamento das florestas tropicais contribui significativamente para as mudanças climáticas globais e acelerou a perda de biodiversidade. Para mitigar esses impactos, um componente central da Convenção de Paris sobre Mudanças Climáticas de 2016 prevê uma redução drástica do desmatamento nos trópicos. Assim, a redução do desmatamento também faz parte das estratégias nacionais de clima e biodiversidade do Brasil. Essa dissertação contribui para o debate científico político sobre o desmatamento nos trópicos ao examinar os fatores infra-estruturais e a política ambiental e seus papéis no desmatamento da Amazônia brasileira. Ela utiliza várias abordagens analíticas nos níveis micro, meso e macro e vai além do escopo dos estudos clássicos de desmatamento ao considerar diferentes opções de uso da terra, incluindo, por exemplo, a conversão de pastagens em culturas annuais e a intensificação da pecuária.

A primeira parte desta dissertação centra-se no papel da infra-estrutura para o uso do solo e mudanças na cobertura da terra à escala macro. Mostra como diferentes conceitos e modelos de acessibilidade da infra-estrutura podem explicar diferentes aspectos do uso do solo. Um modelo de regressão é usado para explicar a distribuição geográfica das pastagens e plantas cultivadas na Amazônia brasileira usando diferentes indicadores de infra-estrutura e um conjunto de dados de painel com informações sobre a cobertura do solo. Os resultados sugerem que (1) a mudança das estações chuvosas e secas em combinação com diferentes qualidades de infra-estrutura é um determinante importante para a distribuição e extensão de pastagens e terras de cultivo e (2) que diferentes medidas para representar a acessibilidade da infra-estrutura (por exemplo, distância aos mercados vs. distância às cidades ou à indústria a jusante) podem explicar diferentes aspectos da mudança do uso do solo. Os resultados sugerem que investigadores e decisores políticos podem beneficiar de medições de acessibilidade melhoradas e específicas do contexto, optimizando potencialmente a conservação da natureza e a produção agrícola em simultâneo.

A segunda parte desta dissertação trabalha sobre a mesoscala, comparando as mudanças no uso da terra em diferentes focos de desmatamento na floresta tropical. O desmatamento e as mudanças no uso da terra na Amazônia brasileira entre 2004 e 2015 são mapeados e diferentes fronteiras são comparados de acordo com seus sistemas de uso da terra e características agrícolas. A teoria da fronteira é

usada para selecionar e desenvolver um quadro teórico que é bem adequado para explicar as grandes diferenças na extensão, contexto e agentes de mudança do uso da terra. Duas novas tendências de desmatamento e mudança no uso da terra na região amazônica foram observadas: Em primeiro lugar, os atuais focos de desmatamento também se caracterizam pela intensificação da pecuária e por uma parcela crescente da produção de culturas annuais, que poderia ser o resultado de um melhor acesso a tecnologias e mercados modernos combinados com uma escassez de terras para a expansão dos sistemas de pastagem extensiva historicamente dominantes. Em segundo lugar, a participação do desmatamento em média e grande escala está inicialmente diminuindo, mas se recuperará em todas as áreas de fronteira durante o período de observação após 2012 - o que lança dúvidas sobre a sustentabilidade a longo prazo das atuais abordagens de controle do desmatamento no Brasil.

A terceira parte desta dissertação tem uma perspectiva microeconómica na sua análise. Mostra que, entre 2005 e 2014, o aumento da punição do desmatamento tornou-se uma força motriz na intensificação da pecuária no estado brasileiro do Acre (Bacia Amazônica ocidental). O estudo utiliza um modelo de seleção estatística baseado tanto em dados primários como secundários. Com base nesses dados, a influência do aumento da aplicação da lei nas decisões de uso da terra dos agricultores foi modelada. Os resultados sugerem que as atividades das autoridades federais podem influenciar a percepção de risco com relação ao desmatamento e, assim, potencialmente induzir decisões de uso da terra mais ecologicamente favoraveis. Por exemplo, foi medido um aumento da restauração de pastagens degradadas que foi então associado a menores taxas de desmatamento. Contudo, os resultados também mostram que as grandes fazendas bem equipadas financeiramente são mais propensas a reintegrar pastagens degradadas no processo de produção do que as pequenas fazendas familiares. Isto poderia eventualmente marginalizar estes últimos económica e juridicamente, prejudicando assim a sustentabilidade dos esforços de conservação das florestas.

Os resultados dessa dissertação geralmente sugerem que programas de infraestrutura e políticas de proteção não só têm um impacto direto na conservação da floresta ao permitir ou restringir o acesso à floresta para as partes interessadas, mas também indiretamente ao influenciar a escolha de diferentes sistemas de produção, que por sua vez estão ligados a diferentes demandas de terra. Nesse contexto, essa dissertação encontra evidências de dois fenômenos menos discutidos mas importantes: Primeiro, a melhoria do acesso à infra-estrutura pode reduzir a pressão sobre as florestas se oferecer aos produtores uma gama mais ampla de opções de produção associadas a uma menor demanda de terra. E, em segundo lugar, as penalidades e a aplicação das leis florestais, combinadas com a escassez de terra politicamente controlada, criam incentivos para aumentar a produtividade agrícola. Nesse sentido, a política ambiental pode contribuir não só para salvar a floresta, mas também, a longo prazo, para o desempenho e a competitividade do setor agrícola no Brasil. No entanto, os decisores políticos devem ter sempre em mente os custos sociais e procurar soluções interdepartamentais.

Kurzzusammenfassung

Die Abholzung der tropischen Regenwälder trägt erheblich zum globalen Klimawandel und zum beschleunigten Verlust der biologischen Vielfalt bei. Um diese Auswirkungen zu mildern, sieht ein zentraler Bestandteil des Pariser Klimaabkommens von 2016 die drastische Reduzierung der Entwaldung in den Tropen vor. Die Reduzierung der Entwaldung ist demnach auch Teil der nationalen Klima- und Biodiversitätsstrategien Brasiliens. Diese Dissertation trägt zur wissenschaftspolitischen Debatte über die Entwaldung in den Tropen bei, indem sie die Faktoren Infrastruktur und Umweltpolitik und ihre Rollen in der Entwaldung im brasilianischen Amazonasgebiet untersucht. Sie verwendet mehrere analytische Ansätze auf der Mikro-, Meso- und Makroebene und geht über den Rahmen der klassischen Entwaldungsstudien hinaus, indem unterschiedliche Landnutzungsmöglichkeiten berücksichtigt werden, darunter beispielsweise die Umwandlung von Weide- in Cash-Crop Kulturen und die Intensivierung der Viehwirtschaft.

Der erste Teil dieser Dissertation konzentriert sich auf die Rolle der Infrastruktur für die Landnutzung und die Veränderung der Landbedeckung auf der Makroskala. Dabei wird aufgezeigt, wie unterschiedliche Konzepte und Modelle von Infrastrukturerreichbarkeit verschiedene Aspekte der Landnutzung erklären können. Es wird ein Regressionsmodell verwendet, um die geographische Verteilung der Weide- und Kulturpflanzen im brasilianischen Amazonasgebiet zu erklären, wobei verschiedene Infrastrukturindikatoren und ein Paneldatensatz mit Informationen zur Landbedeckung verwendet werden. Die Ergebnisse deuten darauf hin, dass (1) der Wechsel von Regen- und Trockenzeiten in Verbindung mit unterschiedlichen Infrastrukturqualitäten eine wichtige Determinante für die Verteilung und Ausdehnung von Weide- und Ackerland ist und (2) dass verschiedene Maße zur Darstellung der Infrastrukturerreichbarkeit (z.B. Entfernung zu Märkten vs. Entfernung zu Städten oder nachgelagerter Industrie) verschiedene Aspekte des Landnutzungswandels erklären können. Die Ergebnisse legen nahe, dass Forscher und politische Entscheidungsträger von verbesserten und kontextspezifischen Erreichbarkeitsmessungen profitieren können, um so möglicherweise Naturschutz und Agrarproduktion simultan zu optimieren.

Der zweite Teil dieser Dissertation arbeitet auf der Mesoskala, indem er die Landnutzungsveränderungen in verschiedenen Abholzungshotspots im Regenwald vergleicht. Dabei werden Entwaldung und Landnutzungsveränderungen im brasilianis-

chen Amazonasgebiet zwischen 2004 und 2015 kartiert, und verschiedene Hotspots werden gemäß ihrer Landnutzungssysteme und Landwirtschaftscharakteristika verglichen. Mit der Frontier-Theorie wird ein theoretischer Rahmen gewählt und weiterentwickelt, der gut geeignet ist, um die großen Unterschiede in Bezug auf Umfang, Kontext und Agenten der Landnutzungsveränderungen zu erklären. Dabei werden zwei neue Trends bei den Entwaldungs- und Landnutzungsveränderungen im Amazonasgebiet herausgestellt: Erstens sind heutige Entwaldungshotspots auch durch eine Intensivierung der Viehzucht und einen zunehmenden Anteil von Cash-Crop Produktion gekennzeichnet, was das Ergebnis eines besseren Zugangs zu modernen Technologien und Märkten in Verbindung mit einer durch die Forstverwaltung verursachten Landverknappung für die Ausweitung der historisch dominanten extensiven Weidesysteme sein könnte. Zweitens nimmt der Anteil der mittelgroßen und großflächigen Entwaldung zunächst ab, erholt sich aber im Beobachtungszeitraum in allen Grenzgebieten nach 2012 wieder - was Zweifel an der langfristigen Nachhaltigkeit der derzeitigen Ansätze zur Entwaldungsbekämpfung in Brasilien aufkommen lässt.

Der dritte Teil dieser Dissertation nimmt in seiner Analyse eine mikro-ökonomische Perspektive ein. Er zeigt, dass zwischen 2005 und 2014 die verstärkte Bestrafung von Entwaldung zu einer treibenden Kraft in der Intensivierung der Viehwirtschaft im brasilianischen Bundesstaat Acre (westliches Amazonasbecken) wurde. Die Studie verwendet ein statistisches Auswahlmodell, das sowohl auf primären als auch auf sekundären Daten basiert. Auf Grundlage dieser Daten wird der Einfluss der verstärkten Rechtsdurchsetzung auf die Landnutzungsentscheidungen der Landwirte modelliert. Die Ergebnisse deuten darauf hin, dass die Aktivitäten der Bundesbehörden die Risikowahrnehmung in Bezug auf Entwaldung beeinflussen und damit potentiell ökologisch sinnvollere Landnutzungsentscheidungen induzieren können. So wird eine verstärkte Wiederherstellung degradierter Weideflächen in den Produktionsprozess gemessen, die dann mit geringeren Entwaldungsquoten verbunden ist. Die Ergebnisse zeigen jedoch auch, dass große, finanziell gut ausgestattete Betriebe eher degradierte Weideflächen in die Produktion reintegrieren als kleine Familienbetriebe. Dadurch könnten letztere möglicherweise ökonomische und rechtlich marginalisiert werden, was die Nachhaltigkeit der Bemühungen zum Waldschutz langfristig negativ beeinflussen könnte. Konventionelle politische Instrumente zur Unterstützung der landwirtschaftlichen Produktivität, wie Kreditprogramme und technische Unterstützung, haben gemischte Auswirkungen auf die Weidewiederherstellung in der Untersuchungsregion. Die Ergebnisse weisen darauf hin, dass eine Harmonisierung der Umwelt- und Landwirtschaftspolitik entscheidend ist, um die nachhaltige Landnutzung in der Untersuchungsregion sicherzustellen.

Die Ergebnisse dieser Dissertation legen allgemein nahe, dass sich Infrastrukturprogramme und Schutzpolitik nicht nur direkt auf die Erhaltung der Wälder auswirken, indem sie den Akteuren den Zugang zum Wald ermöglichen oder beschränken, sondern auch indirekt, indem sie die Wahl unterschiedlicher Produktionssysteme beeinflussen, die wiederum mit unterschiedlichen Flächenansprüchen in Verbindung stehen. In diesem Zusammenhang findet diese Dissertation Belege für zwei weniger prominent diskutierte, aber wichtige, Phänomene: Erstens kann ein verbesserter Zugang zur Infrastruktur den Druck auf die Wälder verringern, wenn er den Produzenten eine breitere Palette von Produktionsentscheidungen bietet, die mit einer geringeren Landnachfrage verbunden sind. Und zweitens schaffen Strafen und der Vollzug der Forstgesetzte, in Kombination mit einer politisch gesteuerten Landverknappung, Anreize zur Steigerung der landwirtschaftlichen Produktivität. Umweltpolitik kann in diesem Sinne nicht nur zur Rettung des Waldes beitragen, sondern langfristig auch zur Leistungs- und Wettbewerbsfähigkeit des Agrarsektors in Brasilien. Politische Entscheidungsträger sollten dabei jedoch soziale Kosten stets im Auge behalten und ressortübergreifende Lösungen suchen.

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CHAPTER 1

Introduction

1.1 Context and Study Relevance

Global greenhouse ssgas emissions have continued to increase at an accelerated rate throughout the 2010s. Without drastic additional commitments and concrete efforts to reduce anthropogenic emissions, the world's global average temperature is likely to increase by 3.0° to 3.7° C compared to pre-industrialized levels by the end of the century (IPCC 2018). Temperature increases above 1.5° to 2.0° C, which can be considerably higher in localized regions, are very likely to drastically transform global marine and terrestrial ecosystems and have a profound negative impact on wildlife and human society (IPCC 2014). In addition to climate change, biodiversity loss is also accelerating, and has already reached a level at which even conservative estimates suggest that earth is entering its sixth mass extinction period (Ceballos et al. 2015).

One of the main contributors to global climate change and local biodiversity loss is the substitution of natural forest vegetation with industrialized agricultural land use through deforestation. Emissions related to agriculture, forestry, and other land Uses (AFOLU) are responsible for up to 24% of global greenhouse gas emissions p.a. (Smith et al. 2014), and more than half of these emissions originate from LULCC (IPCC 2019). Additionally, LULCC and the overexploitation of natural resources have been the major drivers behind the conversion of 47% of natural ecosystems into urban, semi-urban, or agricultural use and the loss of a significant proportion of global mammal populations and floral biodiversity (IPBES 2019).

LULCC related emissions and biodiversity loss is particularly prevalent in the tropics, where evergreen rainforests are removed and converted to agricultural land to produce goods for national and global commodity markets (ibid.). The Amazon rainforest is the world's largest remaining tropical forest area, as well as the biggest terrestrial carbon sink, and a global hotspot of biodiversity (Cardoso et al. 2017). Around 60% (3.3 million km^2) of the Amazon rainforest is situated in Brazil, which subsequently has the biggest responsibility in deterring deforestation in the remaining forested areas.

Infrastructure expansion into the Brazilian Amazon was heavily supported throughout the military government era (1964 to 1985), with the goal of integrating the region with the rest of the country through investment and development policies. However, despite subsequent large scale road construction projects, connectivity remains poor throughout the region, due to its massive size, and because the large road infrastructure is difficult and expensive to maintain. As a consequence, many paved roads are either not paved or broken due to insufficient maintencance which influences agricultural production conditions considerably (Fearnside 2005, Koch et al. 2019). Regardless, the Brazilian government has continually supported the resettlement of landless peasants from the poorer parts of the country to the Amazon region and made settlers accountable for converting the seemingly abundant land for production purposes by including land consolidation quotas as a prerequisite to obtain governmental support.

As a consequence of these and other agricultural policies that encouraged deforestation (see: Bank and Binswanger 1991), forest cover loss spread along new and existing infrastructure and in regions where federal settlement projects were established (Chomitz, Buys, et al. 2007, Barber et al. 2014). All told, 20% of the Brazilian Amazon's original forest cover has been lost (Mapbiomas, 2019) and estimates suggest that an additional 7% have been disturbed and degraded due to logging, ground-fires, and other human activities (Souza et al. 2013).

The most prevalent land use in deforested areas in the Amazon region is extensive cattle ranching. Pasture production systems, compared to cropland expansion, are able to quickly consolidate large tracks of deforested land with relatively few inputs and have low labor requirements (Bowman et al. 2012). As a popular national proverb states: *"Brazil was conquered under the hoof of a bull"* (port. "sob a pata do boi"), which is certainly the case for the Amazon region as well.

Pioneer efforts to protect the Brazilian Amazon and its indigenous populations date back to the early 1960s, but first gained major political attention and international support after the Rio Earth Summit in 1992 (Kirby et al. 2006). This resulted in the 1993 International Pilot Programme to Conserve the Brazilian Rainforest (PPG7) ushering in a new era of cooperation between the Brazilian government and civil society to scale conservation efforts and consolidate environmental protection with socio-economic development (Antoni 2010). The program was supported financially and technically by the seven most industrialized countries (G7) at the time, the World Bank, and international NGOs. PPG7 formed the foundation for a number of subsequent important conservation policies, and helped to build and strengthen several institutions that are fundamental to Brazil's conservation sector today (ibid.). Despite PPG7's large initial investment (US$ 428 million), it took ten years to create effective institutions and reverse growing deforestation trends – peaking at 27.8K sqkm in 2004 (see Figure 1.1).

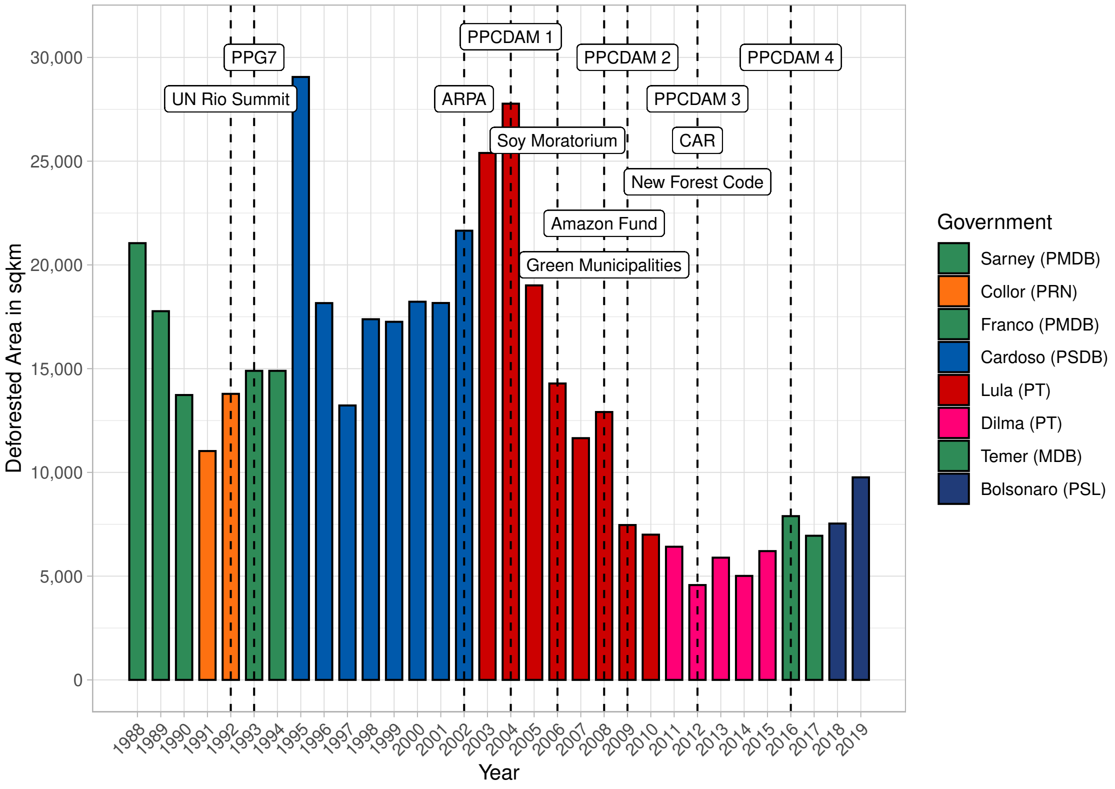


Figure 1.1: Historical deforestation rates in the Brazilian Legal Amazon and policy context

The years following the 2004 peak in deforestation rates are considered the "golden years" of deforestation control in Brazil as rates fell by more than 80% between 2004-2014. This large scale reduction in deforestation rates is largely attributed to ongoing conservation policy efforts coordinated under the Brazilian Action Plan for the Prevention and Control of Deforestation in the Legal Amazon – The Action Plan for the Prevention and Control of Deforestation in the Legal Amazon (PPCDAm) which was effective in assuring that breaking of federal forest laws was no longer gone unpunished (Macedo et al. 2012, Nolte et al. 2013, Assunção, Gandour, and Rocha 2013).

Nevertheless, in 2019 deforestation in the Brazilian Amazon rebounded to 9.8K km^2 and put Brazil well behind the national climate change strategy to reduce deforestation in the Amazon below 3.9K km^2 by 2020 (Brazil 2008). Recent increases in deforestation rates have occurred in a period of considerable political hostility towards environmental institutions and an economic recession, leading current politicians to signal (renewed) tolerance for people who break the law and cause more deforestation in the Amazon. After the national elections in 2018, political forces against deforestation control have gained power and are close to reversing the success of almost 20 years of conservation efforts in Brazil (OC 2019).

1.2 Problem Statement and Research Objectives

In the past two decades, a considerable body of literature has been published regarding deforestation in the Amazon. One of the first and most important research topics brought forth by this research area is the role that infrastructure expansion plays in shaping development in the region and its' (mostly negative) impact on forest resources (e.g. Pfaff, Robalino, Walker, et al. 2007, Weinhold and Reis 2008, Barber et al. 2014).

The general consensus of these studies is that roads generally promote deforestation (for an overview see Chomitz, Buys, et al. 2007). Nevertheless, local studies also found that the effect sizes that road construction has on deforestation rates differ across different sub-regions of the Amazon basin (Aguiar, Câmara, and Escada 2007), and that in some cases roads might even reduce deforestation rates depending on local governance scenarios as well as the quality and condition of the infrastructure (Andersen and Reis 1997, Pfaff 1999).

A closer look at the existing body of literature reveals that, while most studies have sound explanations and arguments for differing effects road construction has on deforestation rates, many use different measurement techniques and concepts of infrastructure accessibility - often without a detailed discussion regarding the rationale of their choice. If authors do not discuss their accessibility concepts and measurement techniques in detail, it becomes difficult to ascertain whether heterogeneous effects are due to the use of different covariates and levels of analysis, or if they are the result of different accessibility concepts and measurement techniques.

The first goal of this thesis is, therefore, to advance our knowledge regarding the role of infrastructure accessibility on LULCC dynamics in the Amazon by analyzing accessibility along different conceptual dimensions and different geographical scales of analysis. Special attention will be given to the effect of low infrastructure quality (broken roads) and the consequences for land use choice at local scale.

An important theoretical approach to describe the role of infrastructure on LULCC dynamics in different local realities is to categorize dynamic land cover change areas into different types of deforestation frontiers. What all frontiers generally have in common is their relative remoteness to existing markets and high deforestation rates as opposed to pristine forest areas or consolidated settlements. What generally differentiates them is a set of local socio-economic dynamics and biophysical factors that shape localized land use change in their respective contexts.

Several studies have been published describing different deforestation frontiers in the Amazon and their underlying dynamics at local scale (e.g. Fearnside 2001; Walker et al. 2002; Pacheco 2005; Aldrich et al. 2006; Jepson 2006; Caldas et al. 2007; VanWey, D'Antona, and Brondízio 2007; Browder et al. 2008; Michalski, Metzger, and Peres 2010; Carrero and Fearnside 2011). However, few studies have created a comprehensive assessment of different frontier categories at regional scale (Pacheco and Poccard-Chapuis 2012), and none have linked different accessibility characteristics across the entire Amazon basin to frontier theory using a holistic conceptual approach. This has resulted in a lack of cross-case comparability among the frontier related literature in the Amazon and a missing conceptual link between different infrastructure conditions and frontier specific LULCC dynamics. We argue that this analytical perspective is important in order to design conservation policies and programs that are able to target heterogeneous frontier conditions (Sayer et al. 2013)- especially considering the massive geographical size of the region, which is larger in size than European Union.

The second objective of this thesis is, therefore, to create a holistic picture of deforestation frontiers and associated LULCC dynamics across the entire Amazon basin and to disaggregate the macro-scale assessment of infrastructure accessibility to a meso-regional analysis with multiple criteria that describe the differing frontier conditions that occur in the region in addition to an analysis of infrastructure accessibility.

Finally, an important sub-section of LULCC literature looks at the effects of conservation policies and environmental regulation on deforestation in the Amazon region. Several studies found evidence that multiple public policies and civil-society led programs (see Figure 1.1) have been effective in deterring people from breaking forest laws during the 2004 to 2012 period. In this context, the most important conservation initiatives have been: the expansion of the protected area network under the 2002 ARPA program (Soares-Filho, Nepstad, et al. 2006, Andam et al. 2008); satellite-based law enforcement of the forest code using fines and property embargoes for illegal deforestation during phase one and two of PPCDAm (Assunção, Gandour, and Rocha 2015, Börner, Wunder, Wertz-Kanounnikoff, et al. 2014); supply-chain-governance initiatives such as the 2006 Soy Moratorium (Nepstad et al. 2014, Lambin, Gibbs, et al. 2018) and zero-deforestation agreements with major slaughterhouses (Gibbs, Munger, et al. 2015, Alix-Garcia et al. 2017); the introduction of a public blacklist for municipalities with severe deforestation rates under the Green Municipalities program (Cisneros, Zhou, and Börner 2015); and the introduction of a public rural cadaster (CAR) to consolidate environmental information about farms (Alix-Garcia et al. 2017).

All of the aforementioned studies use deforestation as their main outcome variable and provide important information on policy effectiveness to control deforestation in the Amazon region. However, they do not provide information about the larger picture of land use change in a transforming policy environment. Our goal is to provide a comprehensive analysis that fills this gap in order to promote a transition towards sustainable production practices. Developing sustainable production altenatives is also one of the main objectives of the PPCDAm – in order to promote the longterm sustainability of deforestation control in the Amazon(MMA 2013). in this context, Barbier, Burgess, and Grainger (2010) argue that designing a policy mix with adequate instruments to promote sustainable land use requires comprehensive knowledge about how land use systems change in response to conservation governance. If not, political instruments to promote sustainable production might fail to provide the correct incentives for conservation (Strassburg et al. 2014), or misspend financial resources on assistance strategies that do not produce additional conservation effects (Merry and Soares-filho 2017).

The third goal of this thesis is to analyze how conservation policies are associated with land cover change dynamics in different agro-economic realities in the Amazon. Special attention is given to the effects of conservation policies on law-abiding behaviour and land use choice in pasture-based production systems, due to the outstanding role cattle ranching has on the rural economy and deforestation in the Amazon.

1.3 Organization of this Thesis

This thesis is organized into five chapters, each of which address the research objectives in different ways. Following this introduction, chapter 2 addresses the role of infrastructure accessibility for LULCC dynamics in the Amazon. It demonstrates how different concepts and measurements of accessibility can influence prediction outcomes of LULCC using a panel-data model to explain the geographical distribution of pasture and crop expansion in the Brazilian Amazon. An open-source tool was created in order to quickly generate and modify different accessibility indicators, allowing for the creation of travel time maps in the R statistical programming language. This open-source tool is briefly described in the second chapter and further documented in the Supplementary Material (SM), Section D and E.

Chapter 3 addresses the goal of creating a holistic picture of deforestation frontiers and associated LULCC dynamics in the Brazilian Amazon. It categorizes frontiers using an empirical approach based on a big-data clustering procedure with a multivariate set of spatial raster layers of the region. The results of this chapter are visualized using a comprehensive frontier map of the Brazilian Amazon. Additionally, broad agricultural production conditions, land use characteristics, and deforestation dynamics are described within the frontiers using land cover and defores tation panel data and geo-referenced data from the 2006 Brazilian Agricultural Census. In this regard, chapter 3 also contributes to the goal of analyzing how conservation policies are associated with land cover change dynamics in different agro-economic realities.

Chapter 4 addresses this question in even further detail on the micro-economic scale. It presents a Heckman two-step estimation procedure to analyze how environmental law-enforcement is associated with cattle intensification and deforestation in the western Brazilian Amazon. It uses data from a field-survey with cattle ranchers that was developed and applied in Acre in collaboration with the Brazilian Agricultural Research Corporation (EMBRAPA).

1.4 Context and Genesis of this Thesis

This study was elaborated within the Robert Bosch funded project "Shaping Environmental Policies for Sustainable Tropical Forest Bioeconomies" (Grant number: 32.5.8043.0012.0). The goal of this interdisciplinary project was to address two central aspects of conservation governance: (1) The role of alternative instrument design options in affecting policy cost-effectiveness and welfare impacts in spatially heterogeneous bio-physical and institutional settings, and (2), quantitative measures of the resulting scope for environmental policy choice and design given multiple tradeoffs among bioeconomy development objectives. The project was hosted by the Center for Development Research (ZEF) in Bonn, Germany and supported junior researchers to develop their PhD thesis with research grants and field visits to Peru and Brazil. The research activities resulted in several peer-reviewed publications (e.g. Cisneros, Zhou, and Börner 2015; Börner, Baylis, et al. 2016; Börner, Wunder, and Giudice 2016, Frey et al. 2018, Schielein and Börner 2018; Giudice et al. 2019; Miranda et al. 2019) and allowed for strong interdisciplinary cooperation amongst researchers in the group.

This project provided me with an opportunity to continue my previous work in the conservation sector in Brazil on an academic level. Before this project, I worked as an intern and later as a consultant for the German Development Agency (GIZ), providing technical support to the Brazilian Ministry of the Environment in conservation policy related aspects. I joined GIZ in 2009 after the PPG7 had officially ended, and the Brazilian Government was elaborating the third phase of PPCDAm. In 2010/2011 I joined a team that supported the establishment of the Brazilian Amazon Fund (Fundo Amazônia) at the National Bank for Economic and Social Development (BNDES). The Amazon fund was established as a body to channel international conservation finance to the Amazon and disbursed 484 Mio. USD in 103 supported projects between 2009 and 2019. As a result of my previous work experience, I learned that there were two questions that were frequently discussed amongst policy makers at the ministry and in the Amazon fund, which in turn influenced the formulation of my research goals:

1. How effective are policies and programs that are designed at the national level at the local level, especially considering the unique contexts and wide spectrum of different places within the region?

Policy makers have to design programs for of places like Manaus or Belem, were people live under the influence of large urban agglomerations with millions of people and an advanced industrial and service industry. However it also has to anticipate the context of places like Sinop in Mato Grosso that serve as a logistical hub for agricultural exports to Europe and China in addition to remote places like Colniza, where cattle ranching is the only viable production option given its remoteness. There are also places such as the RESEX Chico Mendes in Acre, where traditional communities, the former defenders of the forest, are increasingly favoring cattle ranching over the collection of non-timber forest products such as rubber or Açaí berries.

2. Which economic alternatives exist to support livelihoods in rural Amazonia and how can a transition towards sustainable rural production systems be

achieved in order to reconcile environmental conservation and rural development at deforestation frontiers?

Although a single PhD thesis alone is certainly not enough to address these big questions in detail, my goal was to frame my research in a way that contributes to a better understanding of the economic drivers in the Amazon and provide a holistic assessment of local deforestation contexts by altering the scale and methodology of my analysis.

CHAPTER 2

The Role of Accessibility For Land Use and Land Cover Change in the Brazilian Amazon

An article with the same content including Appendix A has been submitted for review to Applied Geography as Schielein, J.,Frey, G., Miranda, J., Souza, R. Börner, J. and J. Henderson: The Role of Accessibility For Land use and Land cover Change in the Brazilian Amazon.

2.1 Introduction

Throughout history, infrastructure has played a crucial role in development by supporting human settlements and the expansion of agricultural frontiers into inaccessible rural areas. As planetary boundaries become binding constraints for human survival, infrastructure also features prominently in research on the drivers of natural habitat conversion, biodiversity loss, and emissions from deforestation and forest degradation (Nelson and Hellerstein 1997, Chomitz, Buys, et al. 2007, Laurance, Goosem, and Laurance 2009, Ibisch et al. 2016). Arguably no other region has drawn as much attention in this field of research as the South-American Amazon biome - home to the largest remaining tropical rainforest in the world.

With around 80% of its canopy cover intact (MapBiomas Project 2017), the Amazon rainforest provides essential ecosystem services to millions of people in the nine countries that share borders in the Amazon basin. It provides livelihoods to many traditional populations including an estimated 130 isolated indigenous communities (Butler 2019) and hosts as much as 11% of global tree plant biodiversity (Cardoso et al. 2017). Furthermore, the Amazon plays a crucial role in regulating precipitation patterns in South America (Spracklen and Garcia-Carreras 2015) making its integrity and conservation of paramount importance in stabilizing the global climate system (IPCC 2013). Nevertheless, forest cover loss is accelerating worldwide (Hansen et al. 2013) and has recently increased again in the Brazilian Amazon (INPE 2019).

Roads have been widely recognized as one of the most important proximate drivers of tropical deforestation because they improve access to remote rural areas (Laurance, Albernaz, et al. 2002, Chomitz, Buys, et al. 2007, Barber et al. 2014). Hence, infrastructure investments and accessibility improvements are often considered a threat to rainforest conservation (Davidson et al. 2012). Some studies, however, describe ambiguous effects of accessibility improvements on the rate of deforestation, suggesting that local factors interact with accessibility and determine whether improvements have a negative or positive effect on deforestation. Andersen and Reis (1997), for example, discusses that investments in already consolidated areas might increase land prices and create an incentive to cultivate more capital- and less land-intensive perennial crops, which might subsequently negatively influence deforestation. Pfaff (1999) and Pfaff, Robalino, Reis, et al. (2018) argue that road improvements to existing cities instead of road expansion into remote areas might reduce deforestation rates and stimulate social and economic development. Chomitz, Buys, et al. (2007) give a comprehensive overview of the effect of roads on deforestation. They summarize that roads generally stimulate deforestation, but also stress the fact that all-weather roads can potentially reduce deforestation if the local agricultural structure allows alternating between pastures and crop plantation. They also stress the importance of local governance and land-tenure systems, which mediate the impact of new or improved infrastructure on deforestation.

In this study, we take a step back and show that how accessibility is measured matters in quantitative research on LULCC. Specifically, we conceptualize access in terms of four dimensions, namely actor specificity (access by whom), origin (access from where), destination (access to what), and seasonality (access when). Most studies on LULCC in the Amazon have so far selectively relied on only one of these concepts.

Building on these accessibility dimensions, we construct alternative accessibility indicators and compare their association with LULCC in the Brazilian Amazon. To estimate the likelihood of an area being converted into either pasture or crops, we adopt a conditional logit model to study impacts of infrastructure improvements between 2004 and 2011 on LULCC in the period from 2012 to 2016.

We find that the "destination" dimension (i.e., access to cattle- and agricultural-markets, urban infrastructure, or land-speculation areas) matters in explaining forest to pasture or cropland conversion i.e. that each product requires its own marketplace, which are often located in different locations. Furthermore, seasonality of access plays an important role that varies across crop and pasture-based production systems in the Amazon and their requirements to access different travel destinations at different moments in time.

Our study comes with the AccessiblityMaps package which is a GIS-framework for creating accessibility maps in R. The current development version of the package includes all necessary tools to create accessibility maps from scratch with custom data or freely available open data sets such as the The Open Streetmap Project (OSM). It was elaborated for this and other land use modeling studies in the Amazon (Frey et al. 2018, Schielein and Börner 2018, Miranda et al. 2019).

2.2 Background: Accessibility and LULCC in the Amazon

2.2.1 Concepts

The term accessibility is defined in the Cambridge Dictionary in very general terms as the ability of something to be reached, obtained, approached or used easily. In the field of economic geography and location theory, this "something" is usually a central marketplace that provides people in remote areas with social and economic opportunities (Yoshida and Deichmann 2009). The central market theorem has its origins in the works of Thünen (1910). His model of agricultural land describes how transport costs and the limited durability of agricultural goods lead to a pattern of concentric land use circles around a central market with increasing land use intensity towards the center. Following von Thünen, extensive land use forms with lower land-rents, such as cattle ranching or forestry, are to be found in the outer, remote areas, whereas intensive land uses, such as the production of fruits or vegetables, are to be expected close to the central market. In this context, the travelers ("who") are farmers, and the travel destinations ("where") are central markets.

Our literature assessment (see SM) suggests that most authors use the idea of a central marketplace to explain the geographical distribution of LUCC and the movement of agricultural frontiers in the tropics. Many studies use different notions of a market, however. 30 out of 40 studies consider urban areas as marketplaces and a few of them also stratify by market size. Stratification is often achieved using somewhat arbitrary population cuts (e.g. a market is a city with a population larger than 10.000 inhabitants). While some studies motivate the choice of marketplaces in greater detail or base them on contextual knowledge (e.g. Nelson and Geoghegan 2002), most of the reviewed studies do not provide a definition of why certain towns had been included as markets or not. Additionally, only four studies considered agricultural infrastructure, such as cattle-slaughterhouses or crushing facilities and silos for grains as potential marketplaces.

Using urban areas as proxies for agricultural markets would appear questionable especially in specialized agricultural systems that produce globally traded commodities, where transport, processing, or storage tends to occur outside urban areas. Nevertheless, access to the right infrastructure for specific produce can be essential when it comes to land use decision-making even in a globalized agricultural economy (Fearnside 2008).

Still, proximity to urban centers matters as people employed in agriculture need access to supermarkets, medical facilities, and schools. In the Amazon, farmers may prefer to live in the city, especially if farming is an secondary income source in addition to urban employment (Tritsch and Le Tourneau 2016). The frequency and seasonality of travel to a given destination would thus clearly matter as a determinant of land use decisions. Few of the reviewed studies, however, address this accessibility dimension in detail. For example, only 3 out of 40 studies discuss the implications of decreasing accessibility during the wet season because of an absence of all-weather roads in their concept and measurement of accessibility (Serneels and Lambin 2001, Chomitz and Thomas 2003, Kirby et al. 2006).

Limited accessibility during or at the end of the wet season might, however, influence land use decisions considerably. In von Thünens' theory, the (limited) durability of fruits and legumes essentially binds them to the close circumference of the city in which they are to be consumed. In the Amazon, where accessibility alters between wet and dry season, farmers may face considerable financial risks if they are unable to market perishable goods harvested during or at the end of the wet season. Animals are, in contrast, a much more flexible product that can be marketed whenever the weather conditions permit road transport (Salisbury and Schmink 2007, Cohn, Gil, et al. 2016). Therefore, ranching can be considered as a low-risk production option in areas with limited accessibility in and around the wet season (Pichón 1997).

Another important aspect of accessibility in relation to LUCC in the Amazon is the phenomenon of land-speculation. The establishment of pastures is a relatively cheap way to secure large land-claims in tenure-insecure environments. Intensive cropping systems, on the other hand, often require large upfront investments (Fearnside 2008), which might be lost if the land-claim cannot be enforced. Often, new land claims are raised in areas where planned investments in infrastructure are expected to increase land values of consolidated (pasture) land in the future (Fearnside 2008, Bowman et al. 2012, Miranda et al. 2019). It is, therefore, important to not only consider current levels of accessibility, but also expected accessibility improvements.

Based on the discussion above we formulate four general working hypotheses but expect variations subject to local context:

1. cattle ranching and crop production are both dependent on access to central markets, but the type of market differs in each case (extending the central market hypothesis in the "destination" dimension).

2. cattle ranching and crop production are both dependent on access to urban areas, which represent important places of social and economic interaction. This access is required year round (permanent interaction hypothesis).

3. In contrast to cattle ranching, crop production dominates in areas with year round accessibility. This is required to minimize the risk of losses if the infrastructure does not allow road-transportation after the harvest season (risk-reduction hypothesis).

4. In contrast to crop production, cattle ranching is more prevalent where infrastructure improvements, most importantly the pavement or roads, are to be expected in the near future (speculation hypothesis).

We analyze these four hypotheses using a data set that allows us to characterize accessibility from varying production systems (and corresponding actors) to different travel destinations, and across major seasons.

2.2.2 Measurements

There are different approaches to measure the accessibility of a travel destination from a given Point of Interest (POI). The most straight-forward approach is to use Euclidean distances from the travel destination to the POI (Figure 2.1, case A). Euclidean distances are straight lines in planar space, and their calculation only requires the geographical coordinates of POIs and travel destinations. Out of 40 studies that we reviewed, 28 use Euclidean distances to measure accessibility, thus abstracting from temporal and spatial heterogeneity in transportation infrastructure quality, as well as geographical barriers or different weather conditions for transport.

In order to account for different transportation types, most studies that use Euclidean distances combine different distance metrics. In our literature assessment, we found that out of 28 studies, nine used Euclidean distances to markets in combination with either distance to roads or distance to rivers; five used combinations of distance to markets with distance to roads and rivers; and eight studies used distance to markets with another type of accessibility measurement. Nevertheless, six studies only used a single measurement, which is prone to produce an oversimplified and distorted representations of access, as shown in Figure 2.1. In this figure, we illustrate common accessibility measures and some possible problems with their accuracy. Measures of Euclidean distance, for example, can produce oversimplified estimations if the transportation network is not accounted for (case A). Also, an incomplete combination of only two measurements (e.g. roads and markets) can be problematic as shown in case B, where river based transportation is neglected, which is the most likely transportation route in this specific location.

Another common accessibility approach is shown in case C. It consists in measuring the density or extension of roads and rivers within a given Area of Interest (AOI). This measurement method can result in relatively accurate accessibility indicators under coarse spatial resolution and if all AOIs have sufficient and evenly distributed travel destinations. If the distribution of travel destinations is not even, however, this might lead to inaccurate or distorted predictions.

Case D in Figure 2.1 shows a more precise method to measure accessibility, which consists of using travel time information based on actual transportation infrastructure. This can be either achieved by using some form of routing and the length of the transportation network (see e.g. Walsh et al. 2008) or by creating a continuous surface of travel-times from each location on the map to the travel destination. The latter is also referred to as a travel time map, cumulated travel time map or travel

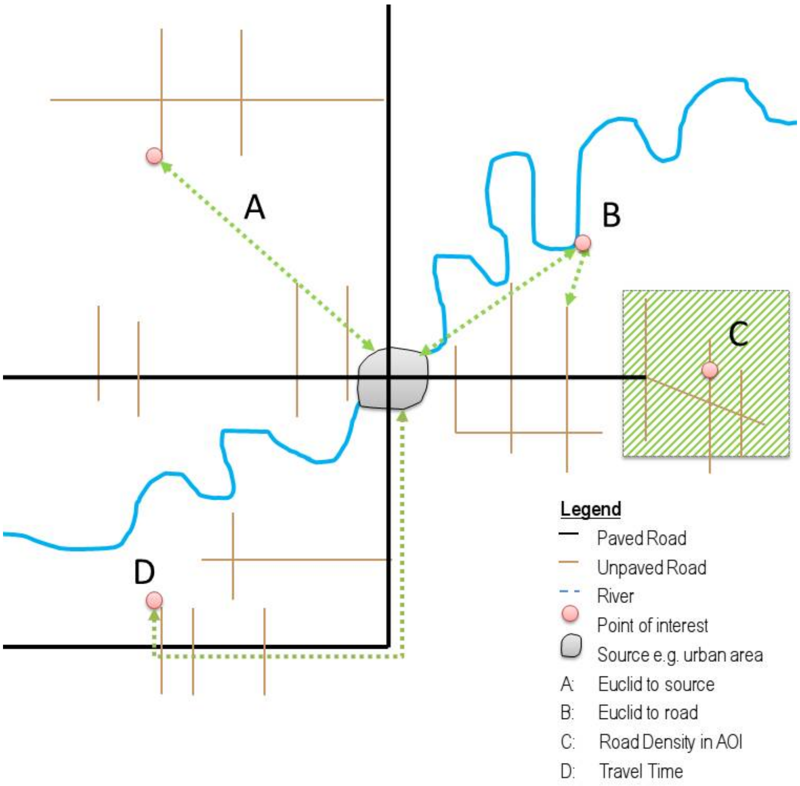


Figure 2.1: Common methods to measure accessibility in the context of LULCC studies

cost map (Baradaran and Ramjerdi 2001, Weiss et al. 2018), and while calculations based on routing require good infrastructure data (e.g. uninterrupted road and river segments), travel time maps are more flexible to use with incomplete data sets. In this paper we use travel time maps for our accessibility measurements.

Another advantage of travel time estimations is that different assumptions on travel speed can be made to account for differences in transportation vehicles, infrastructure quality, and weather conditions. Travel time estimations, which require moderate processing power, are, therefore, highly flexible and adaptable to the researcher's needs. Nevertheless, only nine out of 40 studies from our review used some type of travel time map or least cost calculation to estimate accessibility.

Complex models of accessibility, such as gravitational models, also account for different degrees of relevance of the travel destinations to create some type of attraction factor that is combined with a distance measurement (Geurs and Wee 2004, Kompil et al. 2019, Yoshida and Deichmann 2009). In the context of LUCC, for example, the demand of each market could be included in the measurement to account for both distance and market size when measuring accessibility.

Besides the measurement technique, data quality is also an important element needed to accurately measure access. This is especially relevant in developing countries which often lack adequate infrastructure information, especially regarding unofficial secondary road networks (Bourguignon Boris 2007). Nevertheless, data quality was often not addressed sufficiently in our reviewed literature. Only seven out of 40 studies discussed shortcomings of their data-quality, whereas the others did not discuss data quality in any level of detail.

2.3 Methods

2.3.1 Mapping travel time in R

The general approach to produce a travel time map requires the combination of information about transportation infrastructure and the natural geography of the study area in a so-called friction surface (sometimes referred to as cost-surface). A friction surface is a spatial grid that exhibits the amount of time that is necessary to cross each cell of the study area in both vertical or horizontal directions.

The amount of time necessary depends on the underlying land cover type and resolution of the friction surface map. A paved road, for example, will translate into a lower friction than a cell occupied by forest. It is up to the researcher to define an adequate travel-speed for each surface type. The friction values are then corrected for the effect that a hilly or mountainous topography might exert on travel speed i.e. steep slopes increase friction values.

In a second step, the values from the friction surface are aggregated by an algorithm that optimizes travel time for all cells in the grid. This is done by cumulating friction values from the travel destinations to the periphery of the map and choosing the fastest travel route in an iterative process (see Figure 2.2).

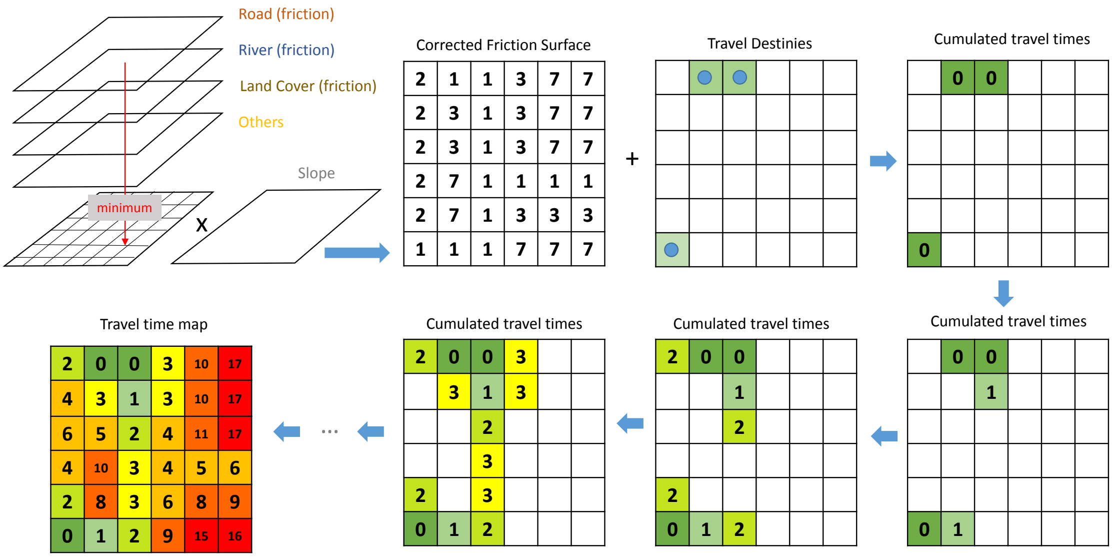


Figure 2.2: Simplified processing steps of a travel time map creation

Adapted from Farrow and Nelson (2001)

There are different commercial and open-source tools to create travel time maps (for an overview see SM, section A.5). However, we could not find a complete solution that does the necessary pre-processing and cumulating steps and is free and open-source software. We, therefore, created this solution using R and its several bindings to powerful GIS libraries such as GRASS (GRASS Development Team 2019) and GDAL (GDAL-OGR Contributors 2019).

Our functions build upon the rgdal package (Bivand, Keitt, and Rowlingson 2019) and the R-internal raster package (Hijmans 2019) to create a friction surface.

Travel times are calculated with the *r.cost* algorithm from GRASS (Awaida et al. 2018) using the rgrass7 package (Bivand 2018). The functions can be used by installing the AccessibilityMaps package from GitHub. A detailed description of the package can be found in the SM with a documentation of the processing workflow and an example tutorial with OSM data (Section D), as well as a section with the code documentation (Section E).

2.3.2 Data and Study Region

We draw on different data sources to create an optimal representation of available transportation infrastructure and travel destinations in the Brazilian Amazon Biome (for an overview see Figure 2.3 and for details the SM, Sections A.2 and A.3). We assume that travelers can combine different forms of transportation (walking/car/boat) to get to their desired travel destination. For each input data set, we assign different travel speeds depending on the surface-type and the weather conditions (dry season vs. wet season) and we calculate travel time maps in 2004 and 2012 in a spatial grid with 200x200 meter resolution.

Our travel destinations are (1) cattle slaughterhouses which act as the most important market for beef production; (2) silos, which are an integral part of the crop value chain and the main storage location before marketing; (3) urban areas, which serve as important input-markets for cattle and crop production, and which are important places for residence and other socio-economic interactions; and (4) roads undergoing paving which we take as an indicator for areas of increased land speculation.

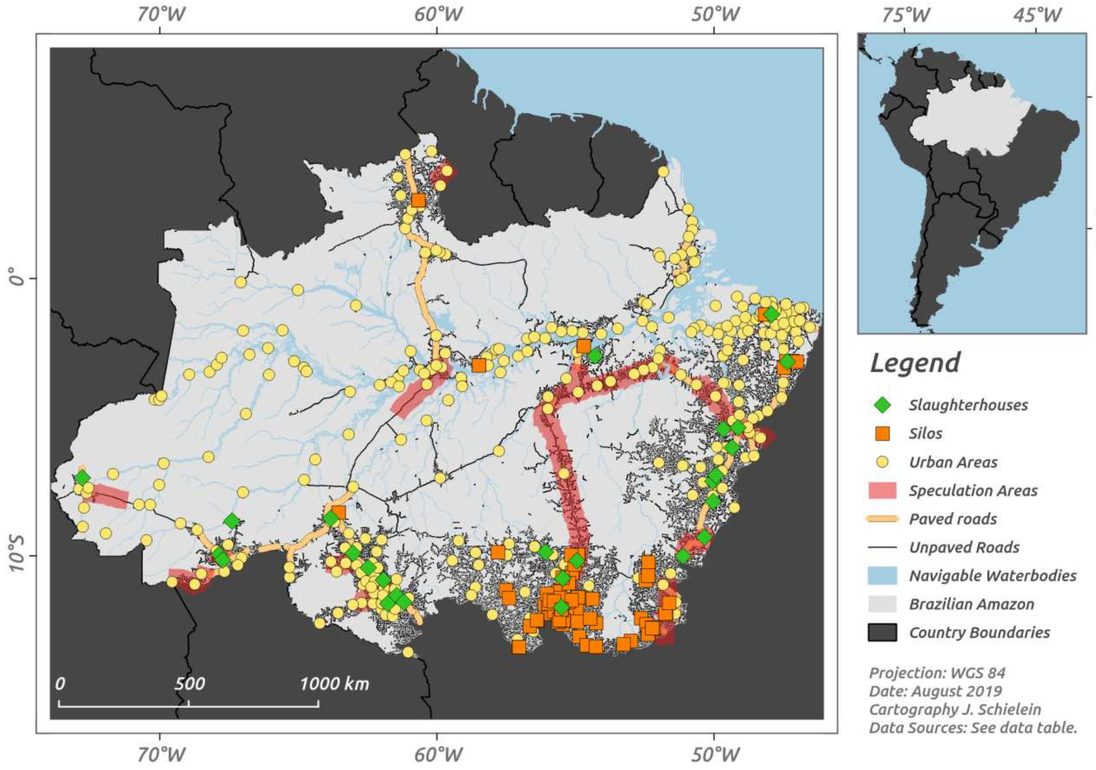


Figure 2.3: Study area for the accessibiltiy analysis and input data

For primary roads, we use official data obtained from DNIT and preprocessed by Miranda et al. (2019). The data set contains official roads and their surface type (paved/unpaved/undergoing paving) for each year from 2004 to 2012. We assign an average travel speed of 81 km/h for paved roads in both seasons. For unpaved roads we assign 49 km/h in the dry season and only 5 km/h in the wet season due to bad road conditions. Assumptions on travel-speed are based on an analysis of car logs from field inspection trips of the Brazilian Institute of the Environment and Renewable Natural Resources – IBAMA (see SM Section A.2 for details).

Unofficial road data was obtained from Soares-Filho, Moutinho, et al. (2010) and Barreto et al. (2017b). Both data sets were derived by visual interpretation of Landsat 7 ETM images with approximately 30 meters spatial resolution, without an accuracy assessment or detailed metadata description of either data set. This creates some degree of uncertainty regarding the reliability of both data sets and we, therefore, cross-checked them with very high-resolution (VHR) imagery on Google Earth Pro. For this we used 250 randomly selected plots of 250x250 meters and compared VHR data to the digitized data sets. In 93% of the cases, we could clearly identify the digitized roads as roads in the VHR images and only in 7% of the cases we found additional roads in the VHR data that were not included in either of the two data sets. This might be due to the higher resolution of VHR data, which allows detection of very small roads, or because the VHR images that are shown in Google Earth Pro were taken at a later point in time (2016-2018). Despite being incomplete, we believe that these data sets are an important source of information because they cover larger areas that are otherwise not covered by any road data set we know of.

The land cover data was obtained from the Mapbiomas Project in 30 meters resolution. We reclassified land cover classes assuming a travel speed of 15km/h for open landscapes such as agricultural areas (wet season 5km/h) and 3km/h for closed landscapes (e.g. forests) in both seasons. Details about all land cover classes and travel-speed assumptions can be found in the SM Section A.2. For the creation of travel maps in 2004 and 2012, we took the land cover data from 2004. This is done because the land cover data contains pastures and cropland areas which are the dependent variables of the regression model described below. Including those areas in the construction of the independent variables might lead to endogeneity in the estimation.

Our river-data set shows potentially navigable rivers which were derived from a digital elevation model in combination with satellite images. A description of the data processing steps can be found in the SM Section A.4 and the data is published in Schielein (2017). The rivers included in this data set have a minimum-width of approximately 5 meters and are potentially navigable with small to medium passenger boats. We assume an average travel speed of 14 km/h for both seasons. Besides rivers, travel-speed on all surface types was corrected for the effect of slope using a Digital Elevation Model with 30m spatial resolution. Details about the slope correction can be found in the SM, Section A.1.

2.3.3 Regression Model to Analyse Accessibility and LULCC

For our empirical estimation, we implemented a probabilistic regression model using our panel data information on land cover change, again obtained from the Map-

Table 2.1: Data sources utilized for analyzing accessibility and LULCC

Infrastructure Data

Name	*Type*	*Spatial scale*	*Source*
Primary road network 2004-2012	vector	-	DNIT
Secondary road network 2004	vector	30 meters	Soares-Filho et al., 2010
Secondary road network 2012	vector	30 meters	IMAZON 2012
Land-cover	raster	30 meters	Mapbiomas Project, 2019.
Rivers	raster	90 meters	Schielein, 2017.
Slope	raster	90 meters	Digital elevation model based on USGS, 2000.

Travel Destinations

Name	*Type*	*Spatial scale*	*Source*
Silos	vector	Lat/Lon	CONAB, 2019.
Cattle slaughterhouses	vector	Lat/Lon	Compilation of different sources (see SM)
Urban areas	vector	Lat/Lon	IBGE, 2010
Speculation areas	vector	Lat/Lon	Brazilian Department of Transportation infrastructure DNIT

Time-variant regression covariates

Name	*Type*	*Spatial scale*	*Source*
Price index for agricultural crops	tabular	Municipality	Miranda et al. 2019
GDP	tabular	Municipality	Gross Domestic Product in the nearest urban area based on IBGE, 2019b.
Population	tabular	Municipality	Urban Population in the nearest urban area based on IBGE, 2019c.
Protected areas	vector	unkown	Soares-Filho et al., 2006
Fines	vector	Lat/lon	SISCOM/IBAMA 2017
Sate dummies	tabular	-	IBGE 2010.

Biomas Project (2017). The outcome of our model shows whether pasture or crop creation occurred in a gridcell of 2x2 km. Using bivariate dependent variables does not allow us to implement a fixed-effects or first-difference model, as would be the case if we had a continuous dependent variable (Wooldridge 2001). To account for time invariant fixed effects on our estimation, we therefore use a conditional logit estimation (Baltagi 2008). Our empirical specification includes lagged control variables to reduce biased estimations due to the presence of endogeneity. Endogeneity might occur because in dynamic frontier areas, it is common to observe improved accessibility following an expansion of agricultural land (Chomitz, Buys, et al. 2007). This could result in a simultaneity bias. Therefore, we include accessibility estimates as well as all other relevant covariates for a period prior to the observed pasture or cropland creation. Furthermore, we also assume that reverse causality can only be contemporaneous and land conversion is only affected by previous accessibility changes (Bellemare, Masaki, and Pepinsky 2015). Our model of pasture creation takes the following form:

$$y_i^* = \Delta Access_i^{'}\beta + \Delta X_i^{'}\theta + Access_{i,0}\vartheta + \mu_i + u_i \tag{2.1}$$

where

y_i^* is a dummy variable for pasture or crop creation in a gridcell of 2x2 kilometers,

$\Delta Access_i^{'}$ is a vector of travel time changes,

$\Delta X_i^{'}$ is a vector of changes in control variables,

$Access_{i,0}$, is a vector of initial accessibility conditions,

μ_i are individual fixed effects,

β, θ, ϑ are the parameters to be estimated,

u_i is an error term.

The model captures the effect of changes in travel time between 2004 and 2012 ($\Delta Access_i^{'}$) on the probability to observe pasture or crop creation between 2012 and 2016 (y_i^*). Our dependent variable is dichotomous and equal to one (1) when the creation of new pastures or cropland is observed in a gridcell, and zero (0) otherwise. Gridcells are 2x2 kilometers in dimension, and we randomly selected 10.000 grid cells from the entire data set. A random sample (SRS) is drawn (instead of using all observations) to avoid the problem of spatial auto-correlation (Nelson and Hellerstein 1997).

The temporal structure that is proposed by our model reflects that farmers' land use decisions are formed and implemented over several years in response to changes in accessibility. Our main model measures pasture and crop creation in a time-frame of five years, however, we also test different time-specifications to assess the significance and strength of the accessibility coefficients for different temporal structures (see SM, Section A.8).

Following previous work, we also include other time-variant covariates in our model ($\Delta X_i^{'}$) that might have an influence on our outcome. These include agricultural prices, the protection status of a given area, fines for environmental crimes, as well as population and GDP growth in the closest urban area.

Time-invariant unobserved characteristics are dealt with by using a conditional logit specification that controls for individual fixed effects, hence there is no obligation to specify time-invariant controls. Nevertheless, Jalan and Ravallion (1998) argue that time-invariant controls may be included in an estimation if the outcome is likely to be affected by pre-treatment local conditions. We, therefore, include a measure of accessibility into our model that captures accessibility in 2004, before any improvements between 2004 and 2012. The rationale behind this step is that, for example, a reduction of travel time to a certain market by 1 hour has a different effect on the likelihood of conversion in an area that is already close to the market, compared to an area that is very distant (e.g. several days away). These initial accessibility conditions are captured in the term ($Access_{i,0}$).

We tested the model above in four different scenarios to assess how different travel seasons influence LULCC:

1. Model *PastureDry* estimates pasture creation between 2012 and 2016 under dry season travel conditions

2. Model *PastureWet* estimates pasture creation between 2012 and 2016 under wet season travel conditions

3. Model *CropDry* estimates crop creation between 2012 and 2016 under dry season travel conditions

4. Model *CropWet* estimates crop creation between 2012 and 2016 under wet season travel conditions

Regarding the coefficient signs, we expect that the likelihood that areas are being converted to either pastures or cropland decreases with increasing travel time to their relevant markets (extended central market hypothesis). A reduction of travel-time to a relevant market between 2004 and 2012 should therefore have a positive effect on the likelihood of areas being converted to pasture (for slaughethouses) and cropslands (for silos). Regarding dry season and wet seasons differences, we expect that the coefficients for the dry season accessibility are larger if access is required during the whole year including the wet season (interaction hypothesis and risk reduction hypothesis). If the precise moment for travel does not matter, however, we expect both accessibility-coefficients to be more or less equal. A simple rationale for this assumption is given in the SM, Section A.6.

For traveltime to urban areas we expect the same associations as with markets, for both pasture and cropland models and we expect that the dry season coefficients are stronger because permanent access is required (permanent interaction hypothesis).Lastly, we expect that a reduction in traveltime to speculation areas increases the likelihood of area conversion. However, we expect this effect to be only significant in the pasture model, because pastures are used to quickly consolidate larger tracks of land (speculation hypothesis).

2.4 Results

The results of our accessibility mapping are presented in Figure 2.5. In general, they show a south-north and east-west gradient in accessibility regarding almost all travel destinations and time specifications. This pattern originates from the existing infrastructure that was historically conceived to connect the Amazon region to Brazil's southern and eastern regions. Most of the primary and secondary road network and most of the agricultural infrastructure is concentrated in these regions. The only notable exception is the map of travel times to urban areas in the wet season where the influence of the secondary road network is limited and the travel destinations are distributed more evenly.

The maps show pronounced differences between wet season and dry season accessibility, indicating that very few areas in the Brazilian Amazon can count on reliable all-year transportation infrastructure. The maximum travel time of 10,621 minutes (7,4 days) is achieved from the most distant point in our sample to the closest slaughterhouse in the wet season.

The maps also show pronounced local accessibility differences between 2004 and 2012. The largest differences are attributable to the creation of new travel destinations e.g. the opening of new slaughterhouses in the north of the research area. Smaller differences are attributable to the paving of roads. In general, the maps highlight increasing accessibility in the dry season as compared to the wet season, and a continuous improvement of accessibility in most parts of the research area.

Figure 2.4 is a coefficient plot which displays the results of our regression. Coefficients are plotted against a common scale as dots with their confidence intervals as horizontal lines around them (horizontal dot-whisker plots, see Kastellec and Leoni (2007)). To create those plots, we used the dotWhisker package in R (Solt and Hu 2015). Figure 2.4 shows only travel time change covariates ($\Delta Access_i^{'}$) for each model. All other covariates can be seen in the regression tables in the SM, Section A.7. The significance level for the plots is set to .05. For the interpretation, this means that only such coefficients are significant at the .05 level, whose confidence intervals do not cross the vertical interception line at zero (0). Coefficients with their confidence interval to the left of this line are negatively associated with the outcome, whereas coefficients with their confidence interval to the right are positively associated. The coefficients in the plot can be interpreted as the average marginal change in the odds of area conversion to pasture (or crops) for one hour of travel time improvement. The logarithmized odds-ratios on the x-axis were converted to probabilities to facilitate the interpretation.

Regarding market accessibility, we can observe that a one hour improvement in travel time to slaughterhouses between 2004 and 2012, increases the likelihood of pasture conversion by 2.5 Percentage Points (pp) on average in the dry season and 1.2 pp in the wet season. The same holds true for the effect of travel time reduction to silos on crop conversion, however, with more pronounced average effects (14 pp for the wet season and 54 pp for the dry season). These results confirm our extended central market hypothesis that cattle ranching and crop production are both dependent on access to their own type of central market.

Comparing crop and pasture models, we also see that the coefficients for dry season and wet season models differ for the market-related travel time reduction coefficients (slaughterhouses and silos). However, we find statistically significant differences only in the crop models, where dry season and wet season confidence intervals do not overlap at all. This confirms our risk-reduction hypothesis that, in contrast to cattle ranching, crop production is preferably done in permanently accessible areas. The initial condition coefficients show the same effect, albeit more pronounced than the travel time reduction covariates (see, SM, Section A.7)

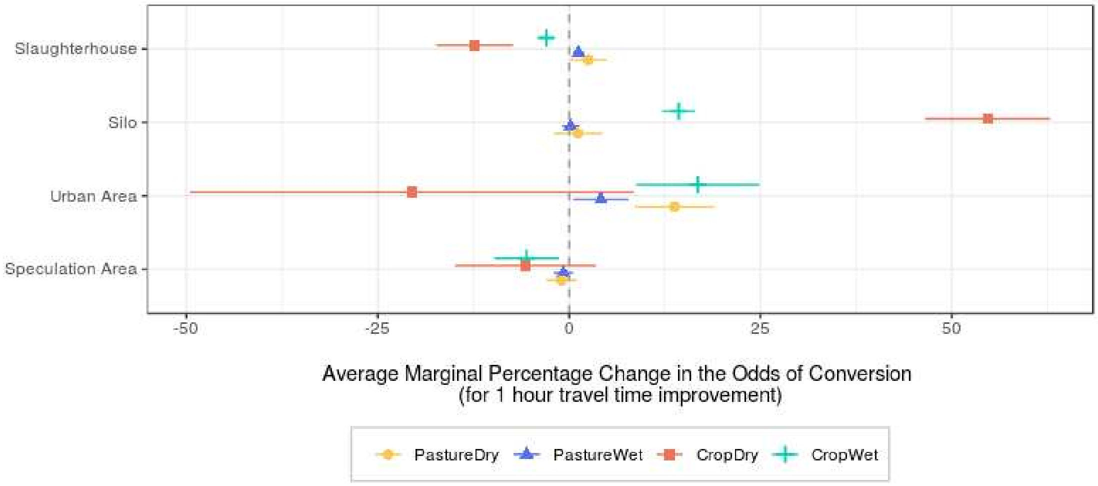


Figure 2.4: Coefficient plot for model results

Access to urban areas is positive and significant for both pasture models with the coefficient for dry season improvements being considerably larger (13.6 pp) than the coefficient for wet season improvements (4.1 pp), which confirms our permanent interaction hypothesis, stating that urban areas serve as an important social interaction arena requiring year-round access. For crops, we cannot confirm this hypothesis, however. Lastly, we hypothesized that, in contrast to crop production, cattle ranching is more prevalent where road infrastructure improvements are expected in the near future (speculation hypothesis). We could not find evidence for this hypothesis from our data. The coefficients of all other covariates can be seen in the SM, in table A.2. Also, in the SM the reader can find the results of our robustness tests regarding different temporal specifications which indicate stable outcomes if different periods for the outcome variable are considered (Section A.8).

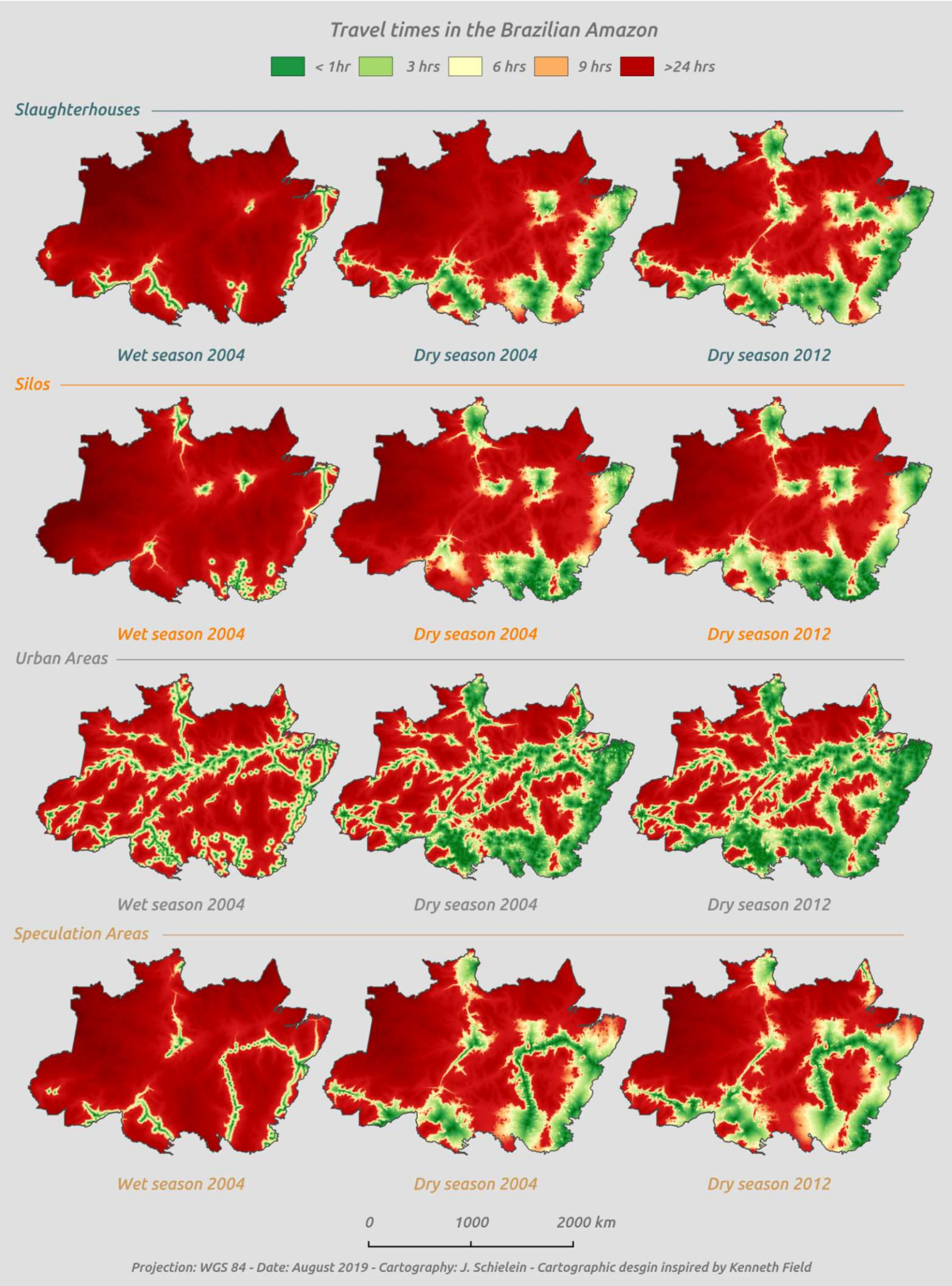


Figure 2.5: Traveltime maps in the Brazilian Amazon

2.5 Discussion and Policy Implications

Our results generally confirm that accessibility plays a dominant role in shaping anthropogenic landscapes in the Amazon. Like other authors, we observe that accessibility improvements increase the likelihood of conversion from natural landscapes to pastures or crops (e.g. Tucker et al. 2005; Chomitz, Buys, et al. 2007). Similar to Frey et al. (2018), we find evidence that land uses respond in idiosyncratic ways to changes in relevant access conditions and that preferences with respect to the timing and frequency of travel also play an important role.

Our results confirm that pasture and crop production systems need access to different marketplaces (slaughterhouses and silos) and that this access is required during the whole year for crops, whereas pasture systems are more flexible. We argue that these differences are plausible given the fact that all-weather access reduces the risks of harvest-losses at the end of the wet season in the case of crop production. Furthermore, coefficients for travel time improvements were significantly stronger in the case of crop production, which also indicates that crop production is likely to be much more affected by infrastructure conditions than pasture systems. As for pasture systems, researchers and policy makers should not only pay attention to road construction as a factor of LULCC, but more importantly to the opening of new slaughterhouses, which reduces travel time much more drastically than the paving of roads.

We also found evidence that year round access to urban areas is especially important for pasture production systems, whereas no such effect was detectable for crop production. This result was not in line with our expectations, but would appear plausible in our study area for the following two reasons.

First, in 2006 around 12% of cattle ranchers in the Amazon lived in urban areas and preferred to travel to their production sites only during the week or sometimes only on weekends (IBGE 2009). Cattle ranching is in this context often used as an additional income source to urban employment (54% of ranchers, ibid) or as a part of individual retirement plans for people living in urban areas. A common phenomenon in Brazil is that urban workers possess small farms (port. "sitios") which are administered by a hired farm-worker (port. "peão") who takes care of a small herd of animals (59% of all farms, ibid).

Second, crop production in the Amazon is often organized on an industrial scale with multi-functional, on-site farm-infrastructure, and permanent professional employees to supervise larger farms. Owners of these farms tend to live in larger urban areas and manage their farms remotely or with a few visits throughout the year. Such owners may be less dependent on all year access to the next urban area, especially if their production system allows for low cost post-harvest infrastructure, such as storage facilities.

Our findings are plausible and robust to alternative temporal specifications (see SM, Section A.8). Consistency among initial travel time and improvement coefficients further increases trust in the results of our analysis.

Researchers focusing on the quantification of LULCC are well advised to carefully think about how to conceptualize and measure accessibility. Access to different travel destination (where to) and local travel conditions at different moments in time (when) play an important role to explain spatial patterns of LUCC in the Amazon. The AccessiblityMaps package for the R statistical programming language is intended to facilitate future work along these lines.

Finally, it is time to acknowledge that infrastructure investments in the world's remaining tropical forest regions are as important for economic development as they are for conservation. As such, they should be regarded as cross-sectoral policy measures with considerable potential to design multi-functional landscapes that attend to the needs of modern agricultural systems and provide to globally and locally valued ecosystem services (Celentano et al. 2012; Sayer et al. 2013; Guedes et al. 2014).

CHAPTER 3

Recent Transformations of Land Use and Land Cover Dynamics Across Different Deforestation Frontiers in the Brazilian Amazon

An article with similar content including Appendix B has been published as Schielein, J. and J. Börner (2018). "Recent transformations of land use and land cover dynamics across different deforestation frontiers in the Brazilian Amazon". In: Land Use Policy 76.January, pp. 81–94. issn: 02648377. doi:10.1016/j.landusepol.2018.04.052.

3.1 Introduction

Frontier theory is a prominent conceptual framework to analyze and describe the dynamics of LULCC in tropical rain-forest areas (Faris 1999; De Koninck 2000; Entwisle et al. 2008; Barbier 2012; Pacheco 2012). Frontiers can be generally framed as "regions just beyond or at the edge of human settlement" (Merriam-Webster 2004), where multiple land-conversion processes take place that are characterized by the substitution of natural vegetation with domesticated plants for food and feed production. Deforestation is among the most commonly studied phenomena in the frontier literature, as it is often associated with negative impacts on the global climate, biodiversity, and local as well as regional hydrological cycles (Werth and Avissar 2002; Fearnside 2005; Foley et al. 2005; IPCC 2013).

One strength of frontier theory is the focus on a set of processes and underlying causal relationships that are specific to remote areas with distinct geographical and social characteristics, for example, in studies on deforestation and environmental change. In the past three decades, multiple theoretical approaches have been proposed to link different causal drivers to frontier development at local scale. This theoretical diversity has become a powerful toolkit to describe LULCC trajectories across different socio-economic settings, for example, in the Amazon region (Fearnside 2001, Walker et al. 2002, Pacheco 2005, Aldrich et al. 2006, Jepson 2006, Caldas et al. 2007, VanWey, D'Antona, and Brondízio 2007, Browder et al. 2008, Michalski, Metzger, and Peres 2010, Carrero and Fearnside 2011).

Case-study based explanatory richness comes with a desirable conceptual pluralism, but it faces limitations in terms of representativeness and cross-case comparability. However, given the technical, financial, and political constraints of policy making at national scale, decision-makers can benefit from spatially explicit landscape and regional scale approaches, when targeting heterogeneous frontier conditions (Sayer et al. 2013). Pacheco and Poccard-Chapuis (2012), for example, have mapped deforestation frontiers in the Brazilian Amazon describing frontiers in terms of land cover and actor characteristics, using data aggregated at municipality level. They classify frontiers based on their deforestation levels and identify priority areas for policy action. Along a gradient of deforestation, however, frontier theory suggests considerable changes in the relative importance of the proximate and underlying causes of deforestation (Angelsen 2007, Walker 2012). Policy design may thus benefit from a theory informed selection of classification variables.

A number of recent studies have, moreover, analyzed policy effects on recent deforestation dynamics at regional scale (Assunção, Gandour, and Rocha 2015, Arima et al. 2014, Börner, Wunder, Wertz-Kanounnikoff, et al. 2014, Gibbs, Munger, et al. 2015, Gibbs, Rausch, et al. 2015).These studies have generally found that the Brazilian government adopted effective policy instruments after Amazon deforestation rates had peaked between 2003 and 2004. However, designing a policy mix that promotes long-term sustainable forest transition pathways requires a more comprehensive understanding of how land use systems have changed in response to forest governance reform (Barbier, Burgess, and Grainger 2010).

Hence, this paper aims to inform national and subnational decision makers by (1) developing a theory-based, spatially explicit frontier classification for the Brazilian

Amazon, and (2) analyzing frontier-specific land use change dynamics for the period of 2005 until 2015. For this purpose, we have developed a theoretical framework that builds on key insights from frontier literature, as well as case study evidence from the Brazilian Amazon. Our framework, inspired by Von Thünen's model of agricultural land rents, is presented in section 3.2.

Our methodological approach and the construction of a spatial database to map frontier regions and quantify LULCC is presented in section 3.3. Results are presented in section 3.4, where we identify eight different frontier types, characterized by distinct LULCC dynamics and socio-economic characteristics. Although those frontier characteristics correspond to most of our theoretical expectations, we observe a predominance and persistence of cattle ranching in all frontier types, which deviates from frontier theory predictions. However, our results also indicate a frontier-wide paradigm-shift towards more intensive cattle production and an increase in annual crop production. Section 3.5 discusses our findings and theoretical shortcomings of our proposed model. Section 3.6 draws conclusions against the backdrop of the current political and economic context of land use change in Brazil and beyond.

3.2 Theoretical Background

The conversion of primary forest to cropland or pastures is commonly framed as an investment decision driven by expected returns to alternative uses or future appreciation of land values (Dias et al. 2016). However, as we move from well-established agricultural landscapes to remote frontier regions, expectations related to return on investment are based on very different socio-economic realities. Our conceptual framework assumes that market-distance and accessibility are a key underlying determinant of deforestation, and is therefore based on Johan Heinrich Von Thünen's model of agricultural land rents. In this model, Von Thünen assumes a homogeneous state characterized by a central market, uniform biophysical conditions, and no foreign trade. Transportation costs increase with distance to the market and agricultural goods have a limited durability. This leads to a distribution of land use classes that follow a pattern of concentric circles with more input-intensive and perishable cultures in the inner circles, and extensive land uses with less perishable products, such as livestock and wood production, in the outer circles around the market (Thünen 1910).

The standard Von Thünen model characterizes the distribution of land use classes in a steady state, where all land has already been converted and assigned to its most profitable purpose. This contrasts with the dynamic nature of land use change at forest frontiers, as infrastructure and market access improve over time. In the extended Von Thünen framework presented in Figure 3.1, agricultural land use expands geographically into more remote areas as the frontier develops and urban centers grow over time. The distribution of land-conversion is dependent on the profitability of different land use forms, which is dependent on its current distance to the next market and the local market demand (see Figure 3.1).

Following Figure 3.1, the spatial location of deforestation changes over time as frontier development advances, with the highest deforestation rates occurring in new frontier areas. However, since deforested area accumulates over time, more deforested areas are found in older frontiers compared to newer frontier regions. If new frontiers attract additional settlers, a similar demographic development pattern is

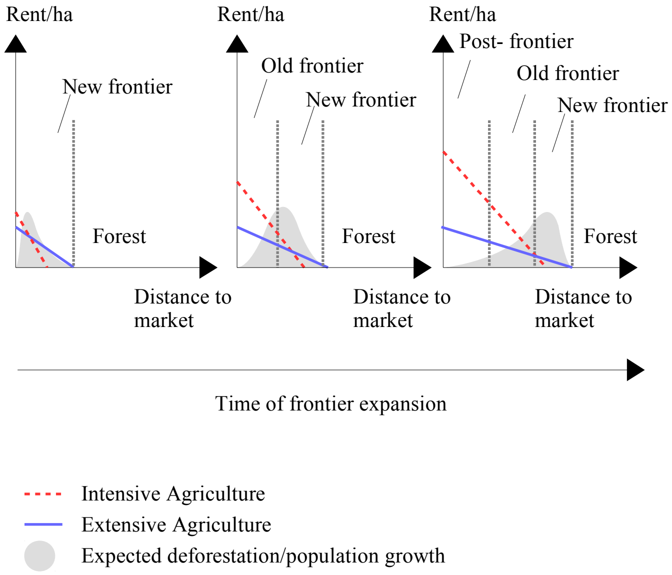


Figure 3.1: Poulation change and deforestation in a time-dependent land-rent model

expected, with population increases in recent deforestation frontiers and decreasing rural population growth after a certain amount of frontier development. This conjecture is in line with structuralist approaches of frontier theory, although the driving forces behind this development are rooted in economic, as well as demographic changes in local family structures over time (Perz 2003, VanWey, D'Antona, and Brondízio 2007, Browder et al. 2008). Consistent with the Von Thünen model, more intensive land use forms predominate regions that have better access to markets, and frontier development comes along with the replacement of extensive land use forms with more intensive land uses - especially in areas close to urban markets.

Despite infrastructure improvements and subsequent population growth, another important factor for frontier development is governance. Policy factors matter, because they can influence land-rents and thereby distort the development of deforestation frontiers (Barbier, Burgess, and Grainger 2010). In the context of Brazil, planned settlement projects play a crucial role in regional development and land conversion (Almeida 1992, Goza 1994, Moran 1997, Fearnside 2008, Pacheco 2009b). In addition to securing property rights, these projects were thought to come along with infrastructure investments and agricultural extension programs intended to provide settlers with credit, know-how, and production inputs. Despite widespread implementation failures (Pacheco 2009a), settlement projects were found to promote forest conversion due to higher land rents and policy-induced migration to remote rural areas (Schneider and Peres 2015). In addition to increased land conversion, we also expect, that the scale of individual deforestation activities is smaller in settlement projects than for other frontier types, because settlers tend to face binding capital and land constraints. Because of these limitations, settlement frontiers are dominated by subsistence-oriented production and/or extensive land use forms like cattle ranching.

The cultural background of settlers has repeatedly been subject to debates in the literature on Amazon colonization. Farmers, who migrate to the Amazon from the Brazilian South, were shown to prefer different production systems than peasants from the poorer north or north-east of the country (Pacheco 2005, Godar, Tizado, et al. 2012). Culture, capital asset endowment, and region-specific agricultural knowledge thus jointly influenced agricultural practices and associated land cover dynamics in Amazon settlements. Federal policies attended to the distinct needs of settlers by providing abundant land to poor northeastern peasants as well as entrepreneurial incentives, such as tax reductions and subsidized credits for commercially oriented southern cattle producers. Hence, we expect that land-conversion is more rapid, and production systems are more market-oriented and intense in settlement frontiers dominated by comparatively asset rich South Brazilian migrants.

Another important policy factor in the Brazilian Amazon are protected areas that either ban market-oriented production activities entirely, or strictly regulate the use of and access to land and forest resources. We expect less dynamic population and deforestation patterns in areas with high densities of protected land than in less regulated regions. Besides that, extensive land use forms are expected to dominate in and around protected areas since agricultural endeavors are more risky (if illegal) and generally more expensive because of higher transportation costs. Furthermore, protected areas are a good indicator for cultural heterogeneity and their corresponding, and often distinct, forest-transition pathways (Perz 2007). In the Brazilian Amazon, protected areas are commonly established to secure land claims of traditional population groups, thereby creating a space for culture-specific land use practices that differ from market-driven investment decisions. We also expect a higher presence of subsistence-oriented production forms, including shifting-cultivation, in areas with a dedicated protection status and, correspondingly, a higher share of secondary forest vegetation than in all other frontier regions (Vliet et al. 2013). Figure 3.2 gives a general overview of how we frame frontier successions in this research, and how policy factors might influence time and intensity of primary forest cover removal in pre-frontier areas.

In this graph, subsequent waves of frontier succession occur over time and policies influence the time and intensity under which a pre-frontier region is incorporated into the frontier development process. Previous studies had a working concept of the frontier as a zone of active land conversion from forest to agriculture (De Koninck 2000, Perz and Walker 2002, Rudel et al. 2005, Angelsen 2007, Caldas et al. 2007, Chomitz, Buys, et al. 2007, Browder et al. 2008). In consideration of the discussion above, we extend this concept and define frontiers generally as *zones of active land conversion with differing degrees of primary forest cover removal and different land use types that depend on the local accessibility, population density, cultural background and governance factors.*

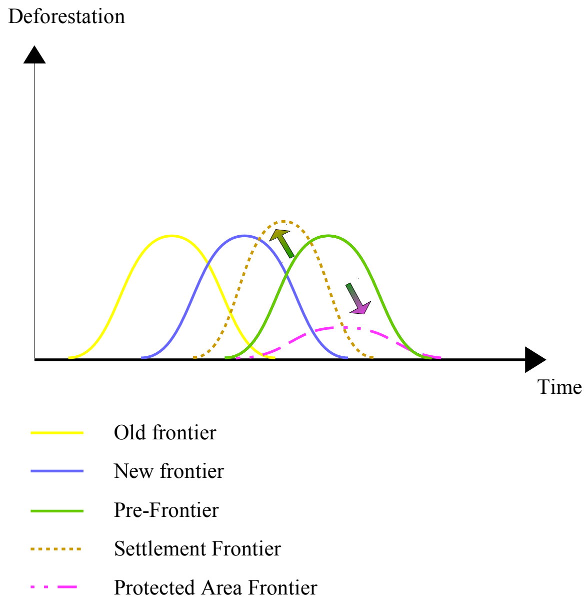


Figure 3.2: Frontier sucessions and the impact of policies on deforestation

3.3 Data and Methods

Our identification and characterization of frontier types in the Brazilian Amazon involves several analytical steps, which are summarized in Figure 3.3 and documented on the sections 3.3.1, 3.3.2 and 3.3.3 below.

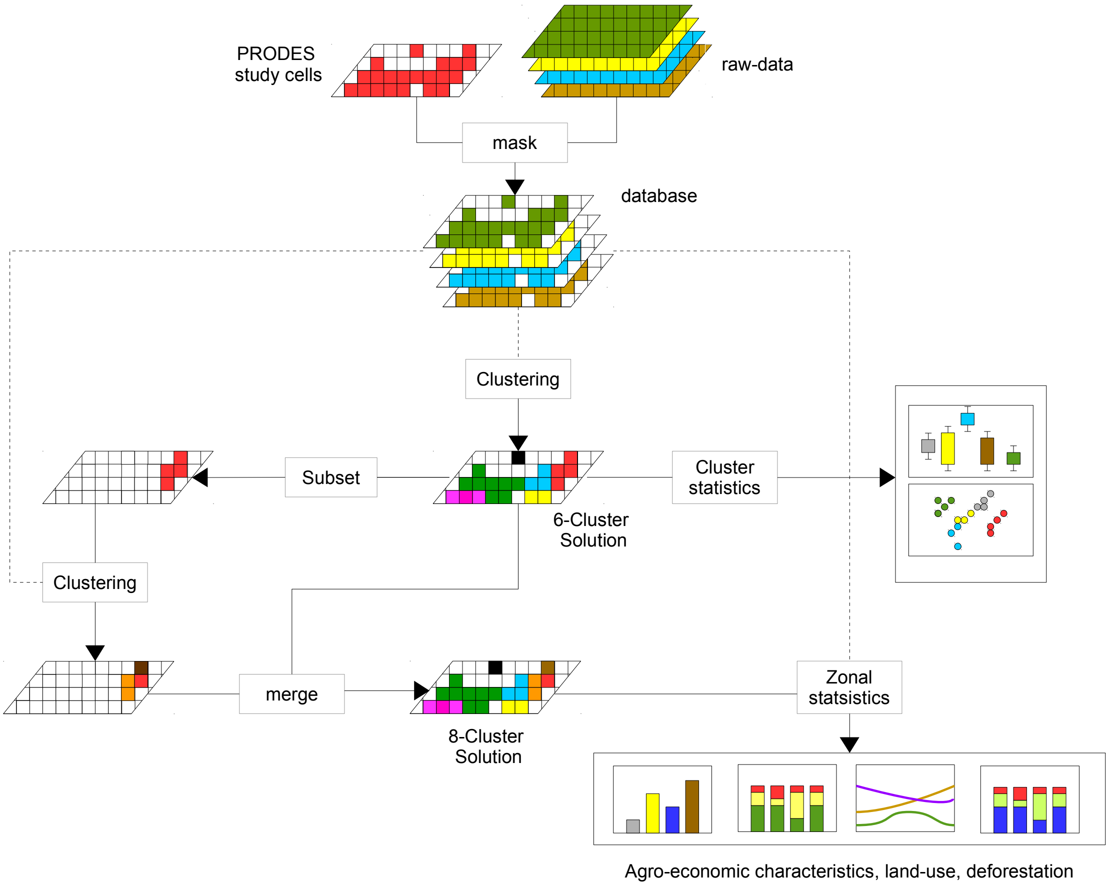


Figure 3.3: Overview of the undertaken analysis steps

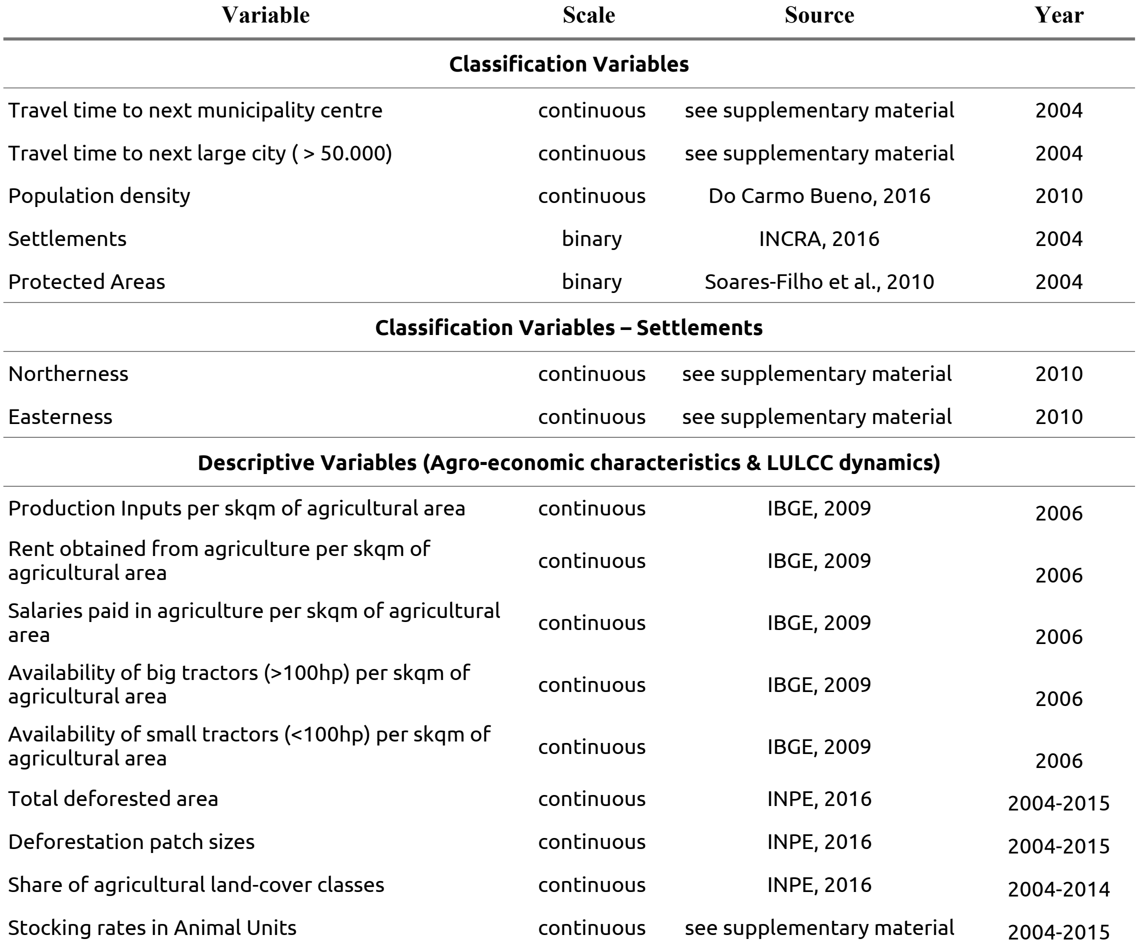
3.3.1 Spatial Database

Our study region comprises all areas with incidences of clear-cut deforestation in the Brazilian part of the Amazon Biome that were monitored by the Brazilian National Institute for Space Research (INPE) until 2015. We excluded quasi-natural landscapes from our sample in order to produce a frontier classification that appropriately differentiates among anthropogenized zones, as opposed to merely distinguishing forest from non-forest areas (see SM B.1). We chose a 1 x 1 km grid resolution as our unit of analysis to minimize the misleading aggregation of spatially heterogeneous deforestation pathways in larger administrative units, as discussed in Perz (2007). We have three groups of variables for our analysis (Table 3.1). The first group comprises the classification variables for the proposed frontier typology based on our methodological framework. The second group contains two variables to sub-classify the settlement cluster according to the origin of settlers. The third group contains variables that were used to descriptively analyze all frontier types in terms of their agro-economic characteristics and LULCC dynamics.

Two travel time maps were created to measure market accessibility. The first map contains accumulated travel times to the next municipality center in 2004, irrespective of its population size. The second map contains travel times to the next large urban market in 2004, which are Rio Branco and Cruzeiro do Sul in Acre. We provide a detailed description of the input data and processing steps to create the travel cost maps in the SM of this article (see SM B.2).For estimating the population density, we used high-resolution population data at the scale of 1x1km from the Brazilian Institute for Geography and Statistics - IBGE (see SM B.3).

Polygon data depicting the location of settlement projects in the Amazon was provided by the Brazilian Institute for Colonization and Agrarian Reform (INCRA 2016). After data cleaning, we converted the polygon data into a binary raster-map (see SM B.4). To describe the origin of settlers within a settlement project, two variables were created using IBGE Census data from 2010 (IBGE 2011). A variable called “northerness” indicates whether the settlers originate from the north or south of Brazil and a variable called “easterness” indicates whether they come from the east or west. Both variables can take values between 0 and 1. For the case of “northerness”, 1 represents a population that originates solely from the extreme north, whereas a 0 a population solely from the extreme south of Brazil. In the case of easterness, a 0 value represents population that originates solely from the extreme west, and 1 a population that is solely from the extreme east of Brazil. All values between 0 and 1 are proportional to their geographical distances in the coordinate space. For more information on how these variables were created, please refer to the SM B.5. As with settlements, we created a binary raster map for protected areas that shows if a study cell is either covered by a protected area or not, using a composed data-set by Soares-Filho, Moutinho, et al. (2010).

Table 3.1: Data sources utilized for mapping deforestation frontiers and LULCC in the Amazon



Variable	Scale	Source	Year
Classification Variables			
Travel time to next municipality centre	continuous	see supplementary material	2004
Travel time to next large city (> 50.000)	continuous	see supplementary material	2004
Population density	continuous	Do Carmo Bueno, 2016	2010
Settlements	binary	INCRA, 2016	2004
Protected Areas	binary	Soares-Filho et al., 2010	2004
Classification Variables – Settlements			
Northerness	continuous	see supplementary material	2010
Easterness	continuous	see supplementary material	2010
Descriptive Variables (Agro-economic characteristics & LULCC dynamics)			
Production Inputs per skqm of agricultural area	continuous	IBGE, 2009	2006
Rent obtained from agriculture per skqm of agricultural area	continuous	IBGE, 2009	2006
Salaries paid in agriculture per skqm of agricultural area	continuous	IBGE, 2009	2006
Availability of big tractors (>100hp) per skqm of agricultural area	continuous	IBGE, 2009	2006
Availability of small tractors (<100hp) per skqm of agricultural area	continuous	IBGE, 2009	2006
Total deforested area	continuous	INPE, 2016	2004-2015
Deforestation patch sizes	continuous	INPE, 2016	2004-2015
Share of agricultural land-cover classes	continuous	INPE, 2016	2004-2014
Stocking rates in Animal Units	continuous	see supplementary material	2004-2015

3.3.2 Clustering

The classification of frontier areas consists of a two-step clustering approach. The first step involved classifying the whole data-set based on the six main clustering variables. In the second step, the settlement-cluster from the first cluster-solution was extracted and the observations were reclassified according to their population origins (see Figure 3.3).

In both steps, the clustering variables were compiled in a raster-stack for our study area. Extracting those values from the raster yields in a data-set of 1.2 million valid observations. For the classification procedure we chose the CLARA Program for Clustering Large Applications (Kaufman and Rousseeuw 2005), which is available in the Cluster package in R (Maechler et al. 2017). The CLARA algorithm performs cluster-analyses on several sub-samples of the original data-set and later predicts the cluster-association of all observations based on the best clustering result. The sub-samples of the current study contained 2.400 observations. After drawing multiple samples from the original data-set, CLARA uses the Partitioning Around Medoids (PAM) algorithm to find stable structures for a specified number of clusters. To calculate the similarity between clustering-objects we used euclidean distances and standardized all variables (z-scores) before applying PAM (details in section B.6 of the SM).

The binary variables (settlements and protected areas) were treated as if they were numerical variables in the clustering process. This results in a strong influence of these variables on the overall clustering process. This desired effect identifies regions with peculiar governance regimes and corresponding implications for environmental law and land use policy design. We compared different cluster solutions with a total amount of 1 to 10 clusters to determine the optimal number of clusters. This number is based on the average silhouette widths of different cluster solutions. We provide a report on the clustering process in the results section with a cluster interpretation based on the variable characteristics of the clusters groups from the chosen classification.

3.3.3 Measuring Agro-economic Characteristics and Land Cover Change Dynamics inside Frontier Clusters

The first set of descriptive variables characterizes our frontier clusters in terms of their local agro-economic production conditions in 2006. We included statistics on available machinery (small and big tractors), utilized workforce, as well as acidic soil correctives, fertilizers, and pesticides (the latter three aggregated and labeled as "inputs"). These inputs indicate the degree of agricultural intensification in each cluster that is associated with frontier development as outlined above. To create a data-set that most closely matches our unit of analysis, census-tracts were utilized (IBGE 2009). Data processing steps are described in the SM (Section B.7). Two frontier clusters in remote areas were excluded from the descriptive analysis due to insufficient numbers of Census observations.

The second set of variables contains information on deforestation dynamics in different frontier clusters. We analyze the sum of deforested areas between 2004 and 2015, as well as size and complexity of deforested patches from the PRODES data-set (INPE 2019). A short description is given in the SM as well (Section B.8). The PRODES methodology defines forest as dense, open, or mixed ombrophilous forest, and deforestation refers to the complete removal of the forest cover in its primary form (port. "desmatamento corte raso"). The data does not cover forest degradation due to selective logging, hence the earliest stages of forest conversion are often undetected. INPE also developed a system for monitoring intermediate stages of forest degradation called DEGRAD which started in 2007 but was discontinued in 2013. We did not include DEGRAD data in our analysis because a quantitative assessment of DEGRAD shows an overall low reliability of DEGRAD areas as early indicators for subsequent clear-cut forest cover removal, with conversion rates ranging only between 1 and 12% (INPE 2017). Furthermore, the intermediate spatial resolution of DEGRAD (6.25 ha) is also unable to detect very early stages of forest conversion and selective logging and thus adds little value to our analysis.

The last set of variables describes land cover trends inside the frontier clusters. To calculate cluster-specific land cover statistics we use data from the TERRACLASS project for the years 2004, 2008, 2010, 2012 and 2014 (Coutinho et al. 2013). This study focuses its analysis on the main land cover types in deforested areas, such as annual agriculture, clean and degraded pastures, secondary vegetation, and mixed land uses. The latter is used as an indicator for small-scale and subsistence-oriented farming. Perennial cultures are not part of the TERRACLASS assessment and therefore omitted from our analysis. Summary statistics for the evolution of different land cover types from 2004 to 2014 were calculated for each cluster, in each year, using the zonal statistics function from the raster package in R (Hijmans 2019). To characterize the intensity of pasture systems, we look at the differences between the shares of clean and degraded pastures from the TERRACLASS base-data. Finally, we also include statistics on annual pasture stocking rates measured in animal units per pasture area. This variable was derived from a combination of data sources described in the SM of this article (Section B.12).

3.4 Results

3.4.1 Frontier Classification and Demographic Characteristics of Different Frontier Types

Our clustering variables exhibited sufficient data-agglomerations in terms of accessibility, population density, and governance to discriminate between different types of deforestation frontiers in the Brazilian Amazon. However, formally acceptable silhouette widths ($s > .5$) were obtained only for either four or six frontier clusters (see SM B.9). Despite the somewhat lower silhouette width, we focus on the six-cluster solution for the purpose of this analysis, because it describes rural areas at a more policy-relevant level of detail. Figure 3.4 represents the cluster-objects from the six cluster solution in 2-dimensional space. The higher dimensional-data is reduced by means of Principal Component Analysis (PCA), with plot dimensions given by the first two components (Pison, Struyf, and Rousseeuw 1999). The cluster-objects are visualized as points and the cluster-groups are visualized as ellipsoids around them. In the proposed six cluster solution, the two vectors represent 64% of the point variability. The horizontal axis can be best described as the accessibility and population dimension where population density and market accessibility increase from left to right. The vertical axis depicts the governance dimension where observations cluster in their respective governance categories (protected areas on top, no formal governance status in the middle and settlements at the bottom). Figure 3.4 provides several important insights.

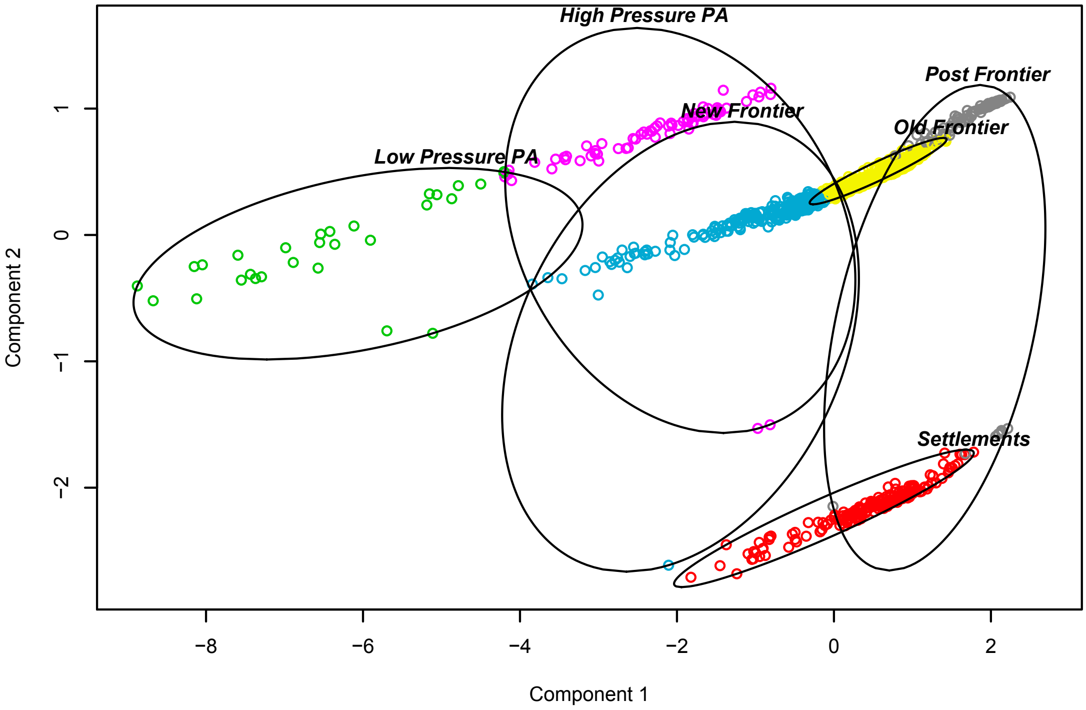


Figure 3.4: Clusterplot of the proposed six cluster solution from the first clustering step

First, the governance categories strongly separate the study cells given that they entered the cluster analysis as binary variables. However, there do exist exceptions such as more populated and accessible urban centers in the settlement cluster. Other important exceptions are very remote areas without particular governance status that appear to be most similar to remote protected areas due to their relative isolation and low population density. Note that Figure 3.4 shows only the sample of the CLARA algorithm with 1200 study-cells, i.e. extreme values are not necessarily statistical outliers.

Second, the accessibility/population dimension shows smooth transitions between the clusters instead of sharp boundaries. In other words, there is a continuum of frontiers in terms of these cluster dimensions, rather than strictly separable local realities. This is different in protected areas, where accessible and inaccessible regions were clustered into two distinct groups. This finding reflects the fact that some protected areas are designated to protect forests and native communities in regions of very high deforestation pressure, and others are created deliberately in regions of lower commercial interest, where protection is politically less contentious (see Pfaff 1999).

Sub-clustering was applied exclusively to the settlement cluster and revealed an average silhouette width above .6 with k=3. Given that the variables are derived from geographic coordinates, the silhouette width increases with k, as higher numbers of clusters more adequately represent geographic concentrations of the local populations. The three cluster-solution, nonetheless, best describes the population in broad geographical terms separating origins into the categories north, south, and north-east of Brazil.

The map in Figure 3.5 shows the spatial distribution of all frontier types after predicting the cluster associations of all grid-cells. Below we describe each cluster in terms of the cluster variables (for a graphical overview see SM B.9).

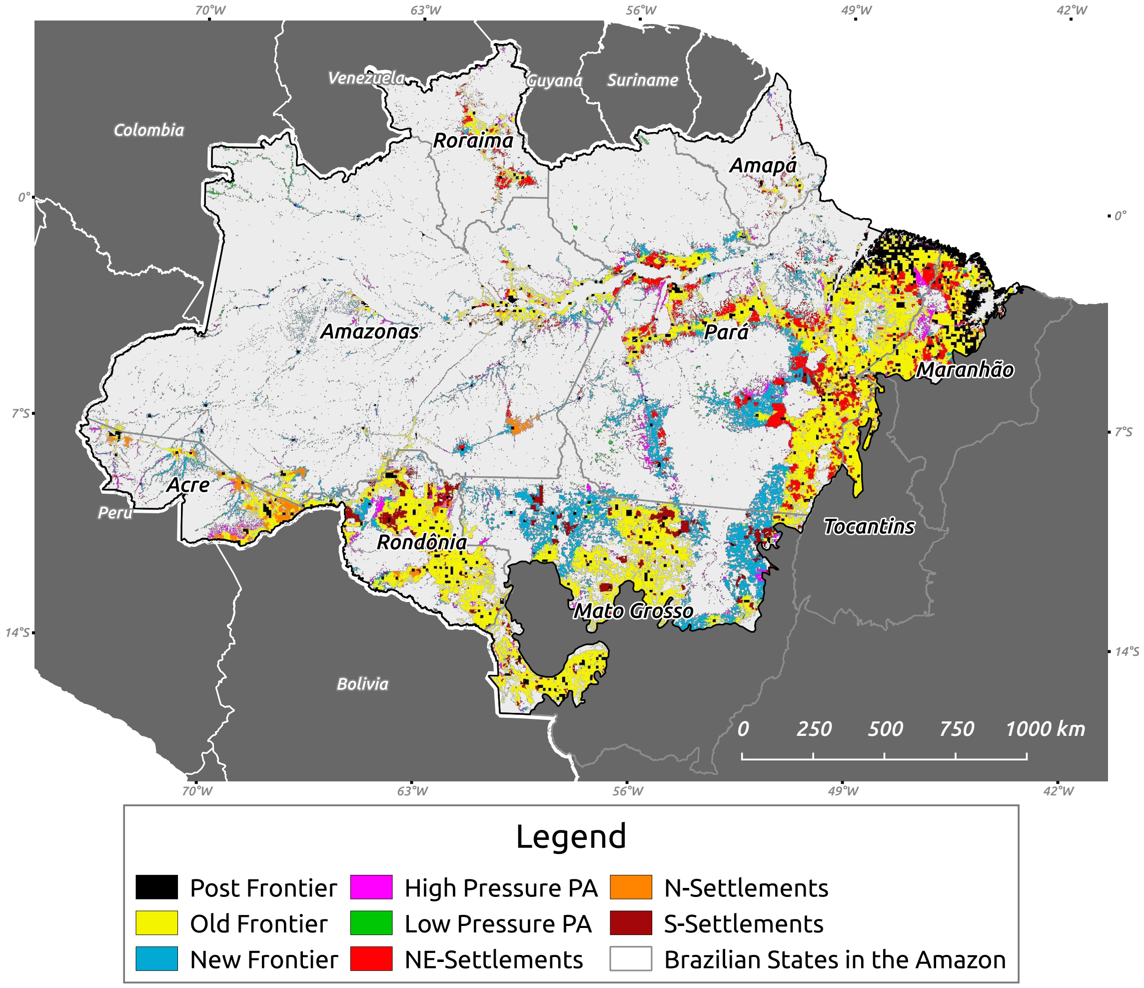


Figure 3.5: Map of current deforestation frontiers in the Brazilian Amazon biome

The Post Frontier boasts very short travel times to municipality center and urban markets, as well as high population densities. Post Frontier areas can be found across the whole region and are either urbanized or at the rural-urban interface. The Old Frontier is comparable to the Post Frontier in terms of its accessibility. However, comprised largely by rural areas, the Old Frontier has a much lower population density than the Post Frontier. Old Frontiers engulf Post Frontier areas, or main transportation routes, such as the eastern part of the Transamazon Highway and the Amazon river. They cover large areas of eastern Acre, Rondônia, central Mato Grosso, eastern Pará, northern Maranhão, and northern Tocantins.

Areas that are characterized by low accessibility are labeled as the New Frontier. They are not only more remote to large urban markets, but also to smaller towns. This is because they rely upon unpaved roads and rivers as main routes of transportation. New Frontier areas are typically less densely populated than Old

Frontiers. From a geographical perspective, it can be observed that there is often a spatial transition from Post Frontier, to Old Frontier, to New Frontier areas and then to protected areas. This spatial distribution reflects the Von Thünen logic of a decreasing land use intensity as we move from central markets to remote areas.

We denominate regions inside protected areas that are well accessible as High Pressure Protected Areas (short: High Pressure PA). They are often found adjacent to the boarder of a protected area and are frequently logged and occupied by agents from other frontiers. Invasions are numerous in those areas and a recent report from IMAZON described prominent cases, such as the National Forst Jamanxim, located next to the BR-163 in southern Pará, the National Forest Bom Futuro and the Extractive Reserve Jaci-Paraná in northern Rôndonia (Araújo et al. 2017).

In some High Pressure PA areas, deforestation is more likely caused by reserve dwellers alone or by both internal and external actors. These dynamics can be seen in the extractive Chico Mendes Reserve in Acre, where land has been converted for market-oriented cattle ranching by outsiders, as well as locally based rubber tapping communities (Vadjunec, Gomes, and Ludewigs 2009). Furthermore, cases exist where protected areas are highly accessible and forest conversion is part of traditional land use practices. This is, for example, the case in the Xingu indigenous area, which is surrounded by infrastructure and agricultural land and where the core area is consequently more accessible than in almost all other protected areas. Despite invaded boarder regions, considerable parts of deforested areas inside the Xingu core region were labeled as a High Pressure PAs and should therefore be interpreted with caution.

Low Pressure Protected Areas (short Low Pressure PAs) are relatively inaccessible protected areas with low population densities. Many Low Pressure PAs may not constitute frontiers in the classical sense, because their rapid occupation from outside is rather unlikely in the near future. Most Low Pressure PA study-cells lie in the states of Amazonas and Pará.

Settlement frontiers are found in the whole Amazon basin except the extreme Northwest. Most settlers who originated from the south of Brazil occupy settlements in the southern Amazon (geographically closer to their origins), which were denominated as the Southern-Settlement Frontier (short S-Settlement Frontier). Migrants from the north-east tend to occupy settlements in the east which were labeled as North-Eastern-Settlement Frontier (short NE-Settlement Frontier). Settlers from the north are often found in the west and in the center of the study region. Those areas were denominated as Northern-Settlement Frontier (short N-Settlement Frontier). However, this trend is not exclusive. Areas of the S-Settlement Frontier can be found in Marabá (eastern Pará), areas of the N-Settlement Frontier in central Rondônia, and NE-Settlement Frontiers can be found along the BR-163 highway, the Transamazon Highway, and in Roraima.

3.4.2 Deforestation Trends in Different Frontier Regions

A key purpose of our theory-informed classification exercise was to inform policymakers about how forest loss dynamics coincide with frontier development. It is worth noting again that the overall deforestation trend was decreasing in the study period until 2012, and has since slightly increased. We find that the frontier clusters described above have contributed to the regional trends in ways that are consistent with our theoretical framework following a distinct pattern of frontier successions.

First, the share in total deforestation changes over time according to the frontier status of different regions. Figure 3.6 illustrates the share of total deforestation for each frontier type over time including the year 2013 when the New Frontier surpasses the Old Frontier. Furthermore, the share of deforested areas increases considerably in the N-Settlement Frontier and NE-Settlement Frontier, and less so in High Pressure PAs. The S-Settlement Frontier shows less pronounced increases when compared to the other settlement frontiers, whereas the participation of the Post Frontier in overall deforestation decreases by a small degree. Recent deforestation increases are, therefore, driven largely by newer frontier areas.

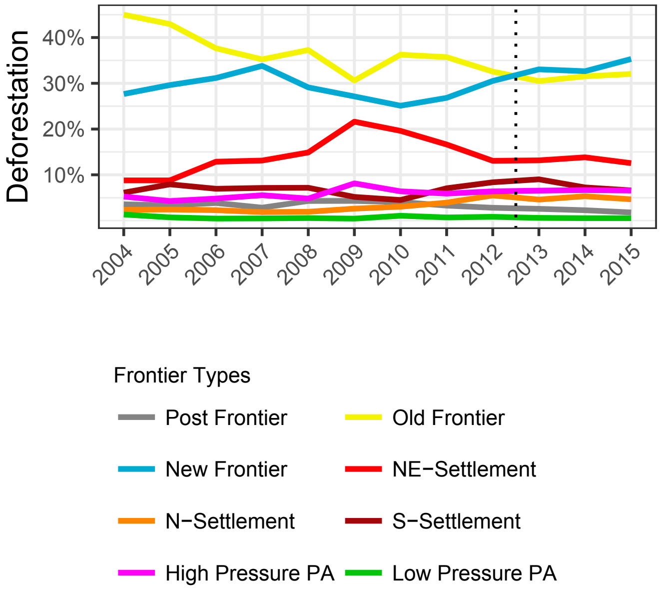


Figure 3.6: Participation of different frontier types in total deforestation

Second, the amount of total deforested areas increases with different stages of frontier development. Figure 3.7 shows how much of the original forest cover was lost inside each frontier between 2004 and 2015. Box sizes in Figure 3.7 describe the variability of deforestation around the median. As expected, the Post Frontier exhibits a high degree of accumulated deforestation, with 50% of its observations ranging between 40-90% of total forest cover loss in 2004. The position of the median inside the box (70%), indicates that more areas are likely to exhibit above-average deforestation levels. Over the observed time period, the interquartile range shrinks, suggesting that a higher level of uniformity is achieved due to rising deforestation levels at the lower bound and relatively stable levels at the upper bound. Upper bound stability points to an advanced stage of forest transition, where no significant forest cover loss is expected to occur in the future. It is worth noting that deforestation levels around the upper bound of the Post Frontier are similar to historical forest transition points observed among European countries (Mather 1992). Old Frontier and Post Frontier exhibit comparable levels of high overall deforestation, however, the spread of the distribution is larger in 2004, with 26%-86% inside the box. The median (60%) is also closer to the upper bound and increases over time as the overall spread slightly decreases, indicating more homogeneous, high deforestation levels over time.

The New Frontier also meets our expectations with intermediate deforestation levels in 2004 of 8-62% inside the box. Compared to the Old Frontier, the New Frontier median is closer to the lower bound, and the spread increases until 2015 due to regions with higher deforestation levels. The evolution of deforestation shares suggests an early transition stage in most of the New Frontier areas.

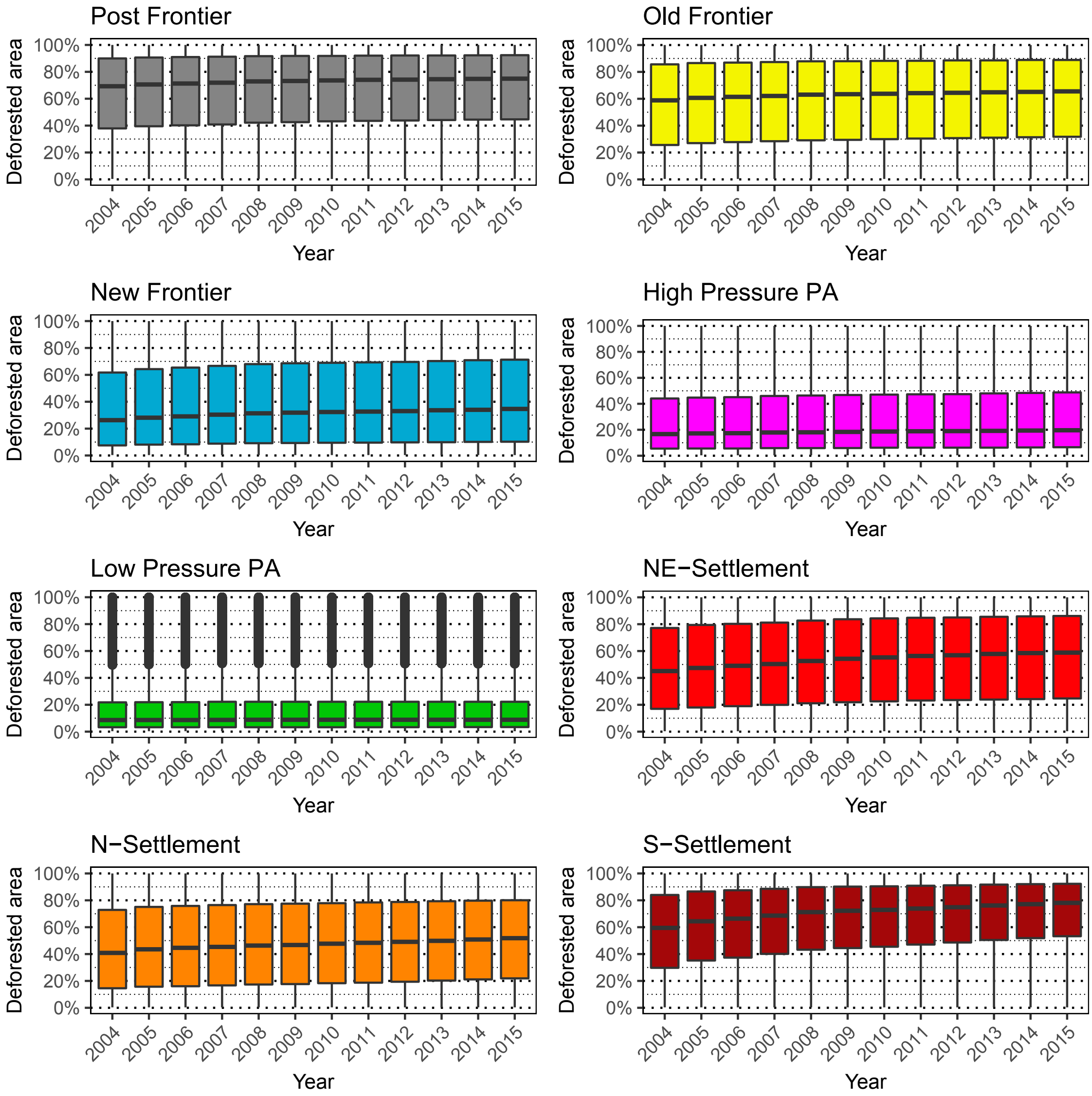


Figure 3.7: Total amount of deforested areas in different frontiers

High Pressure PAs exhibit higher deforestation levels than Low Pressure PAs, however, both have the overall lowest degree of total deforestation. Whereas Low Pressure PAs show no significant changes between 2004 and 2015, High Pressure PAs exhibit significant increases in the upper bound of the box and an increasing median - from 16% in 2004 to 20% in 2015, i.e. an early stage of frontier development. N-Settlement Frontier and NE-Settlement Frontier exhibit average levels of accumulated deforestation, but also the most pronounced increases between 2004 and 2015. The spread in both distributions does not change considerably indicating rising levels of forest loss at the lower and no sign of forest transition at the higher end.

The S-Settlement Frontier shows, as expected, higher overall conversion rates compared to the other settlement frontiers. However, the scope of forest conversion is much more pronounced than we expected in our theory section, where we assumed that settlements were generally more recent frontier regions. Deforestation trends in the S-Settlement Frontier are most comparable to the Post-Frontier, and there are converging levels of high deforestation in 2015 for the whole cluster.

Beyond overall deforestation trends, the patch-size of deforested areas also differs considerably across frontier types. Figure 3.8 shows that small-scale deforestation dominates settlement frontiers and the Post Frontier. In contrast, the overall participation of medium and larger patches is higher in the Old Frontier and New Frontier. However, the contribution of medium and large patches to overall deforestation was unexpectedly high in both High Pressure PAs and Low Pressure PAs. This is especially the case for the latter, where we would have expected more participation of small-scale deforestation in the overall deforestation composition. However, in the case of illegal territory invasion, the rapid establishment of large land claims after intrusion has been documented in the context of land speculation (Fearnside 2008). On the other hand, larger patches in these frontiers could also be cumulative agglomerations of small patches which were detected after several years due to cloud coverage, and could potentially be classified as a connected patch (for a discussion of these classification issues see SM Section B.11).

Beyond the aforementioned differences, we also observe changes in deforestation behavior across frontier types. Our data suggests a decreasing trend in deforestation patch-size until 2010, which is comparable to other studies (Rosa, Souza, and Ewers 2012). Between 2010 and 2015, a few years after the implementation of new forest law enforcement mechanisms in the Brazilian Amazon, the share of medium and large-scale deforestation increased again almost to 2004 levels (see Figure 3.8 for cluster-wise comparisons and, for basin-wide statistics, SM Section B.11).

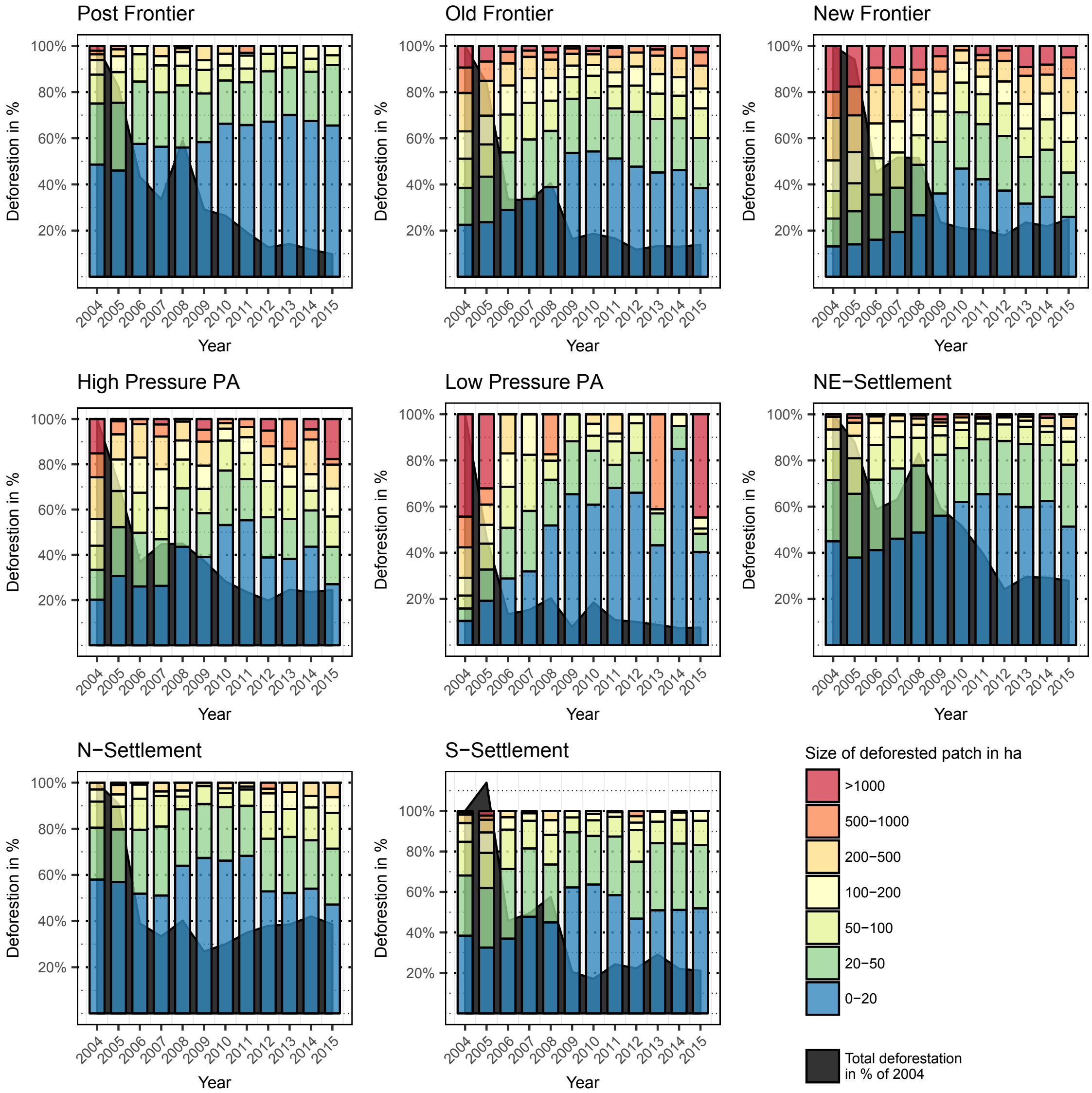


Figure 3.8: Patch sizes of clear-cut deforested areas in different frontier types

3.4.3 Land Use and Land Cover Change and Agro-economic Characteristics in Different Frontier Regions

In line with our theoretical expectations, intensive land uses in the form of annual agriculture show a higher presence in frontiers with good market access compared to more remote areas. Figure 3.9 gives a graphical overview of LULCC trends inside different frontier types between 2004 and 2014. Annual agriculture in 2004 is highest in the Old Frontier (4.5%) followed by the New Frontier (2.3%), the Post Frontier (2.1%), and S-Settlement Frontier (1.1%). Protected areas, N-Settlement Frontier and NE-Settlement Frontier exhibit practically no relevant annual agriculture in 2004 (see also SM B.10).

Mixed land uses have a higher share in the two protected areas and in the N-Settlement Frontier and NE-Settlement Frontier, which is a sign of small-scale and subsistence oriented agriculture. Furthermore, the share of degraded pastures in those regions is comparably high, indicating lower degrees of pasture management and intensification. As expected, Low Pressure PAs are dominated by secondary vegetation, which can be linked to patterns of traditional shifting-cultivation.

The land cover data also confirms our prediction that frontier development coincides with the increase of more intensive land use forms and/or the substitution of extensive land uses by more intensive ones. As expected, increases of annual agriculture are high in the Old Frontier (+3.4%), Post Frontier (+1.1%), and S-Settlement Frontier (+3.3%), and land use transition is much slower in the N-Settlement Frontier and NE-Settlement Frontier.

Unexpectedly, the New Frontier exhibited the second highest share in annual agriculture already in 2004 and, due to high growth rates between 2004 and 2014 (+ 5,8%), boasted the highest share of annual agriculture in 2014 (8,1%). Considering these figures compared to 8,0% annual agriculture in 2014 in the Old Frontier, and only 3.19% in the Post Frontier, this characteristic is probably driven by the strong growth of soybean agriculture in some of these frontiers and a trend in the integration of cattle and crop production systems. Also, the Post Frontier exhibits a high share of mixed land uses (7.2%), which could indicate that the market demand for the local consumption of input intensive products (fruits and vegetables) is mostly met by small-scale agriculture close to the urban markets as predicted by the Von Thünen model.

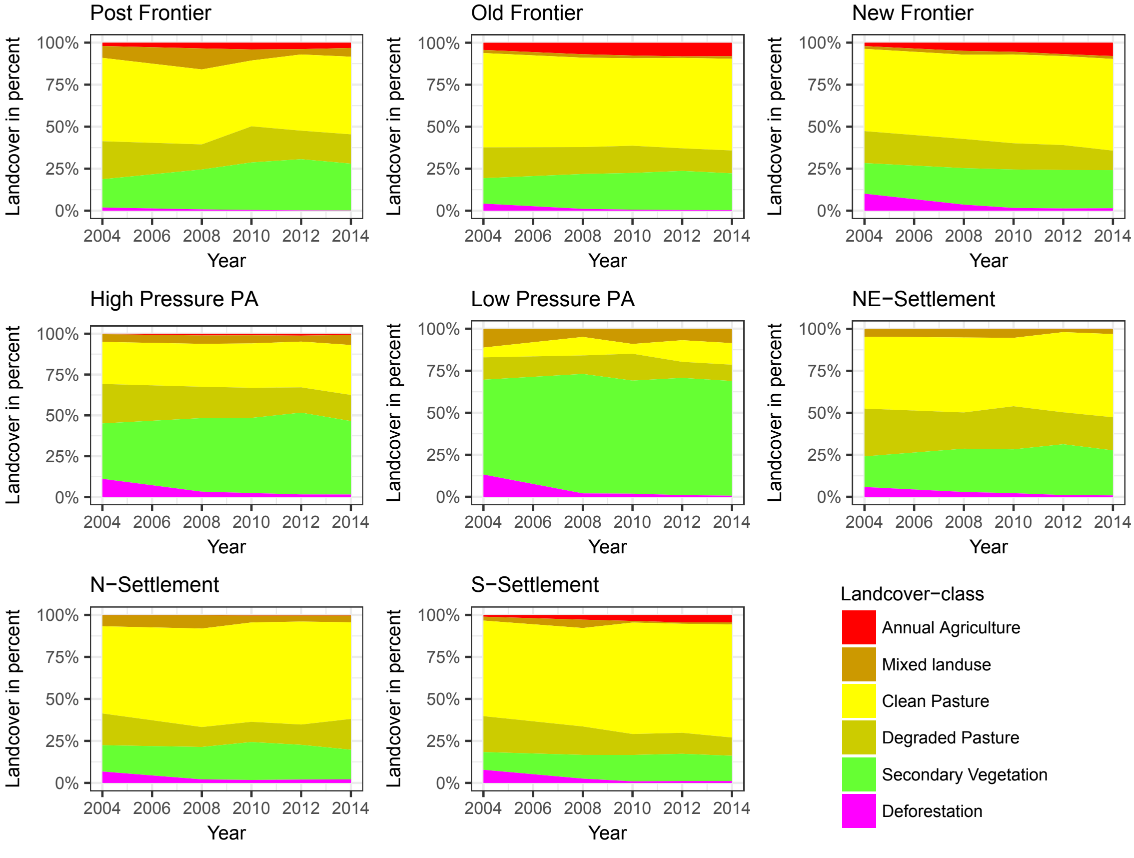


Figure 3.9: Land cover change in different frontier regions

Furthermore, degraded pastures decrease in practically all frontier types, which could either be associated with pasture reform, or land abandonment and subsequent forest regrowth. The total share of pastures (clean pastures and degraded pastures) dropped in almost all frontier types at rates between 5 and 10%, which is different from results obtained for previous study periods (Pacheco 2012). Given the well-documented dominance of pastures among post-deforestation land uses in the Brazilian Amazon, annual agriculture remains underrepresented in the land use mix across frontiers (Andersen 1996, Pfaff 1999, Barreto and Silva 2010, Bowman et al. 2012, Pacheco and Poccard-Chapuis 2012). This is not necessarily contradicting the Von Thünen logic, because our data base does not distinguish between all types of agricultural production and does not capture transformations that might occur over several decades to come.

Despite the substitution of degraded pastures with clean pastures, we do nonetheless observe an increase in stocking rates in the whole study area irrespective of the frontier type. Figure 3.10 illustrates that stocking rates increased on average from 1.3 Animal Units in 2003, to 1.7 Animal Units in 2015. As expected, stocking rates are the lowest in Low Pressure PAs and New Frontier regions, and higher in Old Frontiers and Post Frontiers. Interestingly, the S-Settlement Frontier and NE-Settlement Frontier exhibit the highest stocking rates suggesting either newer and more productive pastures or overgrazing. The latter is typically associated with pasture degradation and expansion, which might explain persistently high deforestation rates in those areas.

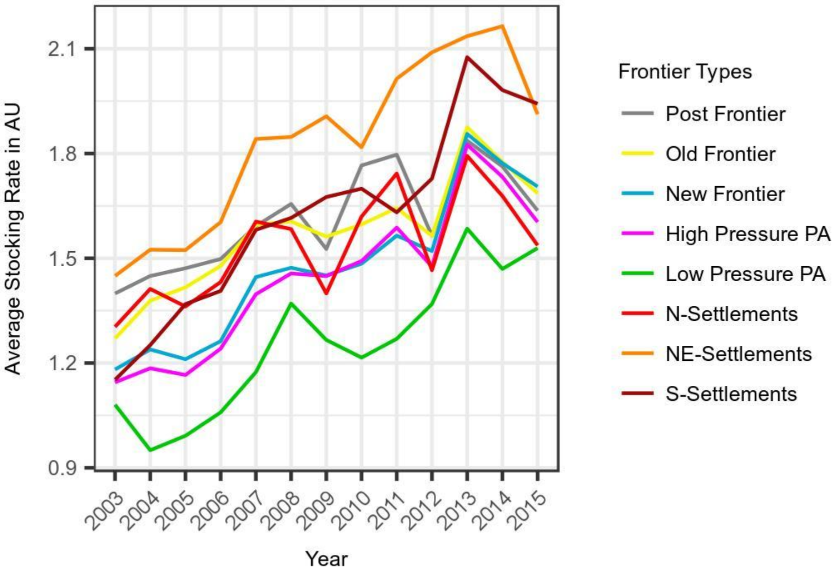


Figure 3.10: Stocking rates amongst different frontier regions

As expected, intensity and pace of land use transformation are also associated with different degrees of capitalization, agricultural input usage, and obtained rents amongst frontier types. Figure 3.11 gives a general overview of the agro-economic characteristics measured in each frontier in 2006, excluding two remote frontiers with insufficient numbers of observations. Post Frontier and Old Frontier exhibit higher densities of agricultural machinery with 7-8 big tractors per squarekilometer of agricultural land versus roughly 4 tractors in other frontier regions. Farmers in these zones rely more heavily on agricultural inputs, such as fertilizers, pesticides, and farm labor (measured in terms of per hectare hired labor costs). In the Post Frontier, the highest land rent was obtained per hectare (325 USD or 1,150 BRL on average) , which is in line with our expectations from the land-rent model. The New Frontier was comparatively less capitalized in terms of machinery, but was more dependent on hired labor - which is a requirement in annual agriculture. The second highest per hectare rents were obtained from agriculture (265 USD or 925 BRL per hectare on average), which can also be associated with the high share of annual agriculture in the New Frontiers' land use portfolio. N-Settlement Frontier and NE-Settlement Frontier relied less on machines, hired labor, and other agricultural inputs. Land rents obtained from agricultural activities were intermediate in those regions in 2006. A unique pattern is found for the S-Settlement Frontier where despite higher production inputs, very low per hectare rents were obtained from agriculture in 2006 (only around 104 USD or 363 BRL per hectare). Associated with lower levels of mechanization levels and paid workforce this finding probably reflects the strong predominance of cattle ranching activities at these frontiers.

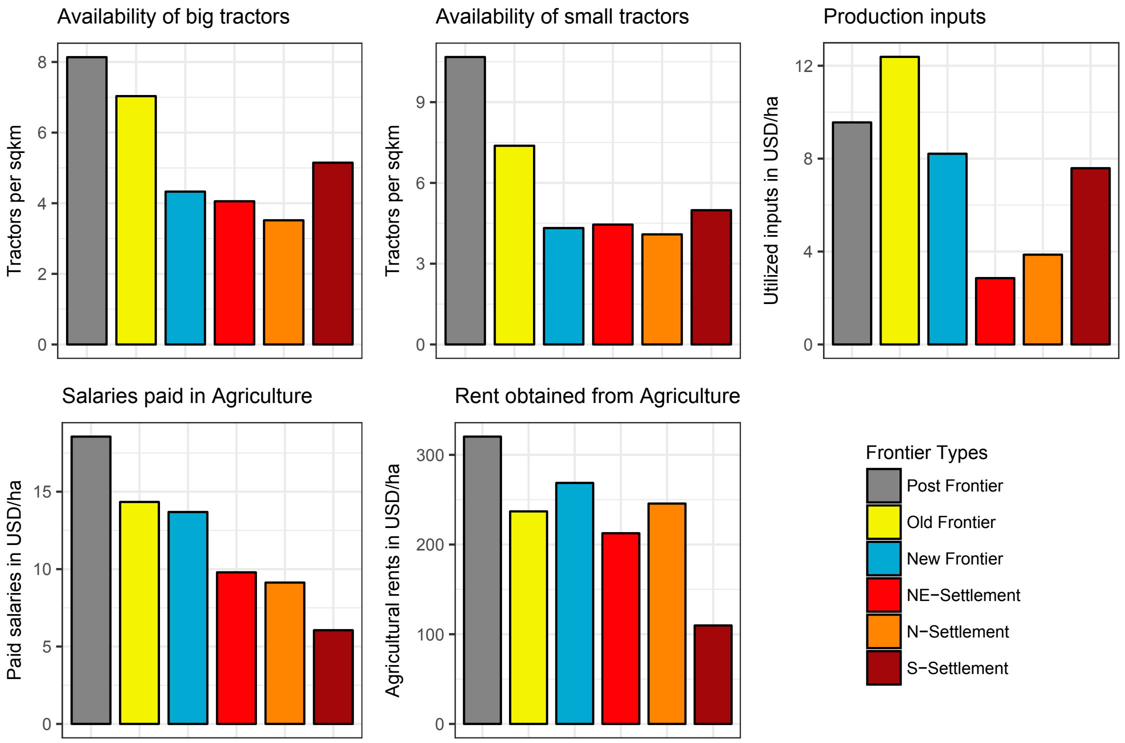


Figure 3.11: Agro-economic characteristics of different frontier types

In conclusion, land use systems are consistent with our characterization of land cover dynamics in the frontiers. Settlement Frontiers are dominated by smaller enterprises and family agriculture and a general persistence in cattle production. In contrast, the Old Frontier and New Frontier have a higher share of large-scale, capital-endowed agriculturists with increasing shares of annual crop production. Protected areas might be subject to land-speculation with large deforestation patterns and pasture establishment to corroborate land claims as well as patterns of shifting-cultivation characterized by secondary vegetation.

3.5 Discussion

Based on our theoretical framework that incorporates site-specific factors and integrates the land-rent model into frontier theory, we identify eight different frontier development types in the Brazilian Amazon using a spatially explicit and data driven classification procedure of anthropogenized landscapes. Our results confirm most, but not all, theoretical expectations with respect to policy-relevant outcome indicators, such as deforestation and agricultural intensification. First and exceptions aside, the land use mix follows an expected gradient of intensive (older frontier regions) to extensive (newer frontiers) land use. This gradient is moderate due to the predominance of cattle production in all frontier types. Contrary to previous studies finding cattle system expansion in all frontier stages (Pacheco and Poccard-Chapuis 2012), we observed a gradual substitution of extensive land use forms with more intensive land uses in almost all frontiers. Given that our study horizon covers a historical and well-documented forest governance reform in Brazil, this result may suggest an environmental governance-induced paradigm shift in modern frontier development in the Brazilian Amazon. While our focus was not on the mechanisms and impacts of this governance reform, we believe that our findings were modulated by two pivotal policy events in our study period: the 2004 Plan for Protection and Control of Deforestation in the Amazon (PPCDAm) and the reformulation of the Brazilian Forest Code in 2012.

Aided by improved monitoring-technology, the PPCDAm leveraged existing and command-and-control policies to deter illegal deforestation (MMA 2013). This led to more fines for illegal deforestation and area embargoes (from 2008 onwards) in areas with high deforestation pressure. These measures can be more effective in old frontier areas with transparent land tenure regimes, where the economic consequences of law enforcement and legal coercion are immanent and accessibility is improved. Conditions at these frontiers are conducive to the farm level responses we observe, such as diversification and the substitution of land-consuming cattle ranching by annual agriculture. Better law-enforcement also represents a plausible explanation for the increasing share of cleaned pastures in all frontier areas, because reforming degraded areas reduces the demand for new land from deforestation.

Additionally, law enforcement prioritized larger deforestation areas (>100 ha), which would explain the decreasing occurrence of large deforestation patches in older frontier regions and the persistence of small-scale deforestation in settlement frontiers up to 2012 (Börner, Kis-Katos, et al. 2015). Finally, the new Forest Code's much debated amnesty for land that was illegally deforested before 2008 may have partially watered down previous conservation commitments, but it also created incentives for legalized farms to convert degraded areas into clean, productive pastures in any of the analyzed frontier types. Further research should analyze these mechanisms in more detail. Despite a general paradigm shift, the pace and nature of agricultural intensification depends crucially on the frontier type. First, while annual agriculture increases mostly in the Post Frontier, Old Frontier, New Frontier, and S-Settlement Frontier, pasture related intensification (measured as an increase of clean pastures against degraded ones and a general increase in stocking rates) dominates all frontier types and protected areas. Simultaneously, secondary vegetation increases in all frontiers, which could indicate a land sparing effect of intensification decisions. Second, newer frontiers still experience more deforestation than older frontiers, and the scale (average patch size) of deforestation has been temporarily reduced after forest governance reforms in 2004. We find, however, that small-holder dominated settlement frontiers have remained largely unaffected by this trend, which explains the observed increase in the overall share of small-scale deforestation until 2010. Additionally, our study found that patch size distribution has largely returned to pre-2004 levels in all frontiers after 2010. This finding contrasts with recent studies who associated post-2004 deforestation trends with small-scale farming (Rosa, Souza, and Ewers 2012, Godar, Gardner, et al. 2015, Assunção, Gandour, Pessoa, et al. 2017).

Furthermore, it suggests that large-and medium-sized deforestation may have only been temporarily put on hold by actors expecting a return to laissez-faire forest governance, or to invest in less scrutinized Brazilian biomes, such as the Cerrado (Richards et al. 2016). This paper's main contribution lies in making use of increasingly rich and publicly available spatial and non-spatial data sources to reconcile a macro-scale perspective on land cover change, with spatial heterogeneity in local constellations of land use drivers. As such, our approach may also prove useful to inform conservation policy design at other tropical forest margins. A couple of theoretical and methodological caveats remain, nonetheless, that should be addressed in future research.

Our frontier classification is more reliable for older frontier types than for dynamic new frontiers. In future analyses, it would be desirable to concentrate on newer frontiers with an additional set of variables that discriminate better between local differences within emerging frontier areas. We experimented with spatial information on large hydroelectric dams and mining projects as frontier classifiers, but the spatial distribution of these variables reduced the stability of cluster solutions. Selected additional clustering variables could be used to relax strong assumptions made in the Von Thünen inspired frontier framework, such as the uniformity of agro-ecological conditions. Differences in agricultural suitability can explain the persistence of extensive land use forms in regions that are not suited for intensive agriculture, even if they are close to an urban market. The same applies to the expansion of input intensive activities in suitable, but remote frontier areas. Furthermore, foreign trade effects could be included by measuring distance to export infrastructure, such as processing facilities and ports, to account for global market forces that can affect frontier development. And finally, land rent theory may be insufficient to explain the persistence of pastures as dominant land use forms across frontier types. Beyond economic factors, cattle ranching fulfills social (Garrett, Gardner, et al. 2017) and cultural functions (Hoelle 2015) that yet need to be incorporated into modern frontier theory.

3.6 Conclusions

Reconciling environmental conservation and rural development at deforestation frontiers in the tropics requires a mix of conservation and rural development policies that appropriately address heterogeneous frontier development stages and coordinate policy action among historically separate governance domains. The publicly available frontier map developed in this study can support the design of such policy mixes, for example, via analyses of policy impacts and the identification of priority regions for government action (available at Schielein 2018). Our findings suggest that higher levels of deterrence could stabilize forest loss in the New Frontier, where deforestation is on the rise and native vegetation reserves are still sizable. Furthermore, policy-makers should be aware that the composition of forces that drive frontier development at New Frontiers changes over time. Agricultural intensification is promoted in the literature as a possible solution to limit land expansion (Cohn, Mosnier, et al. 2014; Strassburg et al. 2014), but only if associated with improved environmental governance (Ceddia et al. 2014). Otherwise, government-supported intensification programs such as the Brazilian Development Bank's Inovagro may stimulate rather than alleviate land demand in new frontier areas.

In capital and labor scarce settlement frontiers and High Pressure PAs land expansion is still strongly associated with cattle-dependent small-scale farming. The reasons for this phenomenon are well-studied and include, among socio-cultural aspects, a mistrust in bank savings, safety-net functions, such as liquidity in case of emergencies, low investment costs and low labor intensity, governmental support, and pasture creation as a mechanism to secure land claims (Aguiar Gomes 2009, Vadjunec, Gomes, and Ludewigs 2009, Dávalos et al. 2014, Garrett, Gardner, et al. 2017).

Nonetheless, relaxing constraints to agricultural development by improving transport networks and technical extension services could stimulate the substitution of land-consuming cattle activities by market-oriented annual or perennial agriculture with positive welfare impacts. Such measures will, however, only produce the desired land sparing effect if accompanied by effective environmental law enforcement to discourage extensive land uses that rely on illegal deforestation. Policy mixes along these lines could pay off most in settlements dominated by north-eastern and northern populations, where forest reserves are still abundant and deforestation rates are rising.

Deterring deforestation in protected area frontiers should remain among the top priorities for government action, for example, under the PPCDAm. Our frontier map identifies large areas of invasions in the High Pressure PA zone where stronger law enforcement is required. In contrast, the trend towards degazettement of protected areas adopted by the Brazilian government in recent years (Bernard, Penna, and Araújo 2014) might have already created additional incentives for protected area invasions in Pará and other regions within the biome (Pires 2017).

Where protected area status allows sustainable land and natural resource use, deforestation pressure could be reduced by strengthening local economic activities that depend on sustainable timber and non-timber forest product extraction. Existing policies such as the National Plan for the Promotion of Sociobiodiversity Value Chains (PNPSB) should specifically target such high risk areas, for example, via effective minimum prices or subsidized credit and insurance schemes.

International cooperation and NGOs increasingly promote specific value chain based governance measures (Lambin, Meyfroidt, et al. 2014). However, to the extent that our Von Thünen inspired framework explains forest conversion dynamics in the Brazilian Amazon, commodity-based value chain governance can only partially influence land use decisions at the frontier and remains vulnerable to cross-frontier leakage and spillover effects. Our results thus emphasize the important role of effective environmental law enforcement as a crucial backbone for conservation success across the diverse frontier landscapes in the region.

CHAPTER 4

Environmental Governance Drives Cattle Intensification but Bears the Risk of a Social Divide in Acre, Western Brazilian Amazon

An article with similar content including Appendix C has been submitted to Regional Environmental Change for publication and is currently undergoing a major revision as: Schielein, J., Börner, J. and J. Valentim: Environmental Governance Drives Cattle Intensification but Bears the Risk of a Social Divide in Acre, Western Brazilian Amazon.

4.1 Introduction

Reducing emissions from deforestation and land-degradation is a key priority to reduce global GHG emissions and combat climate change (IPCC 2014, UN-REDD Programme 2015). The Brazilian government pledged to end illegal deforestation in the Amazon and other biomes by 2030 to achieve national emission targets under the Paris Agreement of the UNEP (Brazil 2015). Brazil's over 111 million hectares of cultivated and often extensively used pastures (Cohn, Mosnier, et al. 2014) are the dominant land use following deforestation and the cattle sector contributes with 75-80% to the total land use related emissions in Brazil (Bustamante et al. 2012). At the same time, cattle ranching is one of the most important pillars of the rural economy in the Brazilian Amazon (Lapola, Martinelli, et al. 2014) and experienced steady growth over the last decades (Bowman et al. 2012). Extensive cattle production has historically evolved as an economically convenient frontier production system and it is associated with positive socio-cultural norms and values (Margulis 2004, Salisbury and Schmink 2007, Aguiar Gomes 2009, Vadjunec, Gomes, and Ludewigs 2009, Dávalos et al. 2014, Hoelle 2015, Garrett, Gardner, et al. 2017).

As cattle-ranchers become key protagonists in achieving Brazil's emission targets, modeling studies suggest that ending deforestation while increasing agricultural productivity is feasible in biophysical and economic terms by intensifying cattle production and reutilizing abandoned and degraded pastures (Cohn, Mosnier, et al. 2014, Strassburg et al. 2014, Mazzetto et al. 2015, Gil et al. 2018). However, little empirical research addresses the socio-economic preconditions of transforming cattle production systems in this way or the effectiveness of the required policy measures (Vosti, Witcover, Carpentier, et al. 2000, Strassburg et al. 2014).

Despite this lack of empirical evidence, substantial investments are made to promote pasture restoration and cattle-intensification as REDD+ mechanisms throughout Brazil. Examples include the Low Carbon Agriculture Plan (ABC) as well as NGO-led initiatives to establish sustainable cattle ranching in agricultural frontier zones (Instituto Centro de Vida 2018, The Nature Conservancy 2017, WWF Brasil 2017). However, Merry and Soares-Filho Merry and Soares-filho (2017) criticize these efforts as premature. Comparing frontier expansion in Brazil and in the US they argue that conservation is a precondition rather than an outcome of intensification. In other words, conservation-induced land scarcity is required to encourage investment in more intensive cattle production systems.

We test this conjecture by investigating the role of environmental law-enforcement in triggering behavioral change among ranchers towards restoring degraded pastures. Our case study is the state of Acre (western Brazilian Amazon), where we collected data from 121 cattle farms. We integrate our primary data set with external data sources to estimate how enforcement activities influenced pasture restoration and deforestation in formerly degraded areas between 2005 and 2014. The survey and data sources are documented in section 4.3.2 of this article and the regression model in section 4.3.1. The results are presented in section 4.4 and indicate that enforcement action exerts deterrence rather among cattle ranchers who witness enforcement activities than on their neighboring convicts. Furthermore we find that capital endowment is of paramount importance for restoration and that the socio-economic inequality in the region translates into a small group of large farms who possess most of the restored pasture area. Descriptive statistics and open questions from our survey furthermore indicate that restoration is associated with a lower degree of deforestation and increased environmental compliance. We discuss our results in section 4.5 where we also address prospects and preconditions of supporting policies such as technical assistance and credits to promote restoration efforts.

4.2 Conceptual Framework

4.2.1 Pasture Area Expansion and Cattle System Improvements

Our conceptual framework is motivated by the observation that farmers generally tend to increase their herd-sizes, which can be either achieved by expanding the productive pasture area or by improving the pasture and animal management within the existing production system. Pasture expansion by means of deforestation is relatively cheap because it has comparatively low labor and capital requirements and some, if not all of the costs can be covered by the sale of the more valuable trees from the to be deforested area (Bowman et al. 2012). At the same time, area expansion is often associated with lowly productive (extensive) pasture management systems, because larger tracks of land are more difficult and costly to manage than smaller parcels.

Technological alternatives to pasture expansion can be summarized under two broad categories: improvements in animal management and improvements in pasture productivity. Both involve the substitution of (newly deforested) land for labor and capital, because they enable farmers to use existing pasture areas more efficiently and thus potentially spare forest areas (Dias-Filho 2014, Latawiec et al. 2014, Strassburg et al. 2014, Valentim and Ferreira Valentim 2016). Improvement of pasture productivity often entails rotational pasture management to boost fodder uptake and suppress invasive species (Corsi et al. 2001, Andrade, Valentim, et al. 2005, Euclides et al. 2010, Andrade, Garcia, et al. 2012). Moreover, pasture productivity can be improved through the use of agrochemical inputs (Garcia et al. 2017, Ermgassen et al. 2018) or legumes (Shelton, Franzel, and Peters 2005, Valentim 2005, Valentim and Andrade 2005). Highly degraded pastures may require a more comprehensive treatment to regain productivity which is called pasture restoration. Pasture restoration consist of mechanical treatment as well as limestone and fertilizer application before improved cultivars of forage grasses and legumes are introduced (Dias-Filho 2015). In our study we focus on these comprehensive forms of management intervention as an alternative to deforestation. Management improvements, however, come along with higher labor and capital requirements than deforestation. Their cost-benefit ratio in comparison to deforestation largely depends on governance, farm-level and agribusiness production conditions, which is discussed in the two subsequent sections.

4.2.2 Environmental Governance and Input Substitution

Historically, deforestation was a cheap and politically supported strategy to accommodate growing cattle herds in the Brazilian Amazon region (Faminow 1998,Margulis 2004). During the last three decades, however, the Brazilian government adopted numerous policies to limit legal and illegal deforestation in the Amazon, which potentially reduce the economic viability of extensive cattle production (Nepstad et al. 2014, Valentim and Ferreira Valentim 2016). Besides the expansion of the protected area network (Soares-Filho, Moutinho, et al. 2010), legal deforestation is limited by the Brazilian Forest Code and requires prior authorization from state authorities. However, due to limited implementation capacities and bureaucratic hurdles, most deforestation happens without prior consent of the state authorities. Fines for illegal deforestation can be issued by the Brazilian Institute for the Environment (IBAMA) or state environmental agencies and range up to 5,000 BRL (or 2,184 USD as of November 2018) per hectare of deforested land. Farmers may appeal against fines, but irrespective of the outcome, transaction costs arise as legal processes can last for several years. Additional costs accrue when farms are fully or partially embargoed during legal processes, which affects access to federal credit programs and markets, for example, when slaughterhouses commit to deforestation free supply-chain agreements (Barreto et al. 2017a). Such governance measures and associated costs discourage deforestation and theoretically stimulate engagement in pasture restoration to increase productivity and maintain output. Following microeconomic theory we can frame this shift from pasture expansion to pasture use intensification as a substitution between the production factors land and capital:

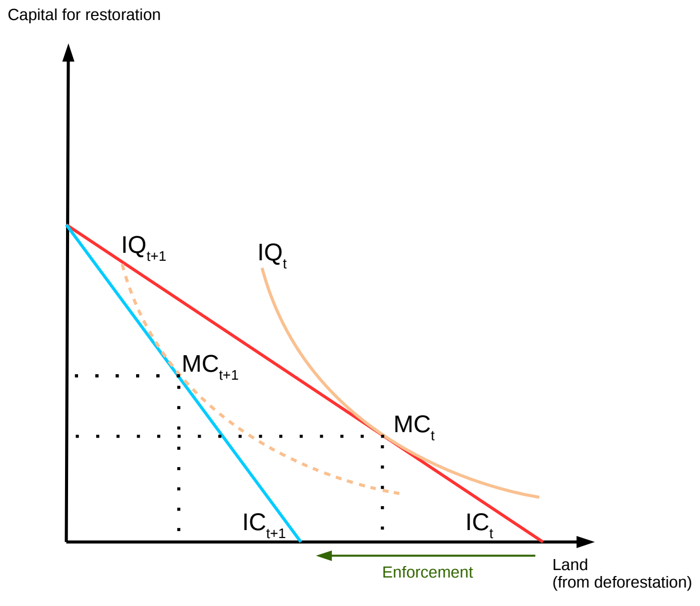


Figure 4.1: Input substitution between capital for pasture restoration and land from deforestation

Figure 4.1 illustrates the effect of forest law enforcement in a given time period t on the slope of the linear isocost line IC in the next period, i.e. pasture expansion by means of deforestation becomes more expensive as a strategy to maintain output levels. As a result, the new minimal cost combination MC_{t+1}, i.e. the point where the new isoquant IQ_{t+1} touches MC_{t+1}, is more capital intensive than the input mix at the point MC_t. Hence, enforcement affects both the level of output and the input mix, where we refer to the change in input mix as the substitution effect.

4.2.3 Farm-level Production Conditions and the Agribusiness Environment

The standard assumptions of microeconomic theory, such as perfect input and output markets, may not generally apply for cattle production at the agricultural frontiers of the Brazilian Amazon. Hence, we expect the size of the substitution effect to depend not only on technological features and conditions at farm-level, but also on the wider agribusiness environment. It thus helps to conceptualize pasture restoration as a two-step decision process, where farmers first decide whether or not to engage in restoration and then determine the scale of restoration activities.

Factors that influence both the initial decision and subsequent areas consist (besides the aforementioned enforcement variables) of input and output prices, agronomic suitability, capitalization, and past restoration as well as deforestation activity. Because of a general lack in infrastructure quality and very large distances that result in high transportation costs, farm accessibility is reported to be the most important factor to influence input and output prices at the farm-gate in the Amazon (Garrett, Lambin, and Naylor 2013). We expect that market access generally increases the likelihood to restore pastures over deforestation because of lower fixed transaction costs to hire and/or buy the required machinery and inputs. Heavy rainfalls and high temperatures are characteristic of our study region (Duarte 2006) and can result in acidic soils and low nutrient availability. Furthermore, steep slopes can reduce the local aptitude for machinery usage in some areas (Acre 2010). These factors collectively determine the agronomic suitability of an area for cattle ranching and we expect highly unsuitable areas to be omitted from restoration efforts. Regarding the role of capital we generally expect capital endowment to increase the likelihood to engage in pasture restoration and the corresponding amount of restored pastures. We expect, moreover, that farms with a high share of deforested areas are likely to engage in pasture restoration, because their expansion options are limited and their pastures tend to be older and less productive than on younger farms (Andrade, Valentim, et al. 2005).

In our framework factors that predominantly influence the initial choice to restore pastures consist of educational level, age, and credit access. We expect that a higher education level is positively associated with pasture restoration because farmers can more accurately assess the associated costs, benefits, and perceived behavioral control of their investments beforehand, which is reported to reduce individuals risk aversion to engage in longer term agricultural investments (Garcia et al. 2017). Studies have furthermore shown that experience in agriculture is of paramount importance for investment decisions and that experience is correlated with farmers' age, which we use as a proxy (Carpentier, Vosti, and Witcover 2000; Perz 2003; Caviglia-Harris 2004; Mercer 2004; Van Niekerk et al. 2011, Davis et al. 2012). Regarding credit access, past studies demonstrated that state-funded credit programs are an influential factor in the Amazon to overcome capital constraints for any type of farm investment (Garrett, Gardner, et al. 2017; Gil et al. 2018; Ermgassen et al. 2018). We therefore expect credit access to be positively associated with pasture restoration choice because initial investment costs to acquire or hire machinery for restoration are comparatively higher than the costs of deforestation.

Factors that predominantly affect the scale of restoration comprise past restoration efforts, the amount of years since a farmer started restoration, pasture improvements before restoration, pasture management techniques applied before restoration, access to regular technical assistance, and regular bookkeeping practice. Restoration efforts that predate our study period may affect the scale of restoration positively or negatively. On one hand, past restoration increases the amount of productive pastures, reducing the need for further restoration. On the other hand, learning from past restoration efforts may positively influence the scale of restoration efforts today. Furthermore, depending on the time elapsed since the first restoration effort, we expect that farmers who engaged early in restoration efforts tend to have a higher amount of restored areas than late adopters. On the contrary, if farms engage in rotational pasture management, they may depend less on restoration to maintain productivity than extensively operating farms (Corsi et al. 2001; Andrade, Valentim, et al. 2005). With respect to agricultural support policies, we expect that extension programs can help to overcome knowledge related adoption barriers of pasture restoration (Garrett, Gardner, et al. 2017). However, we argue that technical assistance is rather not a determinant of the restoration decision itself, because pasture restoration is a widely known production option that is promoted through various communication channels in our research area (Shelton, Franzel, and Peters 2005; Valentim 2005). Finally, regular bookkeeping helps farmers to track productivity changes for different farm management options. We expect regular bookkeeping to be positively associated with the amount of restored areas because it enables farmers to accurately assess the costs, benefits, and risks associated with alternative management options.

4.3 Empirical Strategy

4.3.1 Modeling Pasture Restoration

Our empirical analysis relied on descriptive statistics and a two-step regression model with Heckman correction as commonly used in studies about technology adoption in tropical agriculture and forestry (Mercer 2004). Descriptively we explored: (1) who restored pastures and when in time, (2) the association between pasture restoration and deforestation, and (3) how ranchers perceive the impact of environmental regulations and increased law-enforcement on ranching strategies. Specifically, we relied on Gini-indexes and a Lorenz-curve diagram to characterize the distribution of restored pastures amongst ranchers and statistical tests for the significance of correlations between stated and observed behavior. The latter included a proportional measure of enforcement risk perception that was constructed by asking respondents about the number of farmers that would be subject to fine for illegal deforestation if 10 farmers were to deforest in their intermediate neighborhood. We used this risk-perception index to perform a Wilcoxon-Signed Rank test to see whether there is a significant association between risk-perception and engagement in restoration between 2005 and 2014.

The two-step Heckman-model was motivated by the conceptual discussion in the previous section and applied here to study which factors influence restoration choice and the subsequent amount of restored areas. All calculations were made using STATA's survey functions to correct for our sampling design (see section 4.3.2).

The first stage Probit model estimated a farmers likelihood to choose pasture restoration as a production option and takes the form of:

$$Pr(restore = 1|X) = \Phi(\tilde{X}\hat{\beta}) \tag{4.1}$$

where restore indicates the choice to engage in pasture restoration (1 if choice is positive and 0 otherwise). $\tilde{X}$ is a matrix of explanatory variables including factors such as travel time to the next market and law-enforcement (See table 4.1 for an overview) and $\hat{\beta}$ is a vector of unknown parameters that are estimated using a cumulative distribution function Φ. From this, the inverse Mills Ratio $\lambda(\tilde{X}_i\hat{\beta})$ is calculated using $hat\beta$ on each observation i in the subsample of restorers. The inverse Mills Ratio was used in the second step of our model which is a Maximum Likelihood Estimation of the amount of restored areas, given a matrix X_i of explanatory variables and a vector β_i of unknown parameters.

$$area_i = X_i\beta + \lambda(\tilde{X}_i\tilde{\beta}) + \epsilon_i \tag{4.2}$$

where *area* is the amount of restored pastures in hectare, X_i is a matrix of covariates, some of which had been used in the Probit estimation as well as other covariates (see Table 4.1) and ϵ_i is an error term.

4.3.2 Database

We used a mixed-method approach to build a database that encompasses variables gathered from a farm survey based on a stratified random sampling design and external data sources summarized in Table 4.1 below. Primary data on farm and socio-economic characteristics was collected in a semi-structured household survey. The household questionnaire contained detailed questions about production characteristics in 2014 as well as recall questions on how production systems evolved in the previous ten years. We used a list of all cattle ranchers in Acre in 2015 as a sampling frame excluding a small subset of ranchers that were not accessible via terrestrial transportation. Section C.1 in the SM provides more details on the sampling strategy, survey design, and implementation. Details on other data sources can be found in the SM, section C.2.

Our main outcome variables were (1) a binary indicator whether a farmer engaged in pasture restoration during our observation period from 2005 to 2014 and (2) the subsequent amount of restored areas since the restoration decision was made. Since we knew in which year farmers started restoring pastures, we could construct before-after observations for time variant control variables from external sources, such as a count of fines for illegal deforestation in the close neighborhood (5km). We always used a fixed observation period of five years before the restoration efforts started to have a comparable timeframe. If farmers did not restore pastures during our observation period, control variables were defined with 2014 as the base year.

We used accessibility as a proxy for input and output price differentiation among farms. Accessibility was measured in minutes travel time from the closest farm boundary via terrestrial and riverine transportation networks to the next large city. We considered the two biggest cities in Acre: Rio Branco with 401,155 inhabitants

Table 4.1: Datasources utilized for the Heckman-estimation of pasture restoration in Acre

Variable	Scale	Source	Stage	Min	Max	Mean	SE
Pasture Restoration 2005-2014	dummy	Survey	1	0	1	45	8.4
Restored Pastures 2005-2014 (hectares)	continuous	Survey	2	0	2700	37.2	12.5
Travel time to next large city (minutes)	continuous	Schielein & Börner, 2018	1 & 2	19	489	179	25
Suitability	continuous	Embrapa (SM, section 2)	1 & 2	25	100	81	3
Farm size (hectares)	continuous	Survey	1 & 2	4	9000	234	45
Deforestation before (% total)	continuous	INPE, 2016	1 & 2	2	100	68	5
Literacy	dummy	Survey	1	0	1	0.78	0.08
Bookkeeping items	continuous	Survey	2	0	5	1.47	0.25
Age before restoration	continuous	Survey	1	24	86	57	2
Credit coverage (R$/hectare in district)	continuous	IBGE, 2009	1	0.5	127	10.5	1.7
Technical assistance coverage (% in district)	continuous	IBGE, 2009	2	0	86	8.7	1.5
Fine value in 1,000 R$	continuous	CENIMA/IBAMA, 2017	1	0	303750	1723	605
Count of fines in neighborhood before	continuous	CENIMA/IBAMA, 2017	1	0	15	2.6	0.7
Restoration before 2005 (hectares)	continuous	Survey	1 & 2	0	1440	15	9
Improved pasture areas before (hectares)	continuous	Survey	2	0	695	12.2	9.2
Years since first restoration	continuous	Survey	2	0	10	1.4	0.3
Rotational pasture mgmt. (% in district)	continuous	IBGE, 2009	2	0	100	67	5

and Cruzeiro do Sul with 87,673 inhabitants, because both act as the most relevant hubs for agricultural input and output markets.

Agronomic suitability estimates were based on a study by the Brazilian Agricultural Research Corporation (Embrapa), which categorizes different soil components according to their suitability for pasture systems in the Amazon. We converted the original categorical data set into a normalized continuous variable that takes values between 0-100, where zero indicates a very low and 100 the highest suitability for cattle systems. In the absence of reliable data on capital endowments prior to restoration, we took farm size before restoration as a surrogate.

To measure deforestation, we relied on the PRODES project of the Brazilian Institute for Space Research (INPE), which shows clear-cut deforested areas on an annual basis (INPE 2019). We create two variables from PRODES data: (1) the total share of deforested areas inside a farmers' stated area in the rural environmental land cadaster (CAR) up to the year of pasture restoration and (2) annual shares of deforestation after restoration started.

To characterize farmers' management capacity we worked with three different variables. First, educational background reportedly influences farmers' ability to take complex investment decisions (Huffman 2001). During our research we observed that literacy was a very important pre-condition to access up to date information about optimal farm management in the region and that illiterates make up 22 (±7)% of all ranchers in Acre. We therefore transformed our categorical variable that characterized the schooling degree into a dummy that accounts for the reported literacy. Second, we asked farmers if they were keeping record of important production numbers and to name the items that were written down on an annual basis as an indicator for regular bookkeeping. The third indicator of management capacity was a farmers' age before restoration started which we use as a proxy for farming experience.

Credit access was measured as total amount of disbursed credit in Brazilian Reais in 2006 per hectare of farm-area in a census district, whereas access to technical assistance was calculated as the share of farmers who received assistance in the same year. We thus assumed that the institutional context providing credit and technical assistance was stable throughout the study period, which was confirmed by the institutions who supported our field research.

To analyze the impact of law-enforcement on pasture restoration we linked the geo-referenced database from IBAMA to our farm areas identified in the CAR. Crossing both data-sets allowed us to identify farmers that had been fined for illegal deforestation including monetary fine values. Only fines issued prior to the restoration decision are considered. Similarly, a second variable measured the presence of state authorities by counting the number of fines for unauthorized deforestation in the close neighborhood to a given farm (5km radius).

4.4 Results

4.4.1 Descriptive Statistics

Pasture restoration gained importance in the second part of our research period (Table 4.2). Overall, 45% (±8) of cattle farms in Acre reported to have started mechanized restoration efforts between 2005 and 2014. A significant proportion (29% ±8) of those who did not restore during the research period was considering to start restoration in the next couple of years and only 7% (±4), mainly medium sized farms, had started mechanized restoration already before 2005. In contrast, 19% (±7) of cattle farmers did not consider restoration as a production option at all.

Table 4.2: Number of survey respondents who started pasture restoration in a given year

Year	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**None**	**Total**
Sample	5	0	3	6	3	8	5	15	11	7	58	121

Farm size clearly mattered in determining restoration choices: first, the share of small farms (<400 ha) adopting restoration within or before the research period is 55% (±9) and thus smaller than the share of medium farms (400-1,500 ha) with 85% (±9) and large farms (>1,500 ha) with 78% (±12). Second, restored pasture as a share of total pasture area is also lower on small farms (17% ±4) compared to medium (50% ±15) and large farms (28% ±7). For the whole state of Acre small farms who either did not start restoration yet or have very low shares of restored pasture areas dominate the picture (for a graphical overview see Figure C.6 in the SM, section C.5). And third, high inequality in total pasture possession and restored pasture possession reflects the unequal underlying distribution of land in Brazil. Figure 4.2 shows that around 52% of all pastures and 82% of restored pastures belong to only 10% of cattle ranchers in Acre. The corresponding Gini-Index for pasture area is 0.68 and for restored areas 0.89.

Cattle farms in Acre boast a relatively high share of deforested land in 2014 (68% ±5). Again, we observe notable differences between consolidated land shares on small farms (71% ±5), medium farms (39% ±10) and large farms (43% ±6). The Brazilian forest code currently requires keeping between 50-80% of the farm area in its natural state depending on farmsize, natural vegetation and regional land use planning. According to our data, consolidation rates on almost every farm exceeded

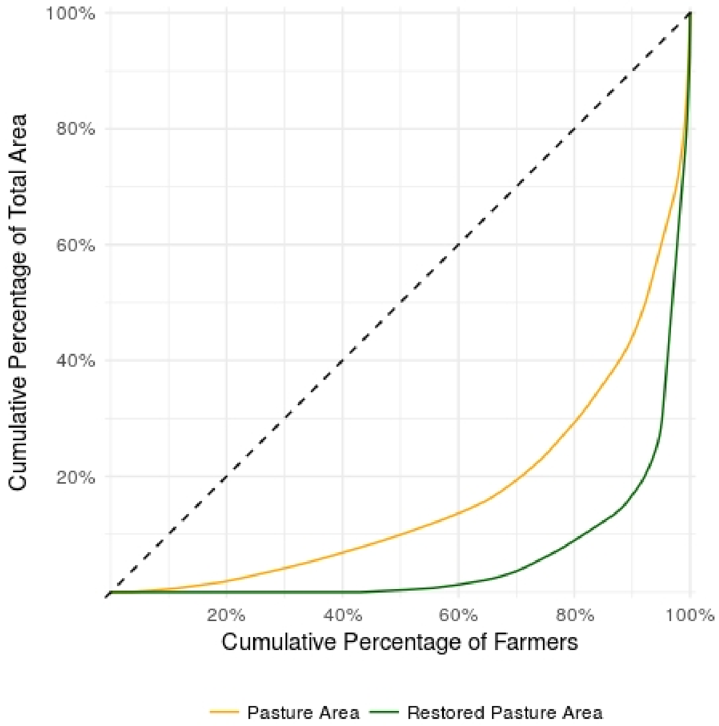


Figure 4.2: Lorentz-curve of total pasture possessing and restored pastures in Acre

the legal limits defined by the Brazilian Forest code (for a graphical overview see Figure C.5 SM, section C.5). During our observational period, farms deforested on average 8.4% (±2) of total farm area, again with notable difference between small farms (9.0% ±2), medium farms (2.7% ±1), and large farms (2.5% ±1). Comparing annual deforestation rates in the group of restorers and non-restorers we found that pasture restoration is associated with lower deforestation rates, except in three years (2008, 2010, 2014) where standard errors from our sample overlap strongly (Figure 4.3).

Altogether 47.4% (±8) of ranchers agreed that environmental law-enforcement and regulations influenced their production strategy, nearly always in an environmentally beneficial way. Of those who reported a behavioral change 87% confirmed that they reduced deforestation and tried to increase productivity in existing pastures in different ways. Statements like “If there was no regulation I would not have any single tree standing here anymore” were commonplace (statement made by a farmer in Manoel Urbano municipality, Acre, 2015). The most frequent change in terms of production strategy is the introduction of rotational pasture management to increase stocking rates. Second most frequent are pasture restoration and pasture improvement as well as animal feed supplementation with maize or soy to increase per hectare productivity. One negative impact to be reported very frequently amongst smaller farmers is the reduction or abandonment of subsistence food production within the traditional “roçado” system which requires small plots of deforestation on an annual base.

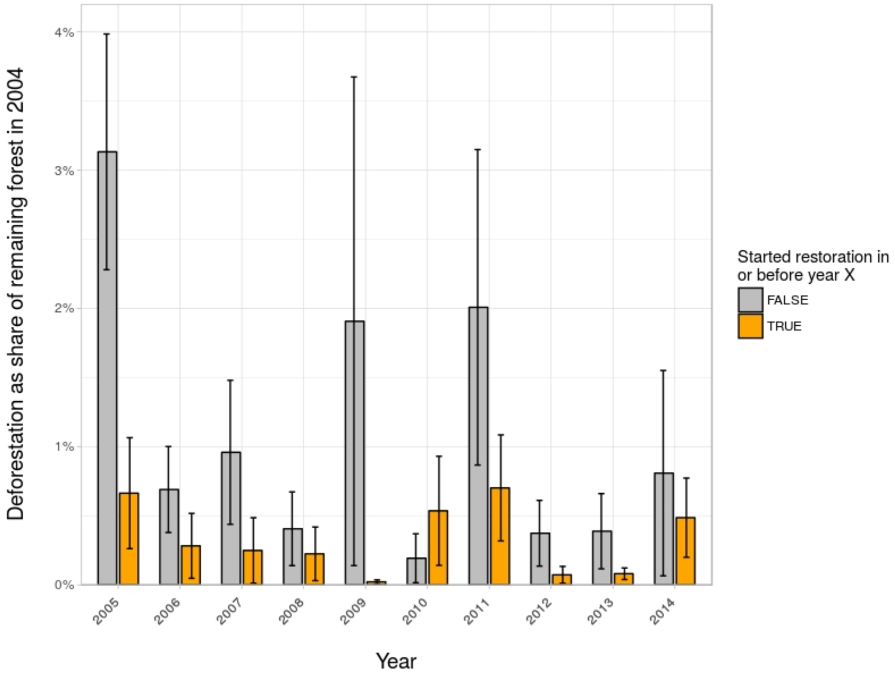


Figure 4.3: Group-wise means comparison of deforestation rates amongst restorers and non-restorers

Finally, results of a Wilcoxon signed-rank test, run to compare the perception of environmental law-enforcement risk amongst restorers and non-restorers, confirm the aforementioned observations. The test result were significant at the 0.1% level showing that increased risk perception was associated with the choice to restore pastures between 2005 and 2014. Figure 4.4 depicts the responses for the group of non-restorers (grey) and restorers (orange).

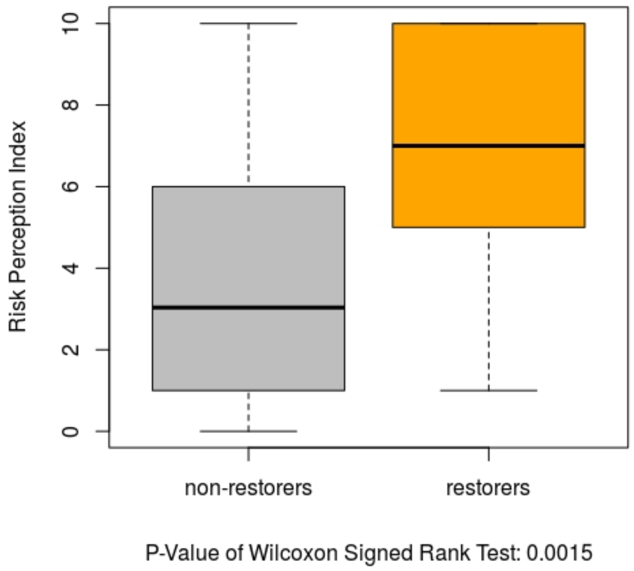


Figure 4.4: Risk perception index amongst restorers and non-restorers

4.4.2 Determinants of Pasture Restoration

The results of the first stage estimation of our Heckman model (fist column, Table C.1) show how policy factors and other covariates influence farmers' choices to restore pastures. As expected, the likelihood to start restoration efforts decreases with travel time to the next large city from 83% (±10) just at the edge of the city to 13% (±8) in a 5 hours travel distance keeping all other variables constant at their mean (for all marginal effects see SM, section 4). Farm area shows a positive effect with the likelihood to start restoration efforts increasing from 37% (±9) in the case of small farms (<400 ha) to 63% (±14) for large farmers with up to 1500 ha and 99% (±4) for very large farms (>5000 ha).

Restoration prior to our observation period is negatively associated with restoration choice. Furthermore, credit access exhibited a strong, positive effect on restoration. The likelihood to adopt restoration increases from 13% (±8) in the absence of credits to 99% (±1) if 50 R$/ha of credits were disbursed on average. A positive effect was also found for age where margins can be found at 12% (±6) for a 35 year old farmer and 52% (±6) for a 50 year old farmer. The same holds true for the effect of literacy with margins at 26% (±10) for illiterate farmers and 51% (±4) for literates. Law enforcement in terms of direct fines for illegal deforestation had an unexpected negative impact on restoration efforts, where a fine of 40,000 R$ decreases the likelihood to engage in restoration to only 17% (±8) vis-à-vis 41% (±8) in the absence of a fine. However, the indirect effect of law-enforcement was positive, as expected. It was measured as fines issued in the intermediate neighborhood of a farm (5km) before restoration. The coefficient is significant and positive indicating that the potential threat of environmental law-enforcement (as opposed to its materialization) increases the likelihood to start restoration. A total number of 10 fines in the neighborhood increases the likelihood to restore pastures to 84% (±11), whereas the likelihood in the absence of fines was only 24% (±8).

The results of the second stage estimation (second column in Table C.1) show the effects of law-enforcement and other covariates on the amount of restored pasture areas, once a restoration decision was positive.

In contrast to the first stage, the second stage estimation did not suggest any effect of travel time on the amount of restored pastures. In other words, once a restoration choice is made, market remoteness does not affect the scale of restoration. The coefficient of farm-area is positive. We observe that for each additional hectare farm-area the amount of restored pastures increases by 0.10 hectares, reflecting lower consolidation rates on large farms and the fact that many medium sized farms already restored considerable amounts of pastures before our observation period. Farm age is negatively associated with restoration area, which went counter our expectation, that old farms are more likely to invest in measures to enhance pasture quality. Neither do we find evidence that consolidated farms are more inclined to engage in restoration than farms with large forest reserves. Pasture improvements before restoration exhibit, nevertheless, the expected negative effect on restoration. The effect was small (-0.14) and thus indicates that improvements are selective and do not serve as full substitutes for restoring heavily degraded pastures. Restored areas prior to 2005 have a weakly significant, positive effect on restoration between 2005 and 2014, which is driven by a few observations with exceptionally large restoration areas. The total time elapsed since the last restoration effort exhibited a positive effect on restored areas ($\approx$ 17 hectares on average per year) indicating

Table 4.3: Regression table for the Heckman selection model for pasture restoration in Acre

First-stage estimation: Restoration (yes/no)		Second-stage estimation: Restored areas (ha)	
Travel time	-.0068749 *** (.0022504)	Travel time	.1031787 (.1894899)
Farm area	.0005776 † (.0003664)	Farm area	.1000571 *** (.0223537)
Prior Restoration	-.0042221* (.0023509)	Prior Restoration	.412147 (.2819016)
Fines in Neighborhood	.1724002 *** (.0574442)	Fines in Neighborhood	2.104721 (2.314311)
Fine Value	-.0185748 ** (.007656)	Fine Value	-.9323064 *** (.3252971)
Suitability	.0132325 (.013544)	Farm age	-.974473 * (.5038715)
Credit coverage	.0806763 *** (.0302238)	Deforestation	19.32534 (44.49323)
Age	.0441566 *** (.0153675)	Improved areas	-.135442 ** (.0629888)
Literacy = 1	1.317492 * (.6737515)	First year of restoration	17.48348 ** (7.295803)
		Bookkeeping	17.3629 ** (7.264521)
		Technical Assistance Coverage	.1459208 (.8296802)
		Rotational Management	.1266165 (.2705441)
Model Parameters			
/athrho	-.1420836* (.0740229)	rho	-.1411351 (.0725484)
/ln sigma	4.324834 *** (.2786509)	lambda	-10.66317 (7.043819)

Note: † p<0.15 * p<0.05 ** p< 0.01 *** p< 0.001

that not all areas are restored at once but restoration is rather implemented gradually. Our count of bookkeeping items is, as expected, positive. For each additional bookkeeping item, roughly 17 hectares are restored on average, suggesting a considerable impact of farm management capacity on restoration. Nonetheless, technical assistance to improve farm management does not significantly affect the restoration area. The direct effect of law-enforcement on restored area is consistent across modeling stages. Direct fines for unauthorized deforestation tend to decrease the amount of restored areas, but the indirect effect of law-enforcement on restoration area is not significant anymore.

4.5 Discussion and Policy Implications

Using descriptive and formal empirical analysis of a spatially consistent set of primary and secondary data sources we provide evidence for an environmental policy induced change of production strategies among cattle farmers in the Brazilian state of Acre. Our findings confirm and add nuance to recent studies focusing on other parts of the Brazilian Amazon (Koch et al. 2019, Garrett, Koh, et al. 2018).

Our first contribution lies in shedding light on the socio-economic determinants of pasture restoration. We show that restored pastures exhibit an extremely unequal distribution among cattle producing farms in Acre. The regression analysis in Table C.1 suggests that concentrated capital endowment and land ownership complement each other in driving this outcome. As a result, small farms seem to be less able to substitute between capital intensive and land intensive pasture management approaches than medium and large scale farms. This could explain results by Gibbs, Munger, et al. (2015) who found conservation gains on medium and large farms that specialize in fattening operations to be counterbalanced by increasing forest loss on small farms that specialize in breeding. We further confirm that market access and management ability co-determine restoration choice and area.

Second, and in line with our hypothesis about factor substitutability (Figure 4.1), we observe a strong negative correlation between deforestation and pasture restoration. Given the small sample size and descriptive nature of this part of our analysis this finding deserves further scrutiny. Others have argued, for example, that intensification at agricultural frontiers is unlikely to bring about lasting environmental benefits as farmers may reinvest additional income from enhanced productivity in expanding operations (Vosti, Witcover, and Carpentier 2002; Cataneo 2002; Fearnside 2002; Valentim 2005).

The third contribution of this paper is to provide micro-level evidence supporting the view that environmental law enforcement is not only critical to secure the potential conservation (i.e. land saving) effects of intensification – it also acts as a driving force (Ceddia et al. 2014; Garrett, Koh, et al. 2018). To this end, our analysis suggests varying effects of law enforcement depending on whether farms are directly fined or merely witness of enforcement actions in their neighborhood (indirect effect). As for the direct effect, we found that farms are less likely restore and restore smaller areas the higher the size of a fine. Witnessing enforcement actions in the neighborhood on the other hand has a positive (indirect) effect on the adoption of restoration, but does not significantly affect the total amount of restored areas. Both findings are plausible if we consider that fines are associated with considerable costs, including the transaction costs of legal processes and decreasing operational returns due to embargoes that restrict market and credit access. Direct enforcement thus diminishes farmers' ability to invest in restoration, whereas the mere exposure to enforcement activity increases risk perception vis-à-vis business-as-usual, which makes restoration a more attractive investment alternative.

As the probability of being fined tends to increase with enforcement activity, the direct and indirect effects may somewhat neutralize each other. Hence, our study only provides limited support to the hypothesis that intensification of cattle production systems is predominantly driven by reducing deforestation opportunities (Merry and Soares-filho 2017). Net conservation effects can be positive though as suggested by Garrett, Koh, et al. (2018) who find that environmental regulations boosted stocking rates and other crop related intensification measures in Mato Grosso state and Schielein and Börner (2018) who find increasing stocking rates and a higher share of clean pastures of degraded pastures in the whole Brazilian Amazon region.

Our analysis of stated responses corroborates this as about half of all interviewed farmers confirmed to have changed their production strategy in response to stricter environmental regulations and law enforcement. The most frequently reported changes include the introduction of rotational pasture management to increase stocking rates (see also Koch et al. 2019) and the adoption of pasture restoration to improve productivity on degraded land.

Further micro-level studies on the relationship environmental governance and land use change are needed to quantify potential leakage effects and negative social consequences that risk undermining societal and political support for conservation policies in the long run. However, social safeguards to increase sustainable production on small farms, such as promoting basic farm management and literacy skills as well as environmentally conditioned credit programs for pasture restoration seem to be sensible agricultural policy measures already today. One key message of this study should find broad support among conservationists and the majority of agribusiness actors throughout Brazil who operate under more rigorous legal scrutiny than farms at Amazonian agricultural frontiers: clear rules and their effective enforcement across all production environments are the basis for fair competition and sustainable investment in the agricultural sector. One can only hope that future regulators will level the playing field by strengthening small-scale producers and law enforcement capacities as opposed to weakening established environmental legislation.

CHAPTER 5

Final Discussion and Concluding Remarks

Reducing deforestation and forest degradation as well as promoting sustainable agricultural production systems in the tropics are some of the most important ways to address global climate change and accelerated biodiversity loss. This study provides valuable knowledge for policy makers and researchers at the intersection of conservation and agricultural policy design, by contributing to general knowledge regarding the effects of infrastructure and conservation policies on LUCC at different scales of analysis in the Amazon region.

The first research goal of this study was to advance our knowledge on the role of accessibility for LUCC dynamics in the Brazilian Amazon. Our empirical analysis of accessibility and LUCC (Chapter 2) generally confirms that accessibility plays a central role in agricultural production and consequently land cover change dynamics that alter the configuration of the landscape at local scale. In addition to the already well documented positive effect that infrastructure has on deforestation rates, the results show that annual crops and perennial pasture systems both depend on access to different markets, as well as to urban areas at different periods of time throughout the year. One important finding is that cattle ranching is more flexible in regard to broken roads and reduced market accessibility during the wet season and that cropland farmers are more dependent on year round accessibility to urban areas. Therefore, spatially explicit information about accessibility in the dry and wet season (see chapter 2, Figure 2.5) can be relevant to policy makers who want to diversify local production portfolios and incentivize farmers to switch from extensive, land-consuming cattle systems to potentially less land-consuming and more lucrative crop production alternatives.

Chapter 2 also demonstrates the importance of paying close attention to the conceptualization and measurement of accessibility, as it is particularly relevant to future LUCC studies. To assist future researchers in achieving this goal, the *AccessibilityMaps* package for the R Statistical Programming Language was developed during this dissertation, with the hope that it can help other researchers to quickly and flexibly create different accessibility measures for their own research areas. This package is available using free and open source software technology.

The field study in Acre (Chapter 4) found evidence that accessibility plays an important role in pasture system intensification at the micro-scale. The results suggest that the likelihood to start restoration efforts decreases as travel time to the next large city increases, ranging from from 83% (±10) when in close proximity to the city to just 13% (±8) at 5 hours of travel distance. Increased transportation costs for inputs and outputs might be a possible explanation for this effect, as well as decreased government presence in terms of assistance programs to promote restoration and other types of animal and pasture related management improvements.

Chapter 3 complements the field observations from Acre regarding the role of infrastructure for land use choice by identifying potential priority areas for govermental-led infrastructure investments to promote cattle system diversification and intensification in the Brazilian Amazon. For example, settlement frontiers, predominantly dominated by farmers originating from north and north-eastern Brazil, are poorly accessible in the wet season due to missing all-weather roads, they exhibit high deforestation rates, but also have considerable remaining forest resources. Furthermore, those areas have a strong predominance of cattle ranching and considerably less production alternatives. In addition they have a significantly lower degree of agricultural intensification and input use as, for example, old deforestation frontiers which underwent a longer time of agricultural development. Supporting policies such as infrastructure investment and agricultural extension services should therefore prioritize settlement frontiers with an integrated sustainable development approach, including investments in the road infrastructure, human capacities, and conservation governance. This type of integrated approach could also prove to be beneficial to increase the level of trust that farmers have with environmental institutions, as they are often perceived as antagonists of the production sector that are seen as mainly responsible for issuing penalties and punishments to farmers. In this context, the results from Acre indicate that agricultural extension services for small-scale farmers may benefit from using a concept of (technical) capacity building that includes alphabetization programs and basic farm management skills.

Nevertheless, agricultural diversification and intensification of active deforestation frontiers, as well as environmental compliance depend crucially on enforcing existing laws and regulations and inducing “artificial” land-scarcity via effective usage quotas, protected areas and the enforcement of existing regulations. Otherwise it is not likely to deter people from breaking the forest laws to increase their economic benefits. The analysis of pasture restoration in Acre showed furthermore that active law enforcement of the forest code does not only deter deforestation, but it also stimulates investment decisions into more productive ranching systems amongst farmers who witness the penalization of *broken (forest) laws* in their neighborhood. Witnessing penalization of other farmers, measured as issued fines for illegal deforestation in the intermediate vicinity of a farm, increased the likelihood of pasture restoration up to 84% (±11) if 10 fines were issued prior to restoration efforts. The likelihood for farmers to engage in pasture restoration in the absence of fines in the immediate vicinity was only 24% (±8). As such, monitoring and enforcement systems are indispensable parts of an effective conservation policy scheme. In this regard, these results also contribute to the to the third research goal of this thesis, which is to analyze how conservation policies are associated to land cover change dynamics in different agro-economic realities at local scale.

The main goal of Chapter 3 was to create a holistic picture of deforestation frontiers and associated LUCC dynamics. It identified several zones of active protected area encroachment which should be the focus for stronger environmental law enforcement and increased support for sustainable income alternatives (e.g. through payments for ecosystem services in traditional communities). Other frontier types were also described in rich detail and the generated frontier map (Figure 3.5) can assist policy makers in targeting areas with frontier specific policy approaches.

Besides statistics on land use shares and agricultural characteristics, Chapter 3 also contains information about deforestation patterns in different frontier types over time. It depicts an increasing share of smallholder deforestation between 2004 and 2012, and thereby confirms the results of other studies (Rosa, Souza, and Ewers 2012; Godar, Gardner, et al. 2015) which suggested that the first two phases of PPCDAm were mainly effective in targeting deforestation by large landholders in the research area. In addition, this study contextualizes these results by describing the agricultural production characteristics, infrastructure conditions and land use systems inside specific frontier regions were smallholder deforestation mainly occured. Furthermore results from Chapter 3 suggest that the observed trend of increasing smallholder deforstation was reversed after 2012 when large-scale deforestation gained significance again. This reversal could be the result of a delay in investments by large landholders during PPCDAm phases one and two, but that are now back to business-as-usual with increased political support for deforestation from the current government.

The persistence of environmental destruction and extensive cattle ranching despite increased conservation governance can be explained by the difficulties that smallholders face when it comes to transforming their production systems, such as increasing productivity on consolidated land. Chapter 4 provides empirical evidence that pasture restoration mainly took place on large farms with increased capital endowment and amongst a smaller group of well-connected smallholders in Acre. Results suggest that 82% of restored pastures belong to only 10% of cattle ranchers in Acre. Half of the smallholders in the research area did not engage in pasture restoration despite most of their farmland already being consolidated, often with marginally productive pasture systems. Future research should focus on this group of farmers in more detail and create knowledge to help prevent this environmental problem from becoming a question of social inequality where only rich farmers comply with environmental laws whereas smallholders are unprepared to meet the legislative demands.

The observed differences between productivity changes on smaller and larger farms is also relevant for the interpretation of the analysis results from the frontier study, which shows increasing cattle stocking rates in all deforestation frontiers. Those results raised the question of whether environmental governance leads to increasing cattle-productivity in all frontiers, or whether it may be an indicator of overgrazing in smallholder dominated settlements with potentially medium to long term negative impacts on productivity and the environment. From the observations made in Acre, the latter seems to be likely, indicating a need for caution when using stocking rates as an indicator of cattle intensification or linking these rates to successful outcomes of conservation governance as seen in Garrett, Koh, et al. (2018). Increasing stocking rates may indicate an improved carrying capacity, which can be the result of investments and pasture management improvements, or it may simply indicate that ranchers have increased the number of animals on existing pastures without any changes to their production system, potentially leading to faster degradation of pastures and increased land-demand in the medium run.

Pasture system intensification is discussed in the literature as potentially reducing deforestation pressure on forests by substituting new pasture creation (by means of deforestation) with restoration and productivity increases on already consolidated pasture areas (Cohn, Mosnier, et al. 2014, Strassburg et al. 2014). We found empirical support for this argument in Acre, where ranchers used pasture restoration to reduce their dependency on deforestation and comply with the existing environmental laws, which were enforced more rigorously during the first and second phase of PPCDAm (2004-2012). As a result, a short-term decrease in deforestation rates could be measured amongst restorers using satellite-based deforestation data on the plot level. Additionally, during the interviews about half of the ranchers referenced changes to their production strategy in response to conservation regulation and increased levels of law-enforcement.

Long-term conservation effects of cattle-system diversification and intensification remain, however, unclear. Although the field study in Acre was able to associate pasture restoration with short-term deforestation decreases at rates comparable to other studies (Angelsen 2010; Koch et al. 2019), it does not provide enough evidence to draw further conclusions regarding long term conservation outcomes. To achieve this, additional longitudinal studies are required that take a holistic picture of land use and land cover change into account. Furthermore, the results from chapter 3 showed that the most dynamic and new deforestation frontier in the study region is experiencing considerable amounts of cropland expansion in addition to pastureland expansion, reflecting a paradigm shift in recent frontier development in the Brazilian Amazon. Ultimately, intensified agricultural production is also associated with biodiversity loss and soil degradation in several tropical countries (IPBES 2019), and should not be considered as a silver bullet for conservation governance in the Amazon. Instead, a locally adapted mix of several policy instruments and closer cross-sectoral cooperation between environmental and agricultural institutions is most likely to produce desirable environmental outcomes and sustainable economic development in the long run.

APPENDIX A

Supplementary Material of chapter 2: Accessibility and LULCC

All travel time maps from this analysis are published as: Schielein, J., Frey, G., Miranda, J., Souza, R., Börner, J., and J. Henderson (2020). Accessibility Maps in the Brazilian Amazon 2004, Harvard Dataverse. https://doi.org/10.7910/DVN/DLYNAM

A geographical dataset with potentially navigable rivers (see Section A.4) is published as: Schielein, J. (2017). Potentially navigable rivers in South America. Harvard Dataverse. https://doi.org/10.7910/DVN/1G8PZI

A.1 Slope Correction for Accessibility Maps

Slope correction is based on Van Wagtendonk and Benedict (1980) for hiking travel times, which was later adapted by Nelson (2008) and Weiss et al. (2018) in a global accessibility to urban areas analysis. As in previous studies, the effect of slope in our model has an exponential, rather than linear, relationship with crossing the friction surface. Travel speed accounting for slope variations (S) is equal to the speed over no-slope areas (S_0) and a friction factor (f), based on the original model, then S is represented as:

$$S = S_0 \cdot e^{-f \cdot g} \tag{A.1}$$

In which g is the slope in a certain location. The calibration of the slope effect is then given by the element f in equation A.1. In our model we set this value at 3.

A.2 Secondary Roads Data and Assumptions on Travel Speed

Figure A.1 shows the two utilized data-sets for a highly dynamic deforestation frontier with intense secondary road construction including the BR-163 highway in Southern Pará (around Novo Progresso) as well as the São Felix do Xingu region in Para and the Alta Floresta region in Northern Mato Grosso.

Figure A.2 shows inspection logs from official IBAMA vehicles used to calculate paved and unpaved roads. We used the mean recorded speed for our assumptions.

Table A.2 gives a more detailed overview of the conversion of land cover classes to travel speeds for the creation of the friction map. All input classes are from the MapBiomas Project (2017).

Table A.1: Conversionmatrix vor landcover classes to travelspeeds

Mapbiomas landcover category	id	Travel –speed dry season in km/h	Travel –speed wet season in km/h
1. Forest	1	3	3
1.1. Natural Forest	2	3	3
1.1.1. Forest Formation	3	3	3
1.1.2. Savanna Formation	4	3	3
1.1.3. Mangrove	5	1	1
1.2. Forest Plantation	9	3	3
2. Non Forest Natural Formation	10	7	5
2.1. Wetland	11	1	1
2.2. Grassland Formation	12	15	5
2.3. Other non forest natural formation	13	7	5
3. Farming	14	15	5
3.1. Pasture	15	15	5
3.2. Agriculture	18	15	5
3.2.1. Annual and Perennial Crop	19	15	5
3.2.2. Semi-Perennial Crop	20	15	5
3.3. Mosaic of Agriculture and Pasture	21	15	5
4. Non vegetated area	22	15	5
4.1. Beach and Dune	23	15	5
4.2. Urban Infrastructure	24	30	30
4.5. Other non-vegetated area	25	7	5
5. Water	26	14	14
4.3. Rocky outcrop	29	3	3
4.4. Mining	30	1	1
5.2. Aquaculture	31	14	14
2.3. Salt flat	32	15	5
5.1. River, Lake, and Ocean	33	14	14

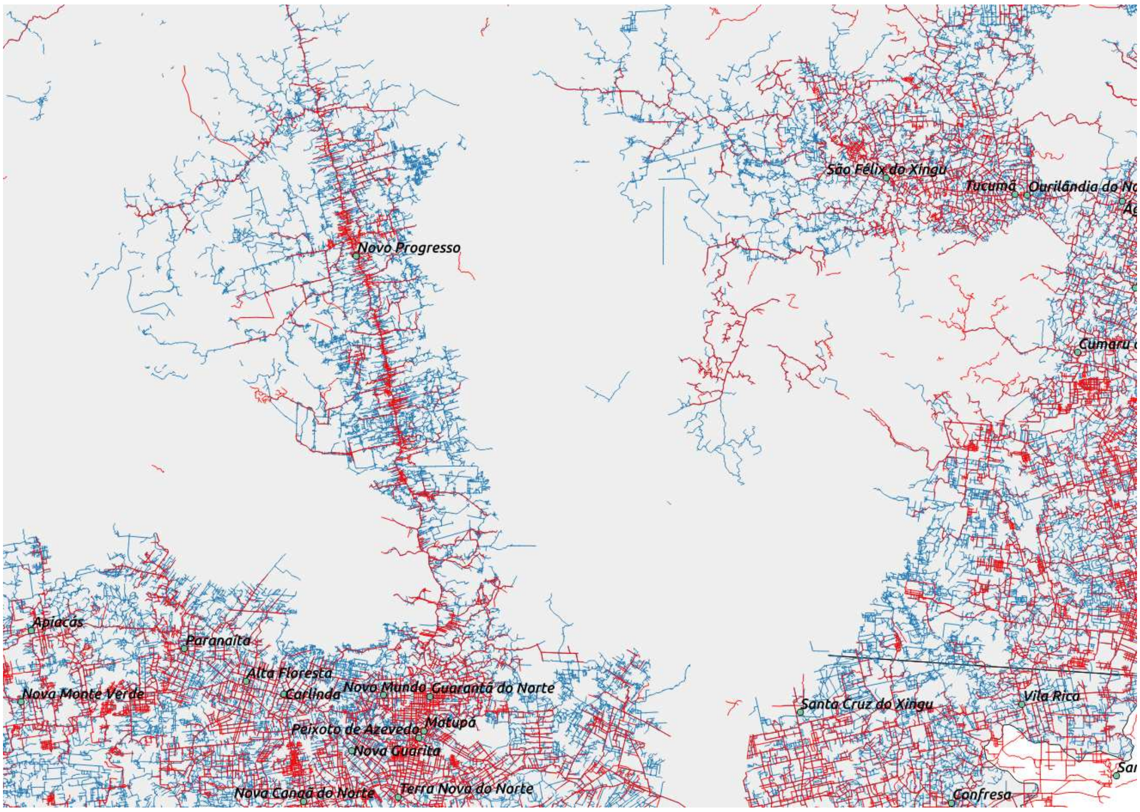
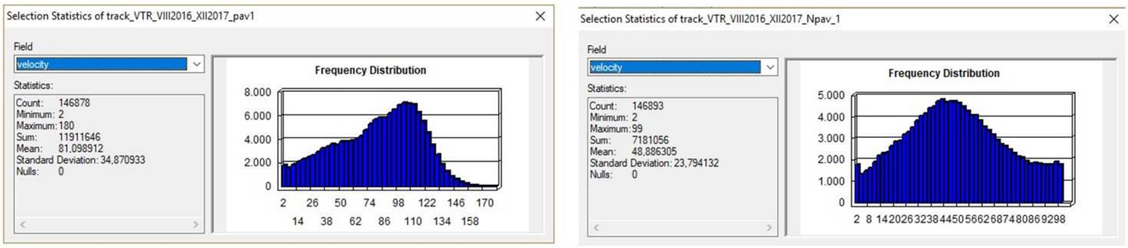


Figure A.1: Comparison of secondary road data for southern Pará/north-eastern Mato Grosso region

Note: Red colors show data for 2004 from Soares-Filho, Moutinho, et al. (2010) and blue colors show data for 2012 from Barreto et al. (2017b)



```
Selection Statistics of track_VTR_VIII2016_XII2017_pav1
Field: velocity
Count:     146878
Minimum: 2
Maximum: 180
Sum:       11911646
Mean:      81,098912
Standard Deviation: 34,870933
Nulls:     0
```

```
Selection Statistics of track_VTR_VIII2016_XII2017_Npav_1
Field: velocity
Count:     146893
Minimum: 2
Maximum: 99
Sum:       7181056
Mean:      48,886305
Standard Deviation: 23,794132
Nulls:     0
```

Figure A.2: Travel speeds from logs for accessibility model assumptions

Note: Left: paved roads, right: unpaved roads

A.3 Silos and Slaughterhouses Data

Silo data was obtained on request from CONAB (Companhia Nacional de Abastecimento) which is a public entity linked to the ministry of agriculture. CONAB maintains a national cadaster of all silos (private and public) called SICARM, which contains information on the storage capacity, the year of construction, and the geographical location of each facility. The original data set contains 16,784 observations, 2,951 of which are located within the nine federal states that comprise the Brazilian Amazon region. In this database, 95% of the observations have a corresponding construction year.

Data for animal processing facilities was obtained from the public website of the ministry of agriculture MAPA. The ministry maintains a system called SIGSIF where all establishments that process animals or animal derivatives are registered. The data from SIGSIF was downloaded automatically using a parsing algorithm. Subsequently, a subset was created for facilities that focus on cattle related activities. This subset was created using a pattern-matching algorithm with the following patterns ("BOVINO—bovino—Bovino"). The raw data contains the current status of the establishment (active/inactive), the year when the facility attained de SIF (port: "Data de reserve), as well as the year when the registration was completed (port: "Data do registro"). Unfortunately, the SIGSIF website only provides the postal code (CEP), municipality, or neighborhood of each establishment instead of specific geographical coordinates for the data.

Since the data does not contain geographical coordinates, several freely available data-sources were used to geocode the CEP information. The project CEP aberto (http://www.cepaberto.com/), for example, provides a public API to retrieve coordinates for all publicly known CEPs (1 Mio.). All told, we were able to retrieve 2,007 of 4,603 CEPs from the raw data. Our impression was that the cause for the missing 2,596 cases was most probably due to incomplete or outdated CEP information in the SIGSIF system. For all observations where we were not able to geocode the CEPs, we cross-checked the data with the CPF database administered by IBAMA, where all polluting industries must be registered. This database also contains geographical coordinates of many of the slaughterhouses. If the data was not found there, we used an additional data sources such as shapefiles from Imazon and the following associated publication (Barreto et al. 2017b). If a specific slaughterhouse could still not be identified, we used the geographical coordinates of urban areas from each municipality from IBGE, as all municipalities are listed in the original SIGSIF data set. This means that we assumed that the facilities which we were not able to geocode are located in the urban area of the listed municipality.

A.4 Identification of Potentially Navigable Rivers in South America

Summary

The river input data contains potentially navigable rivers for small and medium-sized boats in South-America depending on the topography, rainfall, and potential evapotranspiration of the water bodies. Hence, it is an approximation of the location of navigable rivers, not an actual map of waterways. Navigability is defined by the extent of a river, which in this case (1) for small boats accounts to 5-15 meters minimum extent and (2) for medium-sized boats 30-40m meters minimum extent. The model data was parameterized and validated with land-cover data from high-resolution satellite images.

Workflow and processing steps

Step 1: Creation of a corrected accumulated flow map

The base-data for this map is an accumulated flow map from the HydroSHEDS project (Lehner, Verdin, and Jarvis 2006) in the same resolution an extent as the final data-set. The HydroSHEDS map exhibits values for flow accumulation based on a drainage model derived from topographic data. The flow accumulation is expressed as the sum of cells that are affluent to any given grid-cell. Hence, downstream cells of a river show higher values, whereas upstream cells show lower values.

To correct this map with climatic information, a global map for soil-water balance from CGIAR was used (Trabucco and Zomer 2010). This map shows average yearly soil water content (1950-2000), which is a function of precipitation and potential evaportranspiration. Both raster maps where brought to the lowest possible resolution and minimum spatial extent (both based on the HydroSHEDS data). Then both rasters where multiplied, resulting in a corrected accumulated flow map, where moist regions show higher values for (corrected) accumulated flows than dry regions.

Step 2: Sample River locations from remote-sensing data

Data containing the location of water-bodies based on Landsat (30x30m resolution) and Rapid-Eye (5x5m resolution) was obtained. The Landsat-based data was extracted from the PRODES project (INPE 2019) and covers waterbodies in the Brazilian Amazon region. The Rapid-Eye based data was obtained on request from the SEMA- Secretariat of the Environment in Acre/Brazil. The latter is limited to the state-boundaries of Acre in the Western Brazilian Amazon, which is advantageous because most listed water-bodies in the region are natural rivers (no dams, no artificial lakes and canals), making this data source a good match to the theoretical data from the accumulated flow model.

Small water-fragments where removed from both data-sets, which are often either water holes for animal production, or small natural lakes and parts of meandering rivers that were cut off at some point in time. Both data-sets were rasterized on a 5km scale, which creates a buffer around all rivers from the vector data. Then, a random-sample from each rasterized data-set was taken with 650 river cell values. This sample comes as a point vector layer which then again buffered with a 5km radius. The subsequent buffer is used to account for the spatial miss-match between the sample locations and data from the hydrological model with lower spatial accuracy. The buffer is intended to help cover raster-cells from the model at any given sampling point.

Step 3: Extract flow values from sample-data and define a threshold to create a model of navigability.

The buffered layers from (2) were used to extract values from the corrected accumulated flow map for all sampling points. Those values cover theoretical rivers in the flow map, as well as non-river areas. To get the flow-data from the rivers and exclude non-river values, the maximum of the cell-values within each buffered sampling point were extracted (which is always a river). The result is a vector of 650 randomly selected accumulated (and corrected) flow-values for both cases. These vectors are the theoretical representation of (corrected) flow-accumulation in rivers that are detectable on Rapid-Eye data in Acre and Landsat data in the Brazilian Amazon. **The low values within this distribution represent the thresholds where rivers with an extent of 5x5m/30x30m can still be detected from the satellite images**. However, since many parts of the upstream river areas are covered by dense rainforest with forest canopies that span over water-bodies, the actual extent of those rivers might be most close to about 10-15 meters for Rapid-Eye data and 30-40 meters for Landsat-data.

Step 4: Choose adequate thresholds from the lower boundary to define which rivers are still navigable.

The last step consisted of creating new raster maps of navigable rivers based on different thresholds from the flow distribution vectors. Binary raster maps where created based on an ifelse condition that reclassifies values above that threshold to 1, and below to 0/NA. Four values where compared to serve as a threshold: The median, the first quantile, the first decile and the first ventile. These maps where then compared to high resolution satellite images from Google earth (in most cases TerraMetrics data in 15m resolution) to see whether a river still exists where they were theoretically mapped. It was found that the first ventile is the most adequate representation for both layers, meaning that only the lowest 5% of values from the sampled points are excluded from the analysis. Furthermore, the river models were compared to deforestation data from the PRODES project (INPE 2019) and riverine settlements in Acre, where river-transportation determines how far into the forest settlers can migrate. All results indicate a good fit of the model.

Notes:

The resolution of the presented data-set is very large, especially for small rivers, which subsequently influences the spatial accuracy of the data. It is a course approximation of where potentially navigable rivers are to be found rather than a high resolution map. Please compare this data to actual satellite imagery to understand the impact of this effect. Therefore, the present data-set can neither be used for fine-resolution mapping exercises nor can it precisely estimate the extent or length of a given river accurately. To accurately map the extent and actual course of (often meandering) rivers, it is advisable to use high-resolution satellite images in combination with this data.

Furthermore, the reliability of river navigability is higher in flat topographies than in mountainous regions due to the presence of waterfalls, fast currents, etc. Mapping of potentially navigable rivers could be improved by accounting for several additional factors that influence navigability such as deep topographic cuts that lead to waterfalls, sandbanks, climatic variability and others.

Visual comparison of the model results with different thresholds

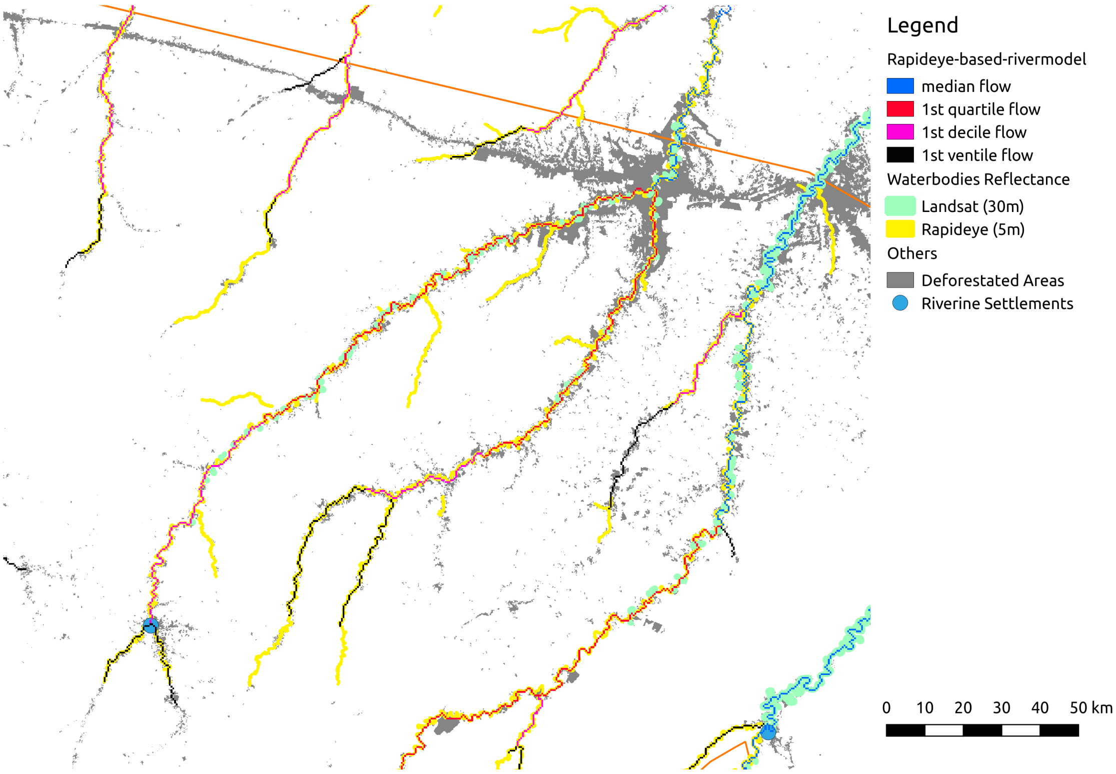


Figure A.3: Comparison of modeled data to water bodies from remote sensing products and deforestation data

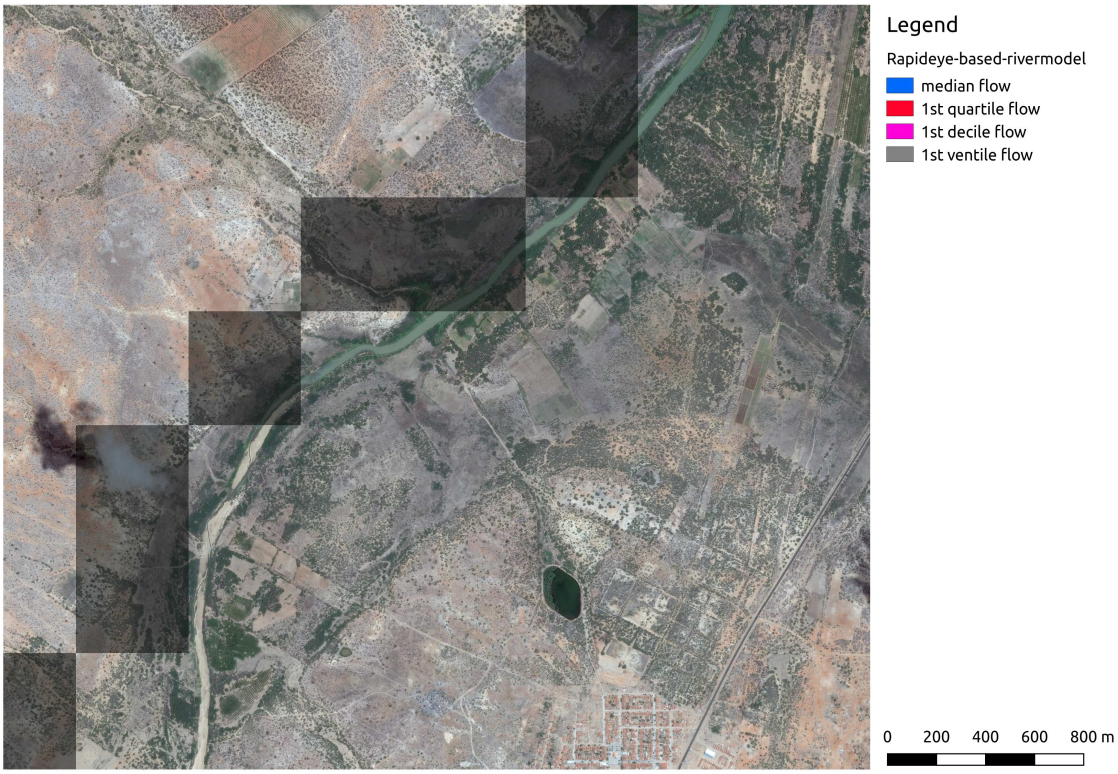


Figure A.4: Comparison of 1st ventile flow to satellite images in the northeast of Brazil

This illustration shows the end of a river of the 1st ventile flow model from Rapideye data which coincides with the end of a navigable river in the dry northeast of Brazil.

A.5 Software Tools to Create Accessibility Maps

The following software and data alternatives exist to the AccessibilityMaps package:

- ArcGIS from ESRI with the Spatial Analyst extension has a completely integrated solution to create friction and travel-time maps, however, it comes with license fees that may prohibit the usage in certain contexts.
- A global friction map is provided by Weiss et al. as a byproduct of their study on global accessibility (2018). If users do not wish to customize the input data or change the assumptions of the model, this source can help to quickly create travel time maps with GRASS or other GIS software.
- Open Trip Planner (http://www.opentripplanner.org/) is an online platform that provides the possibility to create travel time maps based on Open Street Map data. The maps from OTP are easy to create and the tool offers more flexibility regarding the choice of input parameters. The maps are produced online and can be downloaded as vector maps. However, OTP currently does not offer the possibility to select custom input data. OTP can be also assessed from within R with the development package osrm (https://github.com/rCarto/osrm).
- Flowmap is a software dedicated to analyze flow data and display interactions. It can be used to calculate travel time estimates between two different POIs

(http://flowmap.geo.uu.nl/).

- The current version of the Integrated Land and Water Information System (ILWIS) 4 allows calculating simple travel time maps with a friction map (here called weight map) and a map with sources. See chapter 9 on “neighborhood and connectivity calculations” of the documentation for more information: https://52north.org/files/ilwis/Documentation/chap09.pdf.

A.6 Rationale for Expected Coefficient Effects of the Accessibility Regression Model

Figure A.6 presents our expectations about the effect of road quality on the accessibility coefficients in the dry and wet season. Generally, we expect that land rents decrease considerably for crop plantations because of the risk of losing the harvest due to insufficient road transportation at critical points in time. As for pasture systems, there is no such pronounced difference because cattle can be marketed more flexibly (i.e. at a later stage when dirt roads dry up). In effect, the same amount of travel time required during the dry and wet season is associated with large difference in land rents, which will decrease the coefficient for wet season accessibility in the case of cropland systems. For pasture systems, we would expect similar land rents for wet and dry season travel times and therefore no significant difference between the coefficients.

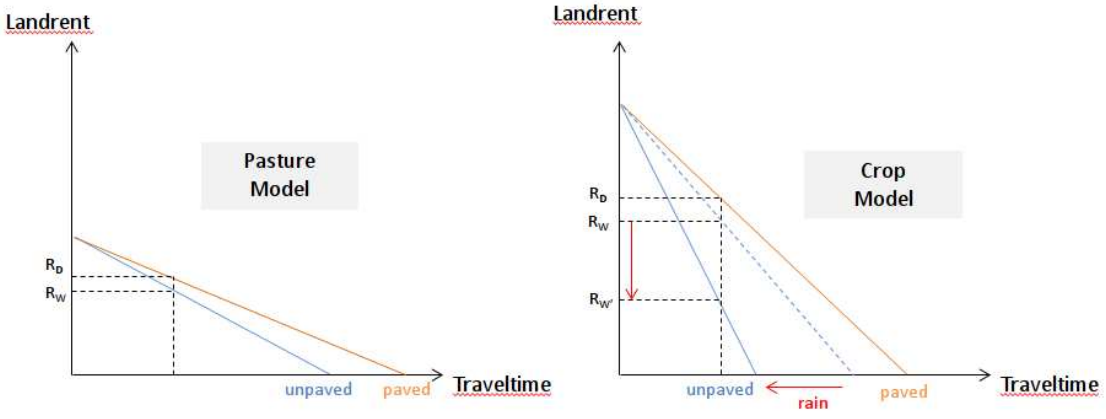


Figure A.6: Effect of infrastructure quality on land rents for pastures and crops in the wet season

A.7 Regression Tables of the Accessibility Model

Table A.2: Tabular results for accessibility models

	Dependent variable:			
	Pasture		Crop	
	PastureDry	PastureWet	CropDry	CropWet
	(1)	(2)	(3)	(4)
Slaughterhouse (time-improvements in hs)	0.025*	0.012***	-0.131***	-0.030***
	(0.015)	(0.004)	(0.030)	(0.005)
Silo (time-improvements in hs)	0.012	0.002	0.436***	0.134***
	(0.019)	(0.007)	(0.048)	(0.013)
Urban Area (time-improvements in hs)	0.129***	0.041*	-0.230	0.155***
	(0.031)	(0.022)	(0.162)	(0.048)
Speculation Area (time-improvements in hs)	-0.010	-0.007	-0.059	-0.057**
	(0.012)	(0.008)	(0.055)	(0.025)
Urban Population growth (1.000 Persons)	0.006***	0.009***	0.0002	-0.003
	(0.002)	(0.002)	(0.007)	(0.007)
Urban GDP growth (1 Mio. BRS)	-0.054	-0.110*	-0.479	-0.339
	(0.058)	(0.065)	(0.322)	(0.266)
Fines (Amount issued)	0.148**	0.191***	0.100	0.153*
	(0.060)	(0.059)	(0.094)	(0.092)
Protected Areas (0/1)	-0.535***	-0.706***	-0.515	-1.434**
	(0.161)	(0.158)	(0.589)	(0.588)
Price-Index	-0.770**	-1.470***	-0.299	-1.444**
	(0.309)	(0.293)	(0.713)	(0.561)
Traveltime Slaughterhouse	-0.034**	-0.021***	0.072***	0.012***
	(0.014)	(0.004)	(0.026)	(0.004)
Traveltime Silo	-0.150***	-0.001***	-0.038	-0.0002
	(0.012)	(0.0001)	(0.063)	(0.0002)
Traveltime Urban Area	0.014	-0.003	-0.596***	-0.160***
	(0.010)	(0.002)	(0.036)	(0.008)
Traveltime Speculation Area	0.004	-0.003	-0.046***	0.024***
	(0.010)	(0.003)	(0.017)	(0.004)
Observations	1,028	1,028	501	501
Note:		*p<0.1; **p<0.05; ***p<0.01		

A.8 Alternative Specifications for the Accessibility Regression Model

The following plots in this section show the crop model in the dry season for crop expansion from 2012-2015, 2012-2016 and 2012-2017 as well as pasture models below. The present alternative specifications suggest that the sign and strength is comparable amongst many coefficients, independent from the temporal specification. There are changes in the strength of the coefficients, especially regarding initial travel time covariates; however, due to large overlaps in their standard errors, we argue that these changes are not relevant for the discussion of our results.

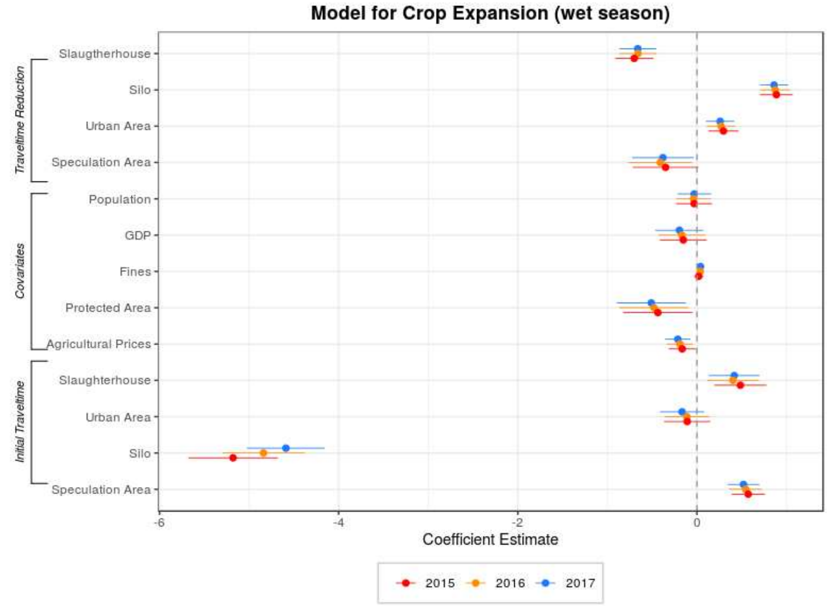


Figure A.7: Coefficient plot for different crop model specifications in the wet season

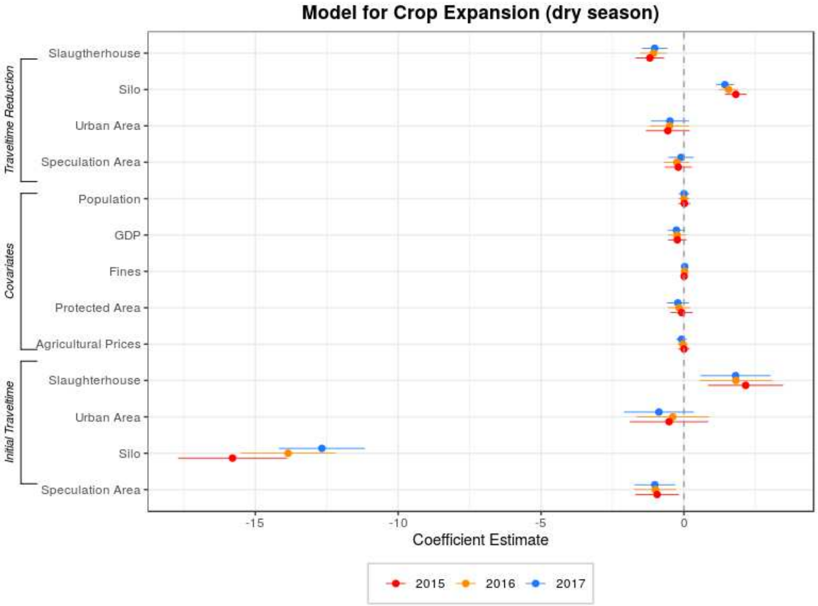


Figure A.8: Coefficient plot for different crop model specifications in the dry season

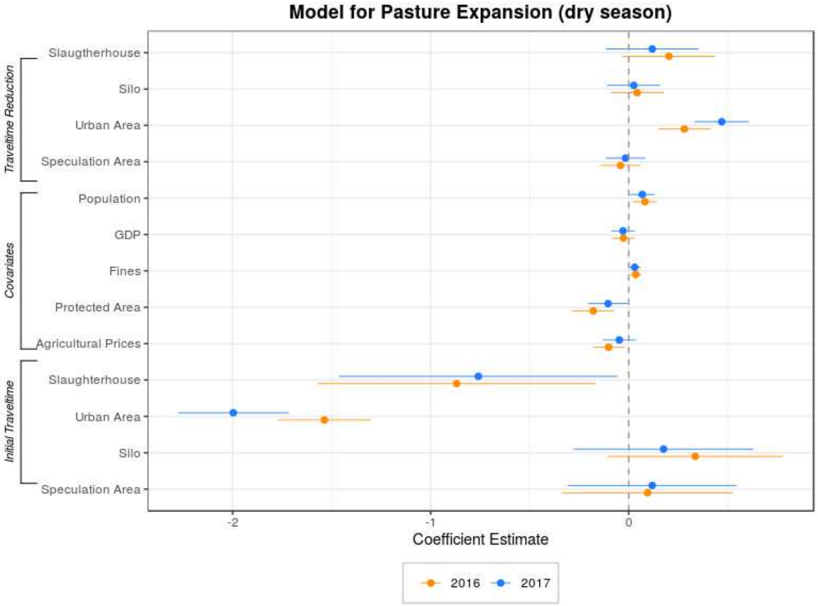


Figure A.9: Coefficient plot for different pasture model specifications in the dry season

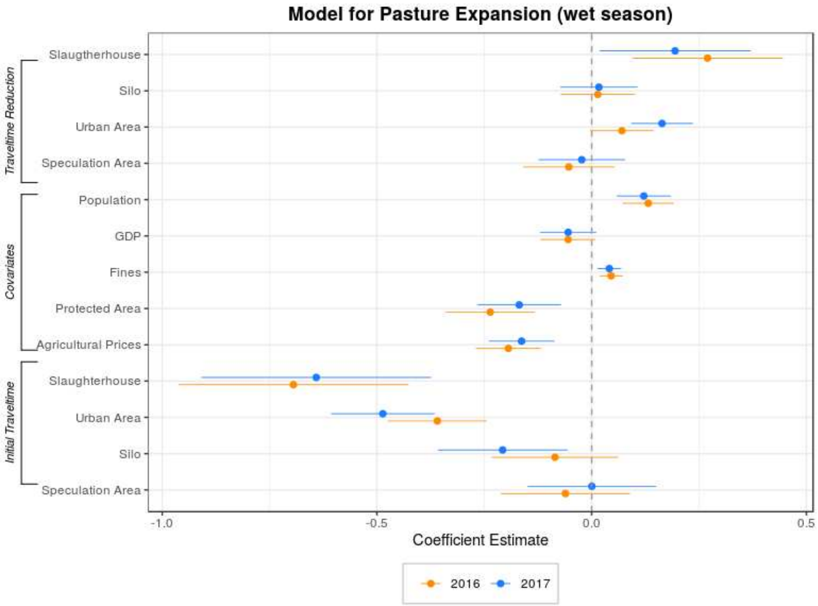


Figure A.10: Coefficient plot for different pasture model specifications in the wet season

A.9 Accessibility and LULCC Literature Assessment

Study	**Region**	**Variables to measure accessibility**	**Discussion of concept** Does the author discuss his concept of distance measurement thoroughly (in article or supplement)?	**Discussion of data quality** Does the author discuss the utilized data-set and eventual shortcomings especially the absence of non-official roads?
(Andersen and Reis 1997)	Brazilian Amazon	distance to the federal capital extension of the road network length of main rivers in region (with more than 2.1 meters of depth at least 90% of the time) level of clearing in neighboring municipalities	**access of who**: farmers **access to what** (what is a market): no **access when** (dry season, wet season): no **access how**: road	No, but points to previous work
(Kenneth M Chomitz and Gray 1995)	Belize	Integrated distance to markets (cities) (cumulative impedance, similar to cumulative cost)	**access of who**: farmers **access to what** (what is a market): city/town **access when** (dry season, wet season): no **access how**: road	yes
(K. M. Chomitz and Thomas 2003)	Brazilian Amazon	Proportion of land within 50 km from main federal roads Distance (buffers) to cities with populations > 25,000 Distance (buffers) to cities with populations > 100,000	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): yes **access how**: road	yes
(Cropper, Griffiths, and Mani 1999)	Thailand	Road density Distance to Bangkok	**access of who**: typical farmer **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road	no
(Cropper, Puri, and Griffiths 2001)	Thailand	Impedance-weighted distance (cost) to nearest market town (Costdistance module in Arc/Info) (roads/rivers)	**access of who**: no **access to what** (what is a market): market towns **access when** (dry season, wet season): no **access how**: road, river	no

Table A.3: Literature Assessment of Accessibility and LULCC

Study	**Region**	**Variables to measure accessibility**	**Discussion of concept** Does the author discuss his concept of distance measurement thoroughly (in article or supplement)?	**Discussion of data quality** Does the author discuss the utilized data-set and eventual shortcomings especially the absence of non-official roads**?**
(Deininger and Minten 2002)	Mexico	straight line distance to the nearest paved road *experimented with friction-weighted distances, results did not differ too much from those reported population density	**access of who**: no **access to what** (what is a market): no **access when** (dry season, wet season): no **access how**: road	no
(Etter et al. 2006)	Colombia	Distance to towns Distance to rivers Distance to roads	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: rivers, roads	no
(Geoghegan et al. 2001)	Mexico	Distance to roads Distance to market Distance to village	**access of who**: farmers **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road	no
(Kirby et al. 2006)	Brazilian Amazon	Mean distance to paved and unpaved roads	**access of who**: no **access to what** (what is a market): no **access when** (dry season, wet season): yes **access how**: roads, mentions importance of rivers	No, but points out/demonstrates the importance of unpaved/secondary roads
(McConnell, Sweeney, and Mulley 2004)	Madagascar	Distance from village (continuous surface of values increasing radially from each centroid, with pixel values representing the distance in meters from the nearest village centroid)	**access of who**: farmers **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: no	not applicable (no use of infrastructure)

Table A.3: Literature Assessment of Accessibility and LULCC (cont.)

Study	**Region**	**Variables to measure accessibility**	**Discussion of concept** Does the author discuss his concept of distance measurement thoroughly (in article or supplement)?	**Discussion of data quality** Does the author discuss the utilized data-set and eventual shortcomings especially the absence of non-official roads**?**
(Mertens et al. 2002)	Pará, Brazil	Distance to main road Distance to secondary road Distance to village Distance to town distance to the nearest river distance to the nearest dairy industry	**access of who**: farmers **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road	no
(Mertens et al. 2004)	Bolivia	Distance to roads and to Santa Cruz	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road	yes
(D. Müller and Munroe 2005)	Vietnam	Distance to long established road network accumulated transportation cost (to define “accessibility catchments”)	**access of who**: no **access to what** (what is a market): village centroids **access when** (dry season, wet season): no discussion, but uses roads that are available all-year only **access how**: road	no
(Daniel Müller and Zeller 2002)	Vietnam	Euclidean distance to nearest all-year road for trucks Euclidean distance to district centers Lagged travel time to all-year road availability of primary schools in years since their opening	**access of who**: no **access to what** (what is a market): district capitals *access to education **access when** (dry season, wet season): no discussion, but uses roads that are available all-year only **access how**: road/trucks	no

Table A.3: Literature Assessment of Accessibility and LULCC (cont.)

Study	**Region**	**Variables to measure accessibility**	**Discussion of concept** Does the author discuss his concept of distance measurement thoroughly (in article or supplement)?	**Discussion of data quality** Does the author discuss the utilized data-set and eventual shortcomings especially the absence of non-official roads**?**
(Munroe, Southworth, and Tucker 2002)	Honduras	Least cost path to nearest village Least cost path out of region (other national markets)	**access of who**: no **access to what** (what is a market): center of exchange (cities, towns, villages) **access when** (dry season, wet season): no **access how**: road	yes
(Naidoo and Adamowicz 2006)	Paraguay	distance to the nearest road distance to the only paved road in reserve distance to nearest town	**access of who**: no **access to what** (what is a market): no **access when** (dry season, wet season): no **access how**: road	no
(G. C. Nelson and Hellerstein 1997)	Mexico	Least-cost route to nearest road Least-cost route to nearest village Least-cost route to large population center Euclidean distances to same above	**access of who**: yes **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: yes	no
(G. C. Nelson, Harris, and Stone 2001)	Panama	Cost of access to border Cost of access to port Cost of access to village Cost of access to nearest town	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: roads, rivers, gulf	no
(G. Nelson et al. 2004)	Panama	Cost of access to border Cost of access to port Cost of access to village Cost of access to nearest town	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road, river, gulf	no

Table A.3: Literature Assessment of Accessibility and LULCC (cont.)

Study	**Region**	**Variables to measure accessibility**	**Discussion of concept** Does the author discuss his concept of distance measurement thoroughly (in article or supplement)?	**Discussion of data quality** Does the author discuss the utilized data-set and eventual shortcomings especially the absence of non-official roads**?**
(Pender et al. 2004)	Uganda	Change in distance to tarmac roads Change in distance to rural market	**access of who**: farmers **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road	no
(Pendleton and Howe 2002)	Bolivia	Walking time to roads Walking time to closest market	**access of who**: small farmers **access to what** (what is a market): city **access when** (dry season, wet season): yes **access how**: walking	yes
(Pfaff 1999)	Brazilian Amazon	Density of unpaved roads Density of paved roads Density of rivers Distance from county seat to state seats (state markets) Distance from county seat to national seats (national markets)	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road, river	Discusses data quality for rivers and roads, but no shortcomings
(Pichon 1997)	Ecuador	Distance to roads Distance to nearest market town	**access of who**: colonists/farmers **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: mechanized travel, foot or canoe	no
(Reis and Guzmán 1994)	Brazilian Amazon	Extension of unpaved roads Extension of paved roads Extension of rivers Distance to state capital Distance to Brasilia	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road, river	yes

Table A.3: Literature Assessment of Accessibility and LULCC (cont.)

Study	**Region**	**Variables to measure accessibility**	**Discussion of concept** Does the author discuss his concept of distance measurement thoroughly (in article or supplement)?	**Discussion of data quality** Does the author discuss the utilized data-set and eventual shortcomings especially the absence of non-official roads**?**
(Serneels and Lambin 2001)	Kenya	Distance to roads Distance to nearest settlement Distance to game lodges Distance to Narok (district seat) Distance to water	**access of who**: farmers **access to what** (what is a market): yes **access when** (dry season, wet season): yes (discussed but not measured) **access how**: road	no
(Southworth et al. 2004)	Honduras	Distance to roads Distance to town	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road	no
(Tucker et al. 2005)	Honduras and Guatemala	Weighted cost of access (least cost path) to nearest town/local market Weighted cost of access (least cost path) out of region (capital city or regional market)	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: roads and foot paths	no
(Vance and Geoghegan 2002)	Yucatan	On-road distance to nearest market	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road	no
(Wilson et al. 2005)	Chile	Distance to roads Distance to town	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road	no

Table A.3: Literature Assessment of Accessibility and LULCC (cont.)

Study	**Region**	**Variables to measure accessibility**	**Discussion of concept** Does the author discuss his concept of distance measurement thoroughly (in article or supplement)?	**Discussion of data quality** Does the author discuss the utilized data-set and eventual shortcomings especially the absence of non-official roads**?**
(Frey et al. 2018)	Brazilian Amazônia	cumulative cost to cities, ports, storage and processing facilities	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no, only for navigable rivers all-year around **access how**: road, river, railway	no
(Schielein and Börner 2018)	Brazilian Amazon	accumulated travel-times to the next municipality center travel-times to the next large urban market	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: roads, rivers, trains	no
(Mena et al. 2011)	Ecuadorian Amazon	Euclidean to the nearest main road	**access of who**: agricultural colonists **access to what** (what is a market): no **access when** (dry season, wet season): no **access how**: road	no
(Walsh et al. 2008)	Ecuadorian Amazon	Euclidean distance to the nearest road distance along the road network (i.e., Network distance) to the closest and distant market or main communities	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road	no
(B. S. Soares-Filho et al. 2006)	PanAmazon	Distance to paved roads Distance to unpaved roads distance to major rivers distance to railways Urban attraction factor	**access of who**: no **access to what** (what is a market): town **access when** (dry season, wet season): no **access how**: road	yes

Table A.3: Literature Assessment of Accessibility and LULCC (cont.)

Study	**Region**	**Variables to measure accessibility**	**Discussion of concept** Does the author discuss his concept of distance measurement thoroughly (in article or supplement)?	**Discussion of data quality** Does the author discuss the utilized data-set and eventual shortcomings especially the absence of non-official roads**?**
(B. Soares-Filho et al. 2010)	Brazilian Amazon	distance to major roads attraction by urban centers	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road	no
(de Espindola et al. 2012)	Brazilian Amazon	Euclidean distances to roads Euclidean distance to large rivers Euclidean distances to urban centers Euclidean distance to wood extraction Euclidean distance to mineral deposits Indicator of connection to ports Indicator of connection to national markets	**access of who**: no **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road, river	no
(Lapola et al. 2010)	Brazil	Road network desity Distance to settlements	**access of who**: no **access to what** (what is a market): no **access when** (dry season, wet season): no **access how**: no	no
(Perz, Walker, and Caldas 2006)	Pará, Brazil	Distance to town	**access of who**: small farmers **access to what** (what is a market): yes **access when** (dry season, wet season): no **access how**: road	no
(Weinhold and Reis 2008)	Brazilian Amazon	Transport cost via paved roads, unpaved roads and rivers from each municipio to each market distance to state and federal market, length of navigable river	**access of who**: no **access to what** (what is a market): no **access when** (dry season, wet season): no **access how**: roads, rivers	no

Table A.3: Literature Assessment of Accessibility and LULCC (cont.)

Study	**Region**	**Variables to measure accessibility**	**Discussion of concept** Does the author discuss his concept of distance measurement thoroughly (in article or supplement)?	**Discussion of data quality** Does the author discuss the utilized data-set and eventual shortcomings especially the absence of non-official roads**?**
(Aguiar et al. 2016)	Brazilian Amazon	Euclidean distance to the closest paved road Euclidean distance to the closest road Euclidean distance to the closest timber extraction and processing center	**access of who**: no **access to what** (what is a market): no **access when** (dry season, wet season): no **access how**: road	no

Table A.3: Literature Assessment of Accessibility and LULCC (cont.)

APPENDIX B

Supplementary Material of chapter 3: LULCC Dynamics in Deforestation Frontiers

The geographical data with deforestation frontiers is published as: Schielein, J. (2018). Deforestation Frontiers in the Brazilian Amazon (Harvard Dataverse No. V1). https://doi.org/doi:10.7910/DVN/AGLYOW

B.1 Identification of Study Cells

To identify deforested areas, we used the comprehensive PRODES archive, which provides data at a 60x60 meter resolution (INPE 2017). The PRODES raster layer was rescaled to 1x1 km grid cells and only cells that experienced deforestation until 2015 were included in the analysis. These "study cells" can be either completely or partially deforested to form part of the analysis. Partially deforested cells may contain forest areas and other classes from the PRODES base layer. The excluded non-deforested grid-cells can be either forest, non-forest, water bodies, unclassified areas, or areas below clouds in 2015, or a combination of these conditions.

B.2 Accessibility Map

The travel-cost maps were created using data from IBGE on the location of municipality centers (IBGE 2010) and their estimated urban population in 2010 (IBGE 2011) in conjunction with a friction surface. The friction surface is based on infrastructure and land cover maps and contains estimated travel times to cross any given grid-cell in horizontal or vertical space using either car, boat, or train transportation as well as estimated walking times (depending on the surface). The friction map was produced on a 90x90 meter grid using a compilation of different vector and raster layers.

Data on primary roads, hydroways, and train tracks were obtained from the Brazilian Ministry of Transportation (Ministério dos Transportes - MT 2009) and data on secondary roads was obtained from a classification study of Landsat images in 2004 (Soares-Filho, Nepstad, et al. 2006). This data-set can be downloaded on the Center of Remote Sensing/Federal University of Minas Gerais' website http://maps.csr.ufmg.br/.

Data on rivers was obtained from the Brazilian Water Agency which offers a drainage model for Brazil based on SRTM data in 2000 (Agência Nacional das Águas-ANA 2000). We used the lower available resolution scale of 1:250.000 to identify major rivers. Each line in this data-set was buffered with a buffer of 10 km on each side insuring the inclusion of main streams and their respective affluents. The buffered areas were than used to clip out the higher resolution data of 1:100.000. The latter more accurately predicts the location and actual length of a given body of water, but is too rich in detail to accurately identify the larger and therefore potentially navigable rivers. The resulting line-data was combined with polygon-data showing the spatial extension of large bodies of water such as the Amazon river, which can have a width of several kilometers. Remote sensing data containing surface reflectance of large bodies of water was obtained from the Brazilian Ministry of Environment (Ministério do Meio Ambiente-MMA 2010) and land cover data was obtained from The Mapbiomas project (MapBiomas Project, 2017). Land cover classes were reclassified into open-landscape and forest classes.

The friction map was created by rasterizing all data-sets in a 90x90m grid and collecting the layers in a raster stack. Each layer was reclassified to contain travel speeds based on its containing surface types. Our assumptions on travel speed are based on a publication of global accessibility (Nelson 2008) and were further validated and modified during field research in the Brazilian Amazon in 2015 and 2016:

- Paved Roads: 60km/h
- Unpaved Roads: 40 km/h
- Bodies of water or navigable stretches in the flooding season: 10 km/h
- Hidroways: 20km/h
- Railways: 40km/h
- Forest: 3 km/h (comprises in the original data: forest, planted forest, coastal zone forest)
- Non-forest: 12 km/h (comprises: Agricultural areas, non-forest vegetation, pastures, others)

The reclassified and rasterized data-stack was reduced to a single raster layer by obtaining the highest travel speed for each cell in the raster stack. Furthermore, the effect of slope on travel-speed was included (for all non-water surfaces) using SRTM topography data (USGS, 2000). We use this data to multiply a constant slope factor that reduces travel speed with the following assumptions:

- slope between 0 and 5 degrees: slope constant 1 (travel speed is not impacted)
- slope between 5-10 degrees: slope constant 2 (travel speed is reduced by half)
- slope between 10-15 degrees: slope constant 3 (travel speed is reduced to a third of the original speed)
- slope above 15 degrees: slope constant is 10 (travel speed is reduced to 10% of the original speed)

The result of this calculation is a corrected friction map showing the average travel time to cross a given grid-cell in seconds. This friction map allows for the creation of aggregated travel time maps in the Brazilian Amazon. Accumulated travel time maps were created with the GRASS algorithm r.cost (Neteler et al. 2012) using the knightmove option to get more accurate results. The initial calculations were done in a 90x90m grid. The aggregated travel time maps were then rescaled to a 1km resolution by taking the average travel time of the merged grid-cells. The following map shows calculated travel times to the next municipality center for the whole Brazilian Amazon region.

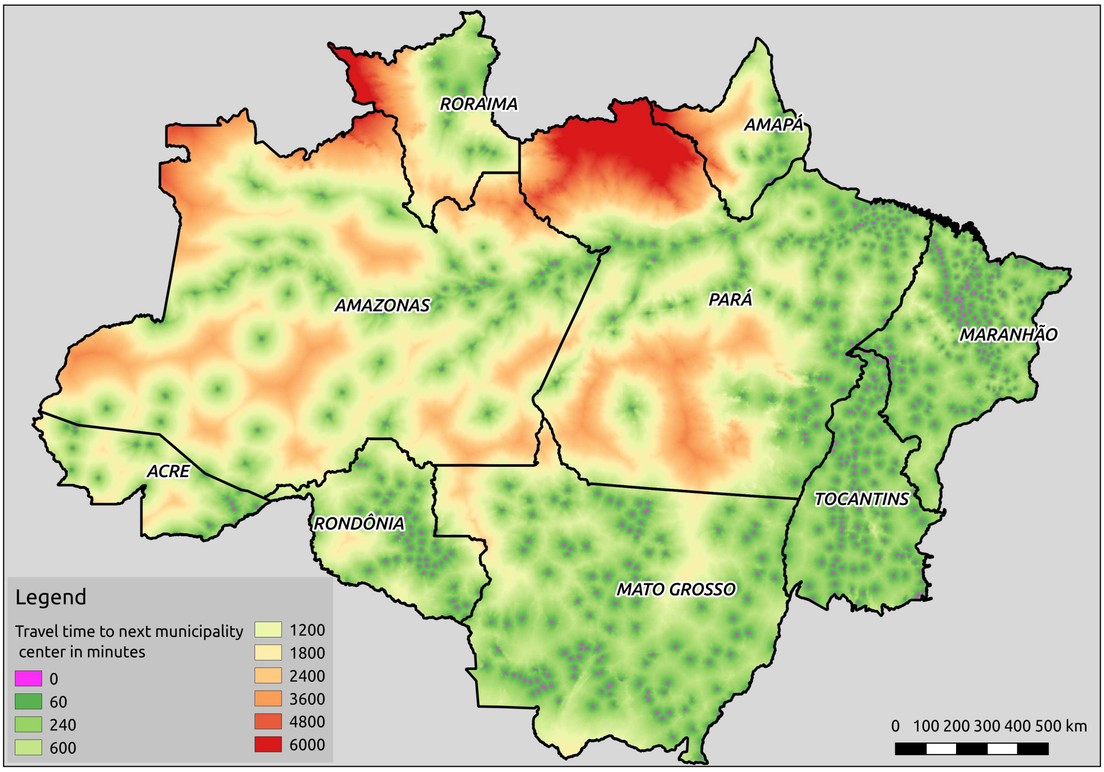


Figure B.1: Accessibility map to municipality centers in the Brazilian Amazon

B.3 Population Data

The original polygon data from IBGE contains the total number of inhabitants in a 1x1km grid for rural and in a 100x100m grid for urban populations (do Carmo Bueno 2016). This data was homogenized in a 1x1km resolution and then aggregated to 10x10km raster. Aggregation was applied because the original data shows where people live with very high precision, but does not indicate where they interact with the landscape. The lower resolution expands the potential zone of influence of population centers to their surrounding landscape. After this, the data was rescaled again to a 1x1km grid to match our unit of analysis.

B.4 Settlement Data

For improving the overall quality of INCRA data, first, the year of settlement creation was extracted and subset for observations equal or prior to the year 2004. Furthermore, settlements with a low family/area ratio were excluded. The latter was applied because some settlements had extensive areas for a small amount of actual settlers. A settlement with a low family to area ratio was defined as a settlement that has more than 500 hectares per family. This was true in 26 cases that were removed from the dataset. Furthermore, 11 settlements did not have any family living there yet and as a consequence they had been removed from the dataset as well. In a last processing step, exact duplicates were removed from the polygon dataset (79 cases).

B.5 Origins Variable

Equation B.1 describes how the variable northerness was created:

$$n_i = \frac{1}{P_i} \cdot \sum_{s=1}^{n=27} p_{si} \cdot n_s \tag{B.1}$$

where

n_i is the northerness of a given municipality i,

P_i is the total population in municipality i,

s are the federal states of Brazil,

p_{si} is the population originating from a given state s in municipality i,

n_s are scaled latitudinal coordinates of the state centroid from a given state s.

Because state names are categorical, there was a need to convert them into a numerical variable (n_s) indicating the geographic location of each state. This was achieved by extracting the geographical coordinates of the center of each state (centroids) and scaling them between 0 and 1. A value of 0 (south) is represented by the centroid of the state of Rio Grande do Sul and 1 (north) by the centroid of Roraima state in the north. In the case of "easterness" the longitudinal coordinates were utilized where a value of 0 (west) is represented by the centroid of Acre and 1 is represented by the centroid of Paraíba state. All other states share values between 0 and 1 and the differences amongst them is defined by the geographical distance of their centroids in the coordinate space. The final variables on municipality scale were rasterized to a 1x1 km grid to match the unit of analysis.

B.6 Determining the Optimal Number of Clusters in CLARA

The optimal number of clusters in CLARA can be identified based on a variable called average silhouette width s. The s variable is a measurement for the overall cluster-stability, derived from the silhouette widths of all clustering objects in a given cluster solution. The silhouette width s of a given object i in cluster a is given by:

$$s(i) = \frac{b(i) - a(i)}{max\{a(i), b(i)\}} \tag{B.2}$$

where a(i) is the average dissimilarity of i to all other objects in cluster A, hence describing how different i is to the other objects that form part of its cluster. And, b(i) is the minimum distance of i to the next object in cluster B (which is the most similar cluster to A), hence describing how different i is to the closest object in its next neighboring cluster.

B.7 Agricultural Census Data

In order to describe the agro-economic characteristics of different frontier types, data from the Brazilian Agricultural census from 2006 was utilized on the scale of census tracts. Census tracts are the most disaggregated format to obtain census data and variables are only released through a formal request to IBGE, which was approved in 2015. The data comes in tabular form and has to be merged with spatial data showing the distribution of census units, which was obtained from the IBGE website for the year 2007 (IBGE 2007).

Matching the data based on geocodes yields in a 92.4% match, hence 7.6% of the original tabular data cannot be clearly attributed to a census tract. Census tracts differ considerably in size and population and the data had to be normalized based on a reliable area measurement. For this purpose land cover data from the TERRACLASS project (Coutinho et al. 2013) was obtained to estimate the total agricultural area within each census tract. The total agricultural area was calculated for 2004 and consists of pastures, annual agriculture and mixed land uses. Compared to the reported farm areas from the census, these estimates were more reliable, because reported farm areas from the census may fall into multiple census tracts. All agro-economic variables were normalized by total agricultural area using this procedure. Descriptive statistics of the standardized data revealed some extreme outliers which were removed from the data-set before further processing. This was done by deleting observations with values above the 95^{th} percentile. The processed data was rasterized, masked and combined with the study-cells data to calculate average values for each variable in each cluster. The zonal statistics function from the raster package in R was used for this task (Hijmans 2019). Two frontier clusters in remote areas had very few observations from the census because IBGE conducted very few interviews in those regions. They were not included in the descriptive analysis.

B.8 Deforestation Data

From the comprehensive PRODES database, binary maps for each year were created and the area of each cluster was extracted separately. Based on these maps, summary statistics were calculated regarding the cumulative amount of deforested areas between 2004 and 2015 in each grid-cell. From the cumulative areas, relative deforestation shares per grid-cell were calculated and then summarized per cluster. These shares exclude bodies of water and non-forest areas from the total grid area, hence shares in deforested area refer to the potential forest area before deforestation occurred. To create a dataset of deforestation patches the PatchStat function from the SDMTools package in R was used (VanDerWal et al. 2014).

To calculate these statistics, we processed large amounts of spatial data, which required tiling the region into 32 several parts and processing them on several CPUs in parallel. This procedure might, in some cases, lead to the division of connected patches into different tiles. This procedure might, therefore, decrease the size of some patches in boundary regions. However, a visual assessment of the tiled data showed that this effect is negligible in light of the overall amount of deforestation patches and their spatial distribution in the study region. The result of the patch analysis is a matrix where deforestation patches are characterized in terms of size and complexity.

B.9 Clustering Results

Figure B.2 shows the average Average Silhouette Widths of the first clustering process. Folloqing equation B.2, s(i) can lie between -1 and 1. The closer s(i) is to 1, the more similar it is to its cluster members and/or dissimilar it is to the closest object in another cluster group. By averaging s(i) over all cluster-observations, we calculate s, which indicates the robustness of the overall clustering process. Variable s should be at least 0.5 to be an acceptable solution based on the experiences of Kaufman and Rousseeuw (2005). For this study we used the six cluster solution.

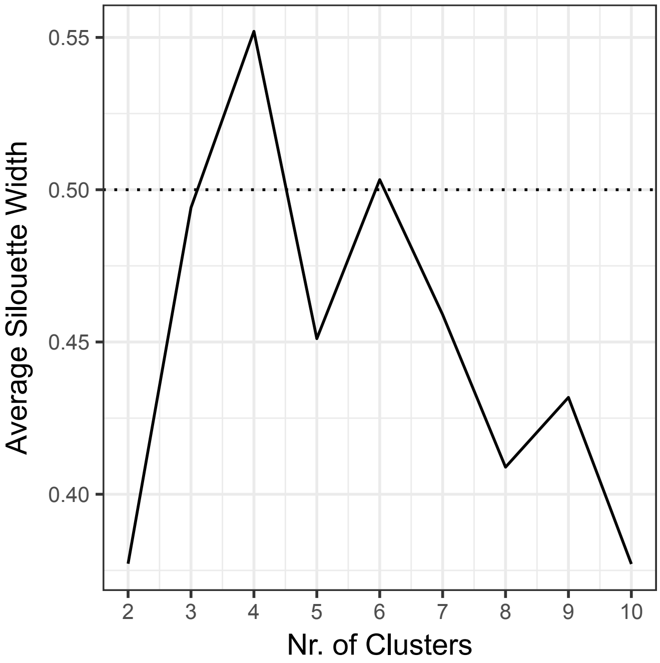


Figure B.2: Average silhouette widths of the first clustering process

Figure B.3 shows descriptive statistics for the six-Cluster solution before the second clustering step.

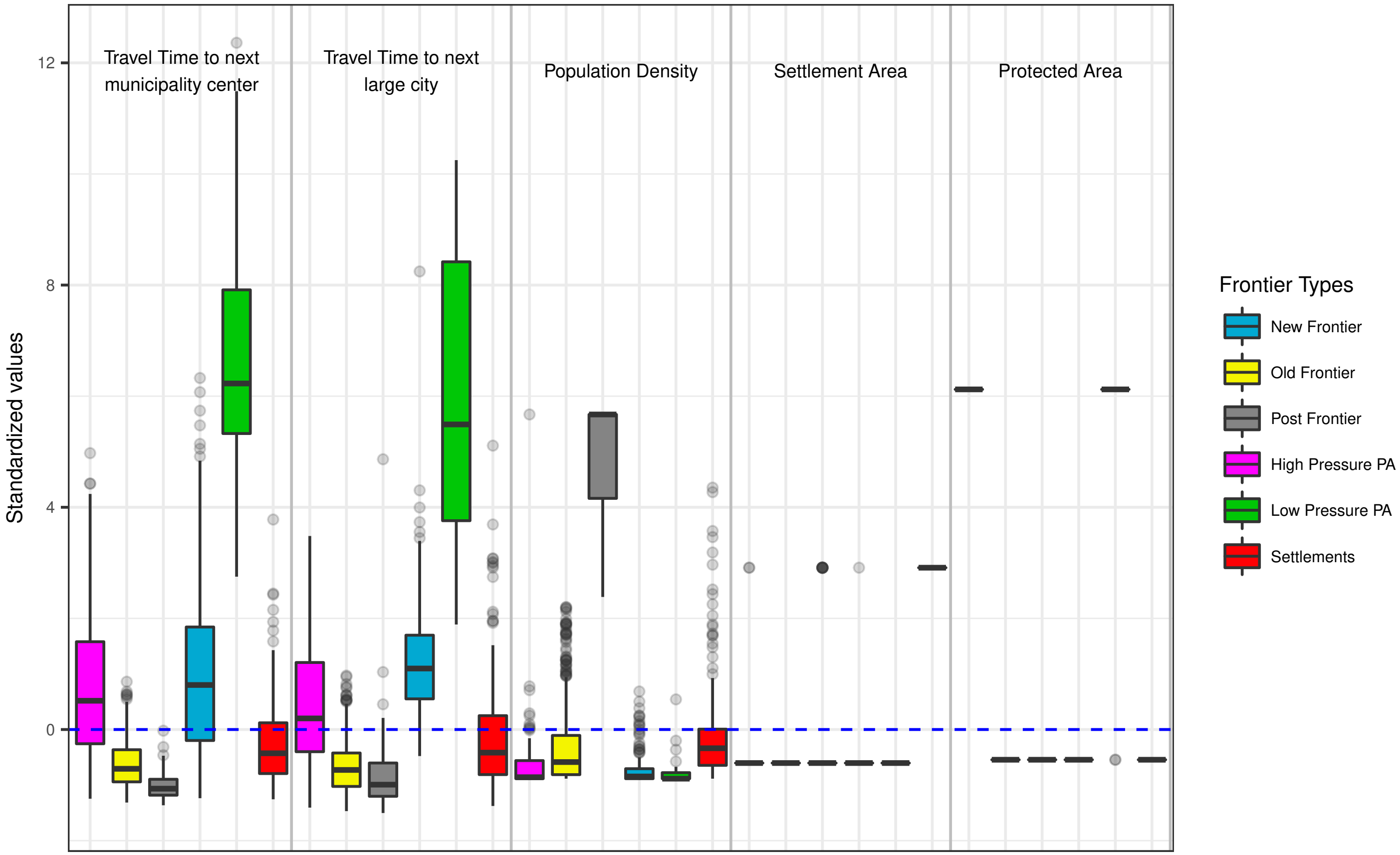


Figure B.3: Boxplot of the six cluster solution from the first clustering step

B.10 Tabular Overview of LULCC Trends in Different Deforestation Frontier Regions

Table B.1: Tabular overview of land cover trends in the analyzed frontier regions

Cluster	Annual Agriculture	Mixed landuse	Clean Pasture	Degraded Pasture	Secondary Vegetation	Total	(Pastures combined)
			2004 (percentages)				
Post Frontier	2.06	7.19	50.56	23.06	17.12	100	73.62
Old frontier	4.49	1.94	58.62	19.17	15.78	100	77.79
New frontier	2.31	1.85	54.42	21.17	20.25	100	75.59
High Pressure PA	0.31	5.35	28.96	27.18	38.19	100	56.14
Low Pressure PA	0.00	12.93	6.72	15.33	65.02	100	22.05
NE-Settlement	0.01	4.99	45.45	30.21	19.35	100	75.66
N-Settlement	0.18	7.07	55.68	20.30	16.78	100	75.98
S-Settlement	1.14	2.50	61.71	23.21	11.44	100	84.92
			2014 (percentages)				
Post Frontier	3.19	5.22	46.23	17.47	27.89	100	63.7
Old frontier	8.03	1.56	54.78	13.68	21.96	100	68.46
New frontier	8.13	1.74	55.35	11.78	23.01	100	67.13
High Pressure PA	0.79	6.24	31.06	16.23	45.68	100	47.29
Low Pressure PA	0.00	8.63	12.93	9.74	68.70	100	22.67
NE-Settlement	0.17	2.99	49.95	19.86	27.04	100	69.81
N-Settlement	0.38	4.10	58.75	18.72	18.03	100	77.47
S-Settlement	4.46	1.27	68.07	11.00	15.21	100	79.07
			Change 04/14				
Post Frontier	1.13	-1.97	-4.33	-5.59	10.77	-	-9.92
Old frontier	3.54	-0.38	-3.84	-5.49	6.18	-	-9.33
New frontier	5.82	-0.11	0.93	-9.39	2.76	-	-8.46
High Pressure PA	0.48	0.89	2.10	-10.95	7.49	-	-8.85
Low Pressure PA	0.00	-4.30	6.21	-5.59	3.68	-	0.62
NE-Settlement	0.16	-2.00	4.50	-10.35	7.69	-	-5.85
N-Settlement	0.20	-2.97	3.07	-1.58	1.25	-	1.49
S-Settlement	3.32	-1.23	6.36	-12.21	3.77	-	-5.85

B.11 Implications of Using Patch Size as an Indicator for Deforestation Agents

It can be argued that linking remote sensing images to agents on the ground requires additional efforts for improving the overall reliability of the results. Our view is as follows: First, PRODES uses visual interpretation to classify deforested areas and create a spatial data-base of vector data that contains deforestation patches assigned to years. Consequently, human interpretation is the basis for delimiting deforestation patches which might cause mis-classification in some cases (the same applies to machines as well). Since the data has to be vectorized by hand it is reasonable to assume, that congruent deforestation patches might be vectorized as a single patch to save time, hence the size of deforested patches might be overestimated. Second and more importantly, PRODES reports deforestation areas in the year of their appearance and large areas are often covered by clouds for several consecutive years. Consequently, smallholder deforestation that is congruent might appear only after several years of cloud coverage and this data is then digitalized as a single polygon. The result is, again, an overestimation of the actual deforested patch area. Figure B.4 shows a very large and complex patch (>400 km^2) in Brasil Novo (Pará) that was classified as a single polygon in PRODES in 2004 but is most probably caused by several agents over several consecutive years before 2004.

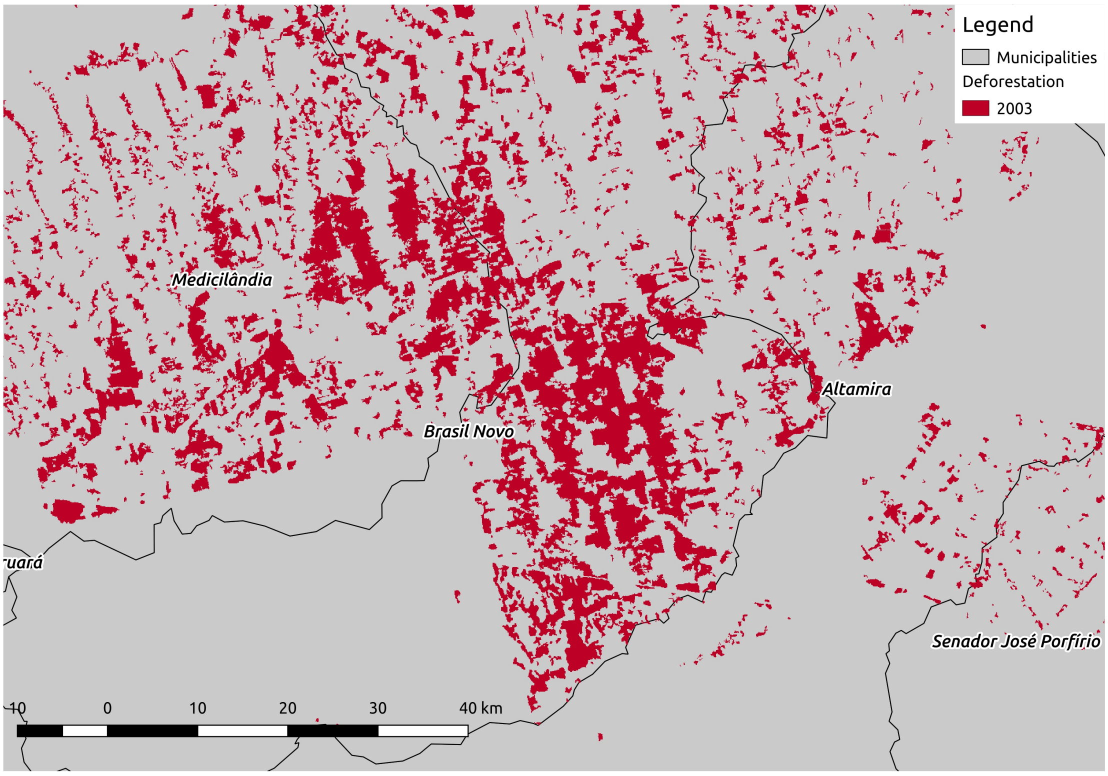


Figure B.4: Large and complex deforestation patch from small-scale deforestation

During our work we excluded very complex shapes from the analysis to see if this would impact our results. Very complex shapes were defined as shapes where the Shape Index S is larger than 2.5. S is the actual perimeter of a patch in terms of cell surfaces divided by its possible minimum perimeter given the number of its cells. If it is equal to 1, a patch is maximum compact. As S increases the patch complexity increases as well. Very large patches that are caused by several agents are most probably very complex. Hence, excluding them from the analysis would remove this error from the estimation. However, it removes some areas of possible smallholder deforestation as well. Figure B.5 gives a comparison of patch sizes with and without complex shapes which indicates that the amount of small-scale deforestation increases between 5 and 10 percentage points per year if complex shapes are removed. However, the overall trend over the years is not influenced.

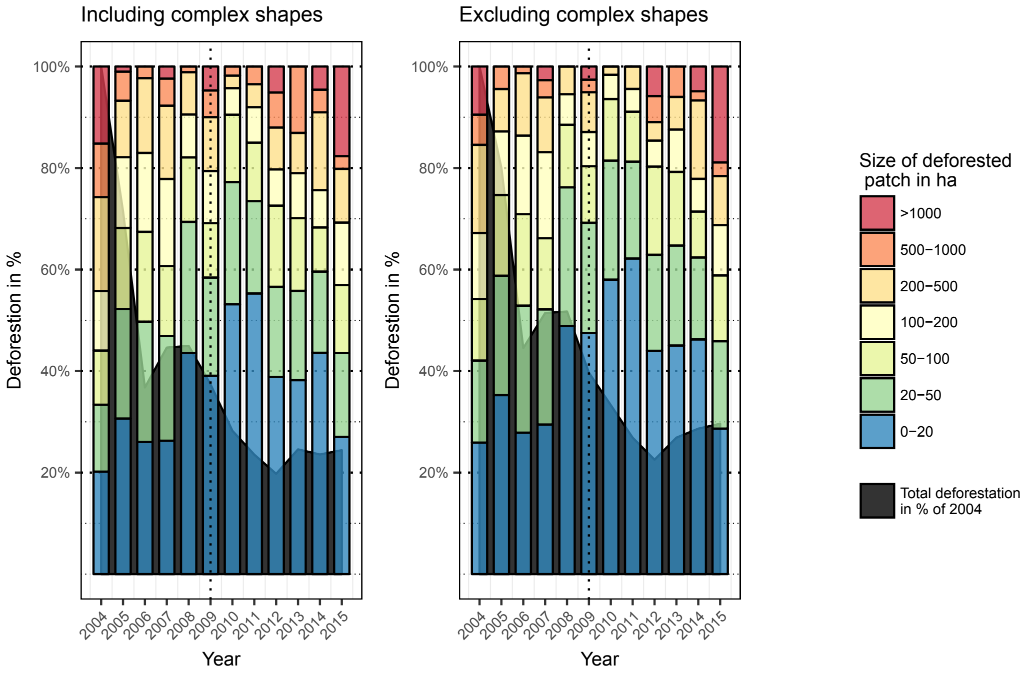


Figure B.5: Comparison of small-scale deforestation with and without very complex shapes

Besides that, it is also difficult to correctly estimate large-scale deforestation patterns since large areas are also often deforested in several subsequent years which attributes in the statistics to small-scale deforestation. However, this deforestation is most probably caused by large landholders. Hence, the share of large-holder deforestation could be underestimated. Figure B.6 shows this phenomenon for a large landholding of 4500 ha in the Altamira municipality in Pará.

In conclusion, patch-statistics should be analyzed with caution since mis-estimations are not thoroughly discussed and quantified.

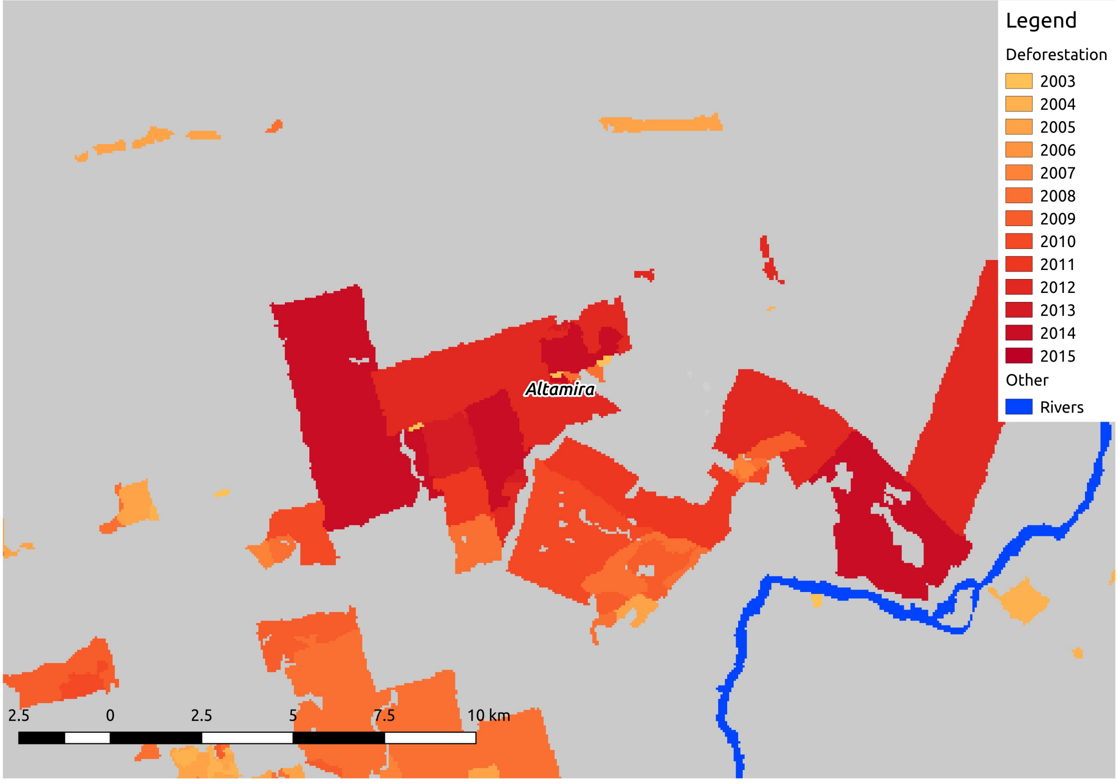


Figure B.6: Large-scale deforestation over consecutive years

B.12 Pasture Stocking Rates in the Brazilian Amazon

The following information describes how we calculated stocking rates based on Animal Units (AU) in Brazil from the year 2003 to 2015. We use the concept of Animal Units (port: "unidades animais") since it is a more reliable estimator for pasture stocking rates than a simple head count of animals per area. Head counts fail to account for the size, and hence fodder requirements of different animal types (i.e. small animals have fewer fodder requirements than larger ones). Therefore, we account for the specific production portfolio of a given municipality (e.g. cow-calve operations vs. fattening) and derive by this a parameter that is better comparable on the country and regional scale. Furthermore, we utilize different sets of land cover time-series data to create a high quality estimation of pasture areas.

Table B.2: Utilized input-data for creating stocking rates

ID	Source	Research Project	Variable	Spatial Scale	Temporal Scale
1	IBGE, 2016	Pesquisa Pecuária Municipal	Cattle Herd-sizes	Administrative Units (municipalities)	2000-2016
2	Coutinho et al., 2013	Projeto TERRACLASS	Pasture areas	30x30m resolution	2004, 2008, 2010, 2012, 2014
3	MapBiomas Project, 2017	Mapbiomas Project	Pasture areas	30x30m resolution	2003-2015
4	IBGE, 2009	Censo Agropecuário 2006	Shares of different cattle animal classes	Administrative Units (municipalities)	2006

(1) Creation of AU profiles per municipality

First a conversion was applied of the shares of different cattle animal classes (2) into AU. The following parameters were utilized:

- Animals of less than 1 year: 0.25 AU
- Animals of 1-2 years: 0.5 AU
- Female animals of more than 2 years: 1 AU
- Male animals of more than 2 years: 1.25 AU
- Working Animals of more than 2 years: 1.5 AU

From this, an average AU share for each municipality was calculated. It is assumed that this share is constant over time (2003-2015).A histogram of these shares is given in the figure below. Notably the mode is below 1 meaning that there are more municipalities with small animals rather than larger ones. Furthermore, we observe a significant spread in the distribution which confirms our hypothesis that animal units are more accurate to describe pasture requirements than a simple headcount.

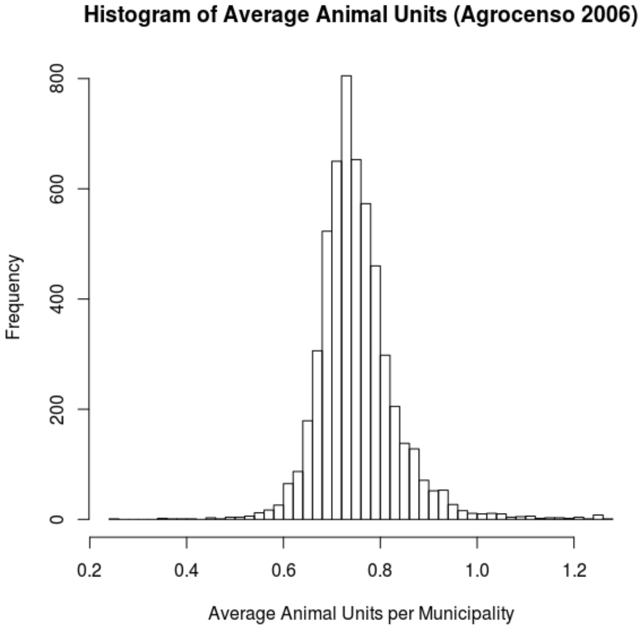


Figure B.7: Histogram of average animal units per municipality in 2006

(2) Creation of a smoothed trend of herd-sizes

A smoothed trend of herd-sizes was created from the raw-data, to account for annual fluctuations in the herd-size following the natural production cycle of the cattle production. This means that farmers do not have a constant stocking rate but their rates change on an annual basis. This change can occur collectively because market prices depend upon animal supply, which in return creates an incentive for farmers to increase or decrease their herds given their current amount of pastures. The smoothing procedure results in an average stocking rate that is presumably more reliable than the raw data. It is assumed that the natural production cycle takes place in a three-year period from birth to slaughter, meaning that the herds grow over a span of three years and then decrease again due to offset of grown animals. Hence, a three-year smoothing average is applied to the raw data. This allows for time-wise comparison of the stocking rates between two specific points in time without being biased from collective herd fluctuations.

(3) Calculate pastures area per municipality

Raw-data from both, the Mapbiomas and the TerraClass project was utilized to calculate pasture areas per municipality. We started using the following categories from Mapbiomas to estimate pasture areas per municipality:

- Pasture (port: pastagem),
- Natural pastures (port: pastagem em campos naturais),
- Other pastures (port: outras pastagens),
- Agriculture or pastures (port: agricultura ou pastagem).

Despite being more comprehensive irrespective of the biome, the data from the Mapbiomas project had a lot of missing values due to cloud coverage at the time it was processed.Those areas of missing values were filled with information from the TerraClass project including the following pasture categories:

- Clean pastures (port: pasto limpo)
- Dirty pastures (port: pasto sujo)
- Pastures with exposed soils (port: pasto com solo exposto)
- Regeneration with pastures (port: regeneração com pasto)

Since TerraClass data is only available for the years of 2004, 2008, 2010, 2012 and 2014 we corrected the complete Mapbiomas annual time-series in the following way:

- Mapbiomas data from 2003 to 2007 corrected with TerraClass data from 2004
- Mapbiomas data from 2008 to 2009 corrected with TerraClass data from 2008
- Mapbiomas data from 2010 to 2011 corrected with TerraClass data from 2010
- Mapbiomas data from 2012 to 2013 corrected with TerraClass data from 2012
- Mapbiomas data from 2014 to 2015 corrected with TerraClass data from 2014

Finally, zonal statistics were calculated per municipality to count pasture cells and convert them into pasture areas.

(4) Calculation of stocking rates per municpality

Smoothed cattle-herd trends were divided by the area of pastures (in ha) per municipality. All in all, we derive 5231 valid observations and 335 missing values. The missing values are unavoidable because some municipalities that are the base of PPM data area not included in the Agrocensus because they were formed only after the Census was completed. Although missing values account only for 6% of all observations they cover some larger areas, especially in the north-western Amazon. The general conclusion of our data-analysis indicates that stocking rates in AU are increasing in most parts of the country, as it can be observed in the following two illustrations.

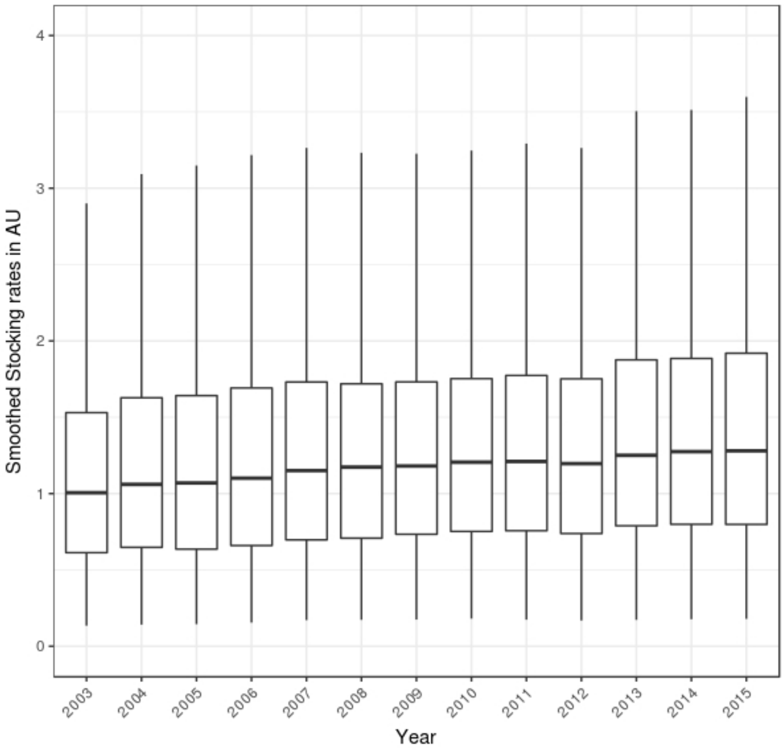


Figure B.8: Boxplot of stocking rate development in Brazil

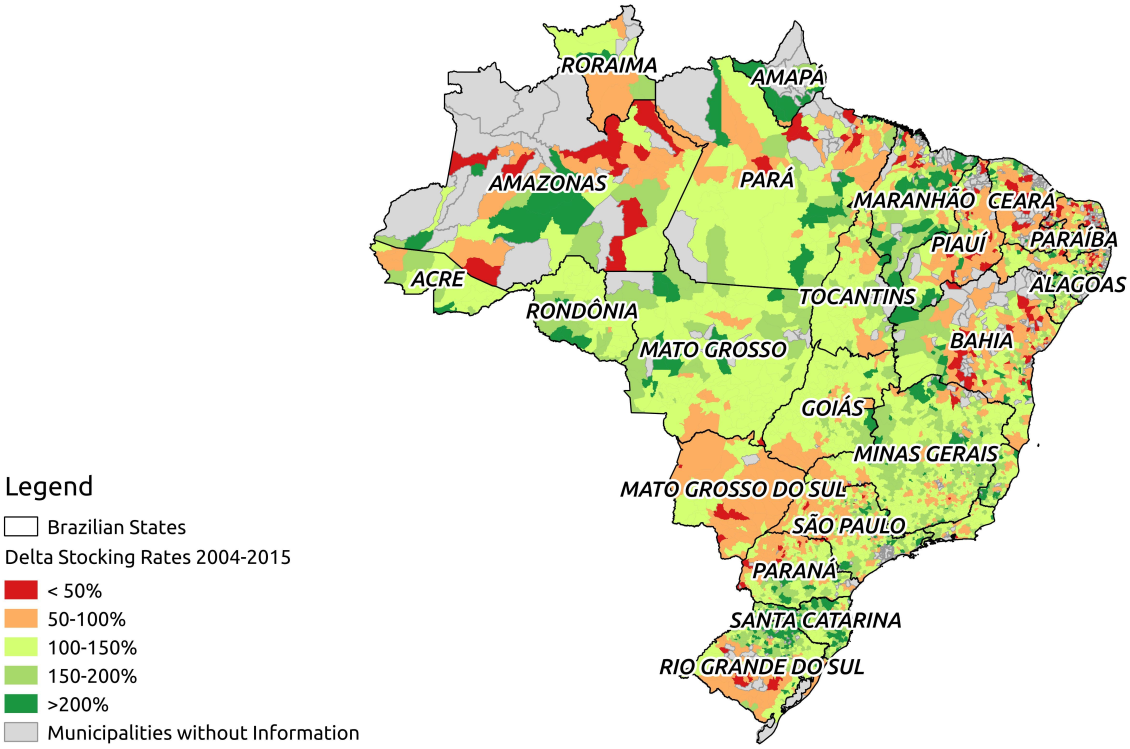


Figure B.9: Map of stocking rate development in Brazil

APPENDIX C

Supplementary Material of chapter 4: Governance and Cattle Intensification in Acre

Part of the data from the survey in Acre is published as: Schielein, J. (2020). Cattle Ranching Trends in Acre 2005-2014. Harvard Dataverse. https://doi.org/10.7910/DVN/TTQWRD.

Researchers are invited to use the dataset for further exploration and publication.

C.1 Sampling Strategy and Survey Design

The target population for our sampling process consists of all the cattle ranchers in the Brazilian state of Acre. However, due to accessibility issues, the survey population is, reduced to municipalities that are accessible by terrestrial (road) transportation via the official roads networks or secondary roads, so called “ramais”. As a result of that we excluded Porto Walter, Marechal Thaumaturgo and Jordão from the sampling frame that includes ranchers from 19 municipalities in total. Excluding those municipalities from our sample frame might lead to a slight underrepresentation of very inaccessible, remote areas in our database. However, we think that these exemptions do not influence significantly the representativeness of our results, since most of the territory in those municipalities are set as conservation units consisting of Indian and Extractive reserves (Acre 2010). Additionally, the overall agricultural activity is very low in those areas and some remote farms were also visited in the municipalities that were included into the sampling process.

Our sampling frame is a list of the Acrean Institute for Animal Defense (IDAF) where a registration of animals is obligatory for ranchers. This list contained over 18,000 names, herd-sizes and eventually contact information of almost all cattle producers in the state. Our stratification criteria is herdsize, which we used as an indicator for the capitalization degree amongst participants. Herdsize was found to be strongly correlated to other socio-economic variables such as land possessions and income. The utilized stratification criteria allowed us to limit the over-representation of small-scale farming in our sample and gather a more complete picture of the production trends in the research area. Our strata were defined as 1-100, 101-200, 201-300, 301-500, 501-1,000 and >1,000 heads of cattle and 20 observations where sampled within each strata. Descriptive Analysis of the IDAF animal registry shows a right skewed distribution with lots of small family businesses and fewer medium and large enterprises.

All in all we interviewed 121 farms were a simple random sample (SRS) within each stratum produced the distribution of interviews per stratum and municipality that is represented in table 1. Municipalities that are marked in red were not visited because they represent a very low number of ranchers in the sampling frame and were not selected through the SRS. Most interviews were taken out in the so called Vale do Rio Acre region which is in the east of the state and hosts most farmers in Acre.

Table C.1: Overview of producers in each strata per municipality

Entrevistas planejadas (=amostragem/10)																	
	FEIJO	ACRELANDIA	MANOEL URBANO	XAPURI	PORTO-ACRE	PLACIDO DE CASTRO	TARAUACA	RIO BRANCO	EPITACIOLANDIA	BRASILEIA	CAPIXABA	SENA MADUREIRA	BUJARI	CRUZEIRO DO SUL	SENADOR GUIOMARD	MANCIO LIMA	RODRIGUES ALVES	Soma
1-100 cabeças	1	1	1	2	1	1	2	3	1	1	1	2	1	1	1	0	0	20
101-200 cabeças	1	2	0	1	2	2	1	3	1	2	1	2	1	0	2	0	0	21
201-300 cabeças	1	3	0	1	2	2	1	3	1	2	1	1	1	0	2	0	0	21
301-500 cabeças	1	2	0	1	1	2	1	3	1	2	1	1	1	0	2	0	0	19
501-1000	1	2	1	1	1	2	1	3	1	2	1	2	1	0	2	0	0	21
>1000 cabeças	0	1	0	2	1	1	1	5	0	1	1	1	3	0	1	0	0	18
Soma	5	11	2	8	8	10	7	20	5	10	6	9	8	1	10	0	0	120

We did 28 pilot interviews in different regions to adjust how our questions were framed which improved significantly the quality of the obtained information. We trained two local postgraduate students from the agricultural faculty over a period

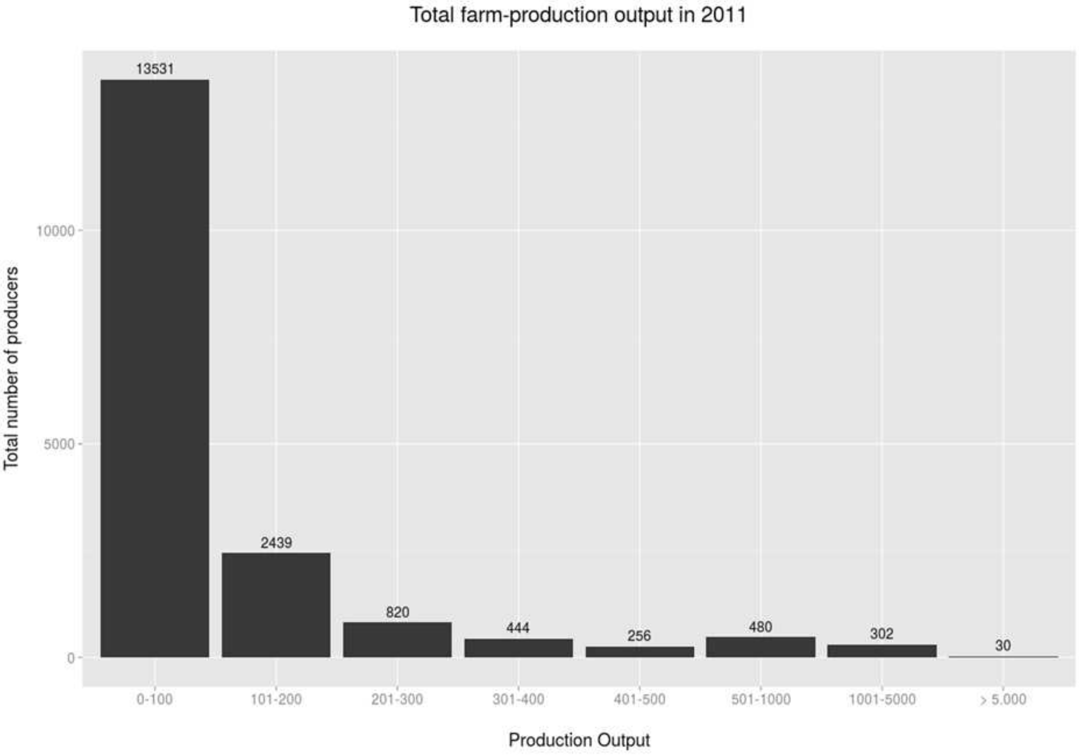


Figure C.1: Histogram of cattle producers by production output (heads of cattle)

of three months and 68 interviews to apply the survey to the remaining respondents from a total of 121 sampled participants. We tried to contact the farmers from our sample with information provided by the IDAF database, with help of other farmers, neighbors and agricultural extension or research agencies such as local EMATERs (mainly for smaller farmers) and EMBRAPA (for medium and larger farmers). If no contact information was available or if farmers did not agree to be interviewed, we took another sample from the original list to obtain new potential participants. Successful contact and response rate in the first sampling round was however high (74%), so we did not have to repeat the sampling process too often. Regarding our participants we could not identify a general characteristic that differentiated people who agreed to participate from those who didn't, which could have ultimately biased our results. However, using extension services as one of the contact-channels might lead to a slight over-representation of farmers with access to technical assistance.

After establishing contact with the farmers we marked a date for an interview with the main decision maker of the farm. The interviews took between 45 minutes and 120 minutes and formal consent was obtained before starting the interview process. Participants were informed about the purpose of the study, the confidentiality of provided information and their right to not respond to any given question. A map with interview locations can be found in Figure C.3.

To obtain spatial information for our study we contacted the State Secretary of the Environment (SEMA) and obtained data from the Rural Environmental Registry of private properties (CAR). At that time around 90% of all rural properties had been registered according to SEMA staff. We than tried to find our respondents in the registry using the GPS position of the interview as well as the full name of the respondent. Altogether, we were able to identify 105 out of 121 respondents in the

CAR. One difficulty was to overcome the mismatch between reported farm-areas and the areas calculated from the polygons in the CAR. Those mismatches can have different reasons. First, the farm-area might be inaccurate, either as reported by the respondent or as digitized in CAR. In this case, mismatches tend to be small and can be therefore ignored. Second, and more importantly, farmers tend to have multiple registries in the CAR that correspond to the total reported farm area. Those registries can be either in the same name, which facilitates the identification, or in some cases areas can be registered in the name of a family member or friend which poses a difficulty. We compared possible farm-area candidates to eliminate those mismatches based on the family name of the respondent and registries with the same name in the immediate neighborhood in the CAR. By doing that we were able to create a comprehensive data-set encompassing all areas from our survey without considerable mismatches.

Based on this data-set we were also able to create different variables from a set of external data sources that characterize farm accessibility, farm suitability for cattle ranching, law-enforcement by means of fines for illegal deforestation as well as statistics on accumulated and recent deforestation for all observations with a valid CAR registry.

The reference-year for recall questions is 2005, which was referred to as the year of the mayor drought, which took place at that moment in time. Locking the base-year to an event that was easily remembered by all participants facilitated the recall of information for participants. Although recall questions may suffer from bias, they are essential for our study, where no baseline data was available.

Information on how participants perceived environmental governance was obtained at the end of the interview in a semi-standardized manner. They help us to explain how policies impacted the production systems in general. Putting those questions at the end of the questionnaire and framing them as mostly open questions facilitated us to gain participants trust on that very sensitive topic. The survey-data was later combined with the database of the Rural Environmental Registry of Private Properties (CAR) which allowed us to obtain spatial information on the farm-level from external data sources.

C.2 External Data Sources and Database Creation

Accessibility

Technical details on the creation of this data can be found at Schielein and Börner 2018. We argue that accessibility, which translates directly into transportation costs, has a considerably larger influence on input and output prices and can be more accurately measured than annual market price developments. Market prices are difficult to assess in our research area because of a general lack of local price information. This is especially true for price information about the variety of ranching outputs (small to medium animals, lean and fattened cattle, milk and other cattle products) as well as the differing inputs, machinery and equipment requirements for successful pasture restoration across farms. Furthermore, during our data collection, a large quantity of ranchers reported to experience limited annual price fluctuations since changes for ranching outputs are not thoroughly passed through the market chain by middle men who acquire animals at the farm-gate (71.6% ±7.9 of ranchers in Acre). The same holds true for medium- to large-sized neighboring farms who acquire animals from their smaller-scale counterparts, often in exchange for production inputs (24.2% ±7.1). Low data resolution, input and output differentiation as well as imperfect market conditions and information asymmetry could hence result in a bias if using conventional price information such as data on annual beef prices at country-scale

Suitability

Our suitability variable is derived from an expert classification of soil-suitability for cattle ranching that was taken out by EMBRAPA staff based on the methodology of Ramalho-Filho and Beek (1995). The authors classified soil-components in deforested areas in the Brazilian Amazon except areas inside strictly protected territories (Fraga da Silva et al. 2016), based on five restricting criteria: Fertility deficiency, water deficiency, oxygen deficiency, susceptibility to erosion and limitations for mechanization. Each criterion was ranked on an ordered scale expressing the degree of limitation that it imposes on the suitability of a given soil-component. Those ranks comprise: zero limitation, light limitation, moderate limitation, strong limitation and very strong limitation and intermediate scores are given in between. The conversion to suitability classes is then based on the minimum-law of Liebig were the most severe limitation of one criterion defines the overall suitability of the soil-component. Limitations are judged depending on the possibility to overcome deficiencies with moderate technological improvements (Port: nível de manejo B see Ramalho-Filho and Beek (1995) for details). The authors used a soil-map from the Brazilian Institute for Geography and Statistics (IBGE) in the scale of 1:250.000 to spatialize their classifications. We converted the ordered scale from EMBRAPAs original work to a numerical scale assuming equal distances between the suitability classes from 1 (very strong limitations) to 5 (no limitations). If a soil was composed of several soil-components, we created weighted averages based on the inverse vector of one to the total number of soil components. If for example a soil was composed of three components, the first/principal components' suitability score was weighted with 3, the second with 2 and the third with 1 before averaging them out. Finally, we normalized all suitability values with the maximum suitability score of the region. By this we derive a normalized scale between from 0-1 where 1 represents the maximum soil-suitability for pasture systems with moderate technological adaptations. Figure C.2 shows the resulting data as well as the locations of our interviews in Acre.

Farmsize

In order to not neglect the possibility of changes in farmsize over time we included statistics on a farmers' activity on the land-market into our variable creation as well. We asked farmers whether they acquired or sold areas between 2005 and 2014 as well as the respective years of that transaction and calculated from that farmsize before restoration.

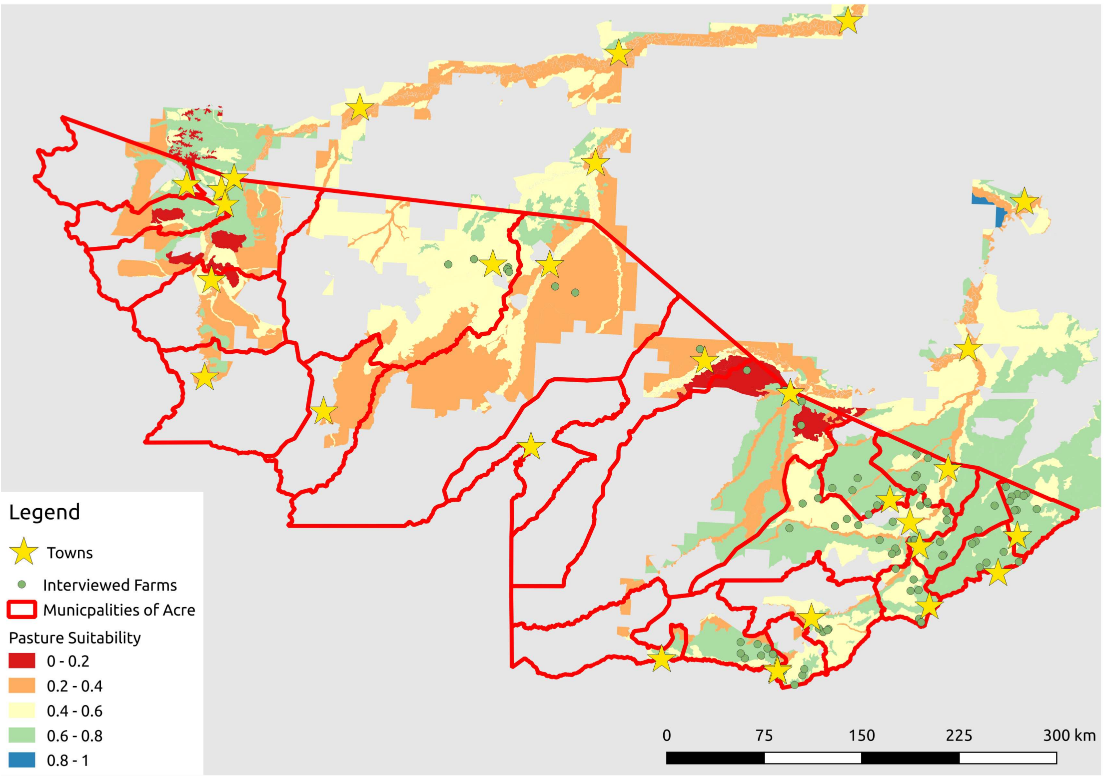


Figure C.2: Map of soil suitability for pasture systems in Acre

Bookkeeping

Since farmers have very different production inputs and outputs the total amount of possible responses also differs. To allow for cross-comparability we focused our measure on five key indicators that ranchers should write down on an annual base: born animals, deceased animals, soled animals, acquired animals and stocking rates. We assume that literacy and bookkeeping, which were measured in 2014, were stable over time and that they are not influenced by restoration efforts itself.

Credits and Technical Assistance

During our interviews we observed that capital resources, such as different types of machinery, are often shared amongst neighbors in exchange for labor or cattle. We therefore argue that credit availability is best measured on a low administrative scale in the farm neighborhood rather than the individual farm-level. Furthermore, we think that credit access, as it was first measured in our survey, might be endogenous to restoration. To measure credit availability in the neighborhood we used data from the Agricultural Census 2006 on the census district scale. Districts are the most disaggregated form to obtain data from the Agricultural Census and due to the random nature of our sample, around 90% of the interviewed farms fall into distinct districts. As with credits we argue that technical assistance is best measured on the neighborhood scale and not the individual level, because knowledge resources are, as in the case of capital assets, shared amongst farming neighbors and might be endogenous to restoration if measured after the decision to restore pastures took place.

Restoration before 2005 and Pasture Improvements

To account for restoration efforts before our study period we asked farmers if they had already restored pastures before 2005. We also asked farmers to report the area where eventual pasture improvements were made by e.g. applying fertilizers or pesticides and whether this was made before or after restoration took place. From this we calculated the amount of improved pastures before a restoration effort started. Furthermore, we calculated the time that had passed since the first year of restoration until 2014 and we asked farmers to report the amount of areas under rotational pasture management in 2005.

C.3 Field Study Area and Interview Locations

The following map shows the location of our interviews as well as travel time to the next large agricultural market, farmsize and the amount of restored pastures as a share of total pasture area. We can observe that larger farms are often in the medium range regarding the share of restored pastures of total farm pasture area. Small farmers, on the contrary, can be found in the low range where most of them did not start any type of restoration efforts yet or in the very high range with most of their pastures already restored. A Co-plot of restoration against farmsize (section C.5) confirms this observation and allows us to plot survey weights. It clearly shows that the majority of small ranchers has not started pasture restoration yet. Figure C.3 also depicts the influence of accessibility on restoration. Generally, a lower travel time comes also in hand with a higher share of restored pasture areas.

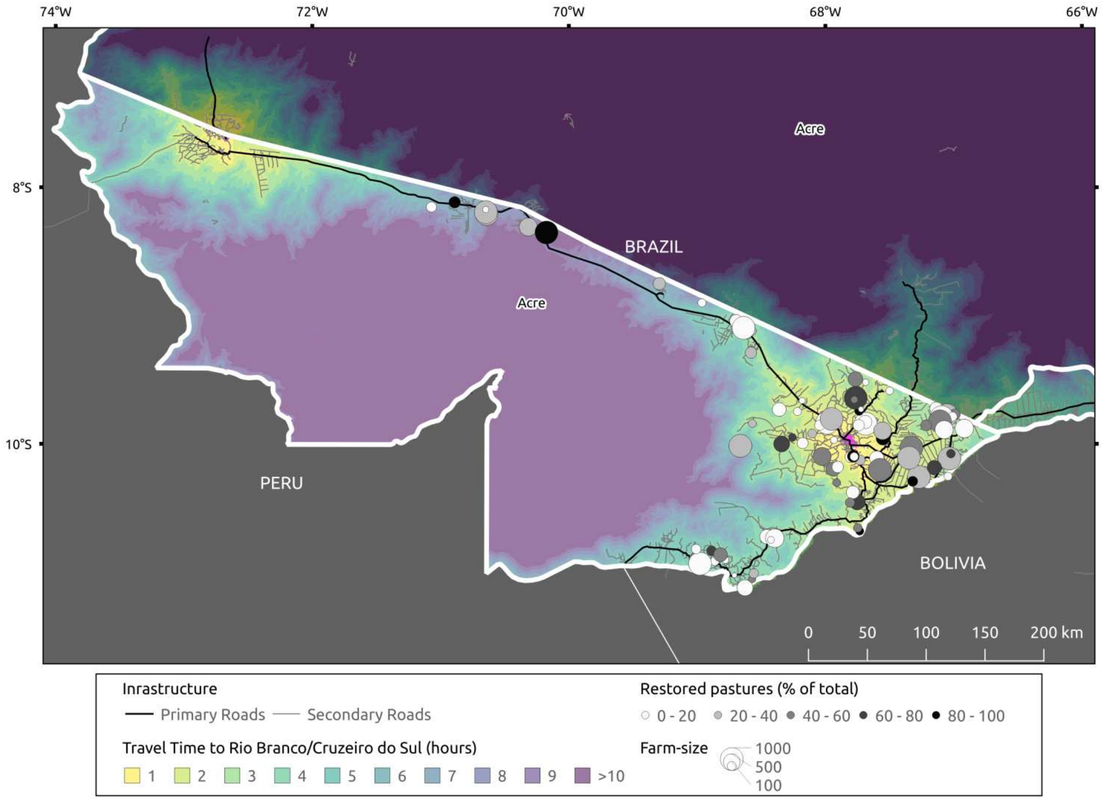


Figure C.3: Overview of the research area and interviewed farms

C.4 Marginal Effects of the Selection Model

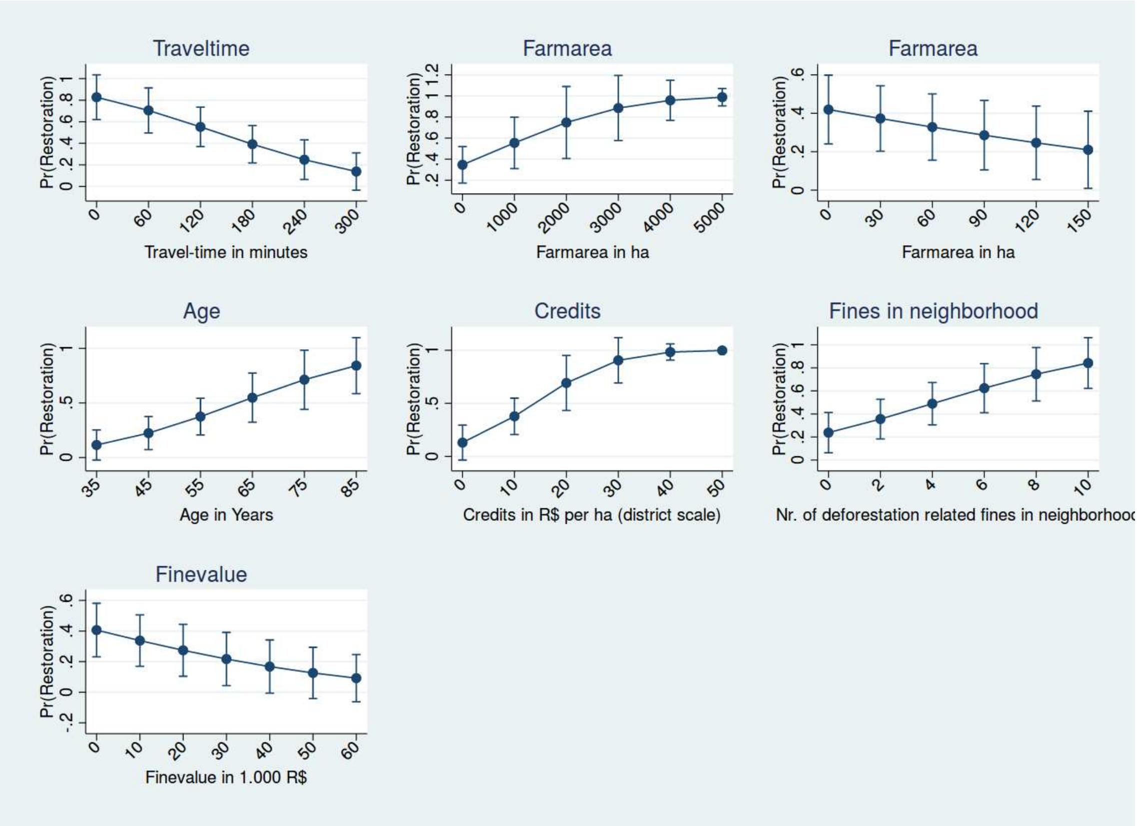


Figure C.4: Margins for selected variables of the first-stage Probit-Model

Note: Adjusted predictions for all variables at 95% Confidence Intervalls. Only significant margins at .05 are shown.

C.5 Pasture Restoration, Farmsize and Deforestation

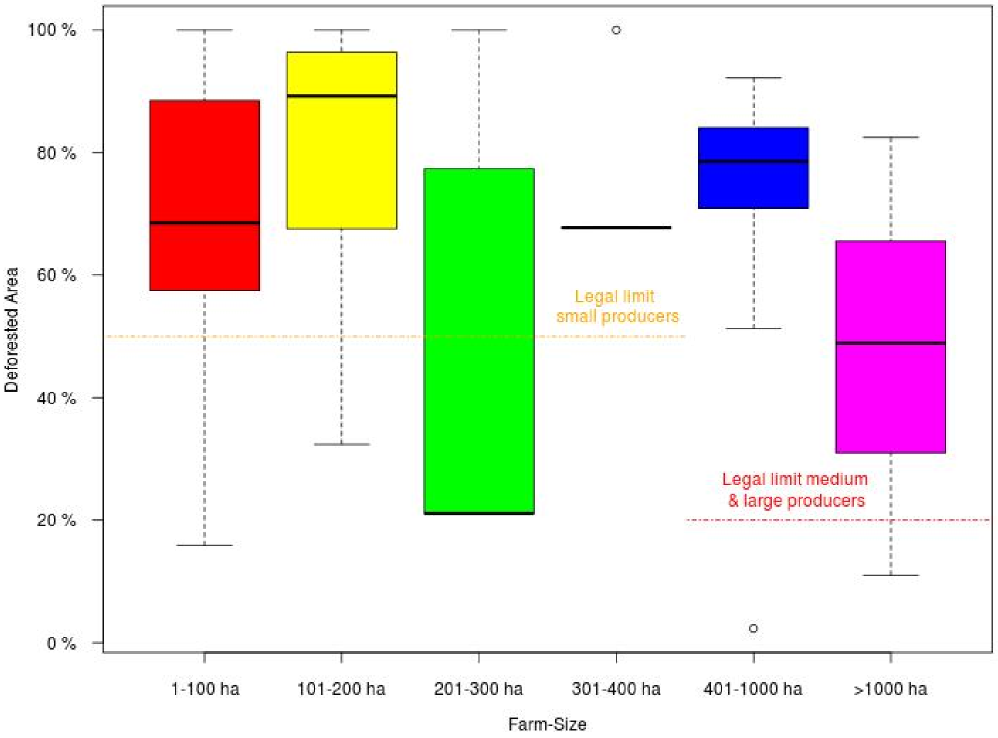


Figure C.5: Deforestation and legal consolidation limits according to the Forest Code and a pardon granted to small farmers in 2012 for deforested areas before 2012

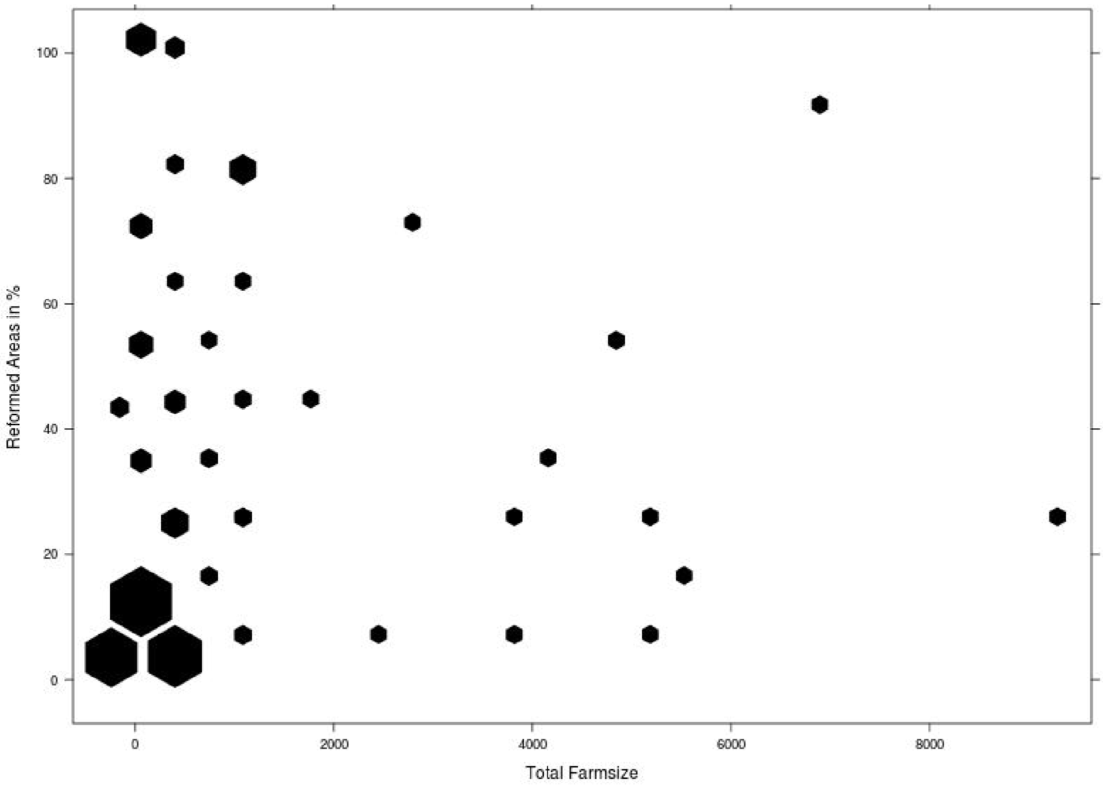


Figure C.6: Coplot of restored areas and farm size

Note: Restoration is referred to here as ”reform”

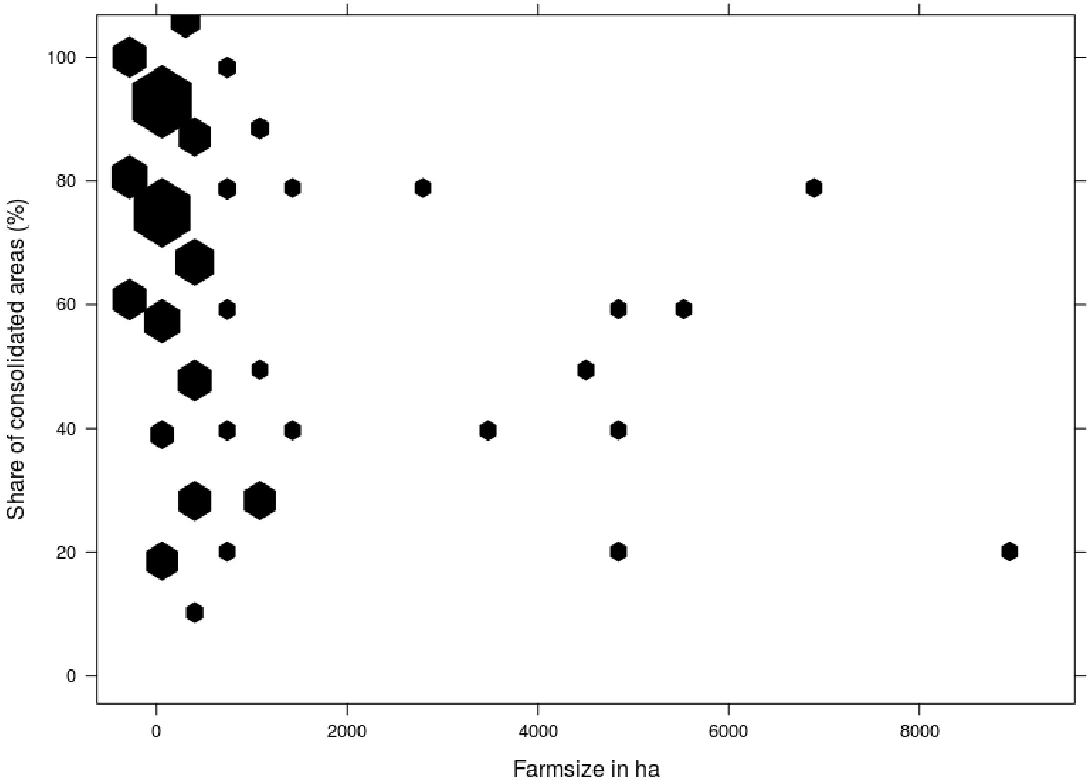


Figure C.7: Coplot of consolidated areas and farm size

APPENDIX D

The AccessibilityMaps Package Tutorial

This tutorial is published at: https://s5joschi.github.io/accessibility/.

A Github repository with the Code of the AccessibilityMaps package is available under MIT liscence at: https://github.com/s5joschi/accessibility

Introduction

This tutorial showcases the creation of *Accessibility Maps* also referred to as *Travel Time Maps* in *R* using the *AccessibilityMaps* package. We will create maps that exhibit travel times from any given location in Burkina Faso to cities with more than 1 Million inhabitants inside the country. We will create two distinct maps, one for the dry-season where unpaved roads function better for car transport and one for the rainy season where travel speed on unpaved roads is dramatically reduced.

The *AccessibilityMaps* package provides mostly wrapper functions to existing packages and FOSS GIS-libraries in and outside of *R*. Most geodata processing is done with *GDAL* which is more efficient and less depended on *RAM* in contrast to the *raster* package in *R*. This allows you to process large raster data-sets with several million raster-cells to create high resolution maps for larger research areas. The *Travel Time Maps* are calculated using the *r.cost* function from *GRASS* (for a documentation see here).

Installation

In order to use the package you need to have *GRASS GIS* and *GDAL* installed. Both are automatically installed if you install Quantum GIS which is also a good software solution for visualizing the results and creating good locking, publishable maps. If you prefer to install *GRASS* and *GDAL* as standalone versions please refer to https://gdal.org/ and https://grass.osgeo.org/download/ to obtain the latest stable versions. It is highly recommended to use the stable versions instead of the latest, but unstable releases. Please make sure that you have *GRASS* and *GDAL* installed before you proceed.

The easiest way to install the *AccessibilityMaps* package is using the *devtools* package in *R* which allows you to download and install everything within *R*. The installation command should look like this:

```r
library("devtools")
install_github("s5joschi/accessibility")
# once installed just activate the package with
library(AccessibilityMaps)
```

The *AccessibilityMaps* package will install automatically a set of R-packages in the background, if they are not already installed on your system Those packages are *raster*, *rgdal*, *gdalUtils*, *rgrass7* and *sp*.

We will proceed by setting up the basic working environment for this tutorial and by downloading the input data. The input-data consist of a vector layer with roads in Burkina Faso from the Open Streetmap project. This data was downloaded from the website of Geofabrik. Furthermore we will use a prototype of land-cover data for Africa in 2016 which is provided in 10 meters resolution from the *ESA copernicus mission* (found here). You do not need to download the data from these websites. Just use the script below to download a copy from this repository.

```
# create and set working directory
dir.create("accessibility_example")
setwd("accessibility_example/")
# download data and unzip it
download.file(
  "https://github.com/s5joschi/accessibility/raw/master/example/input.zip",
  destfile = "input.zip"
)
unzip("input.zip")
# create an output folder for the results
dir.create("output")
```

The Friction Map

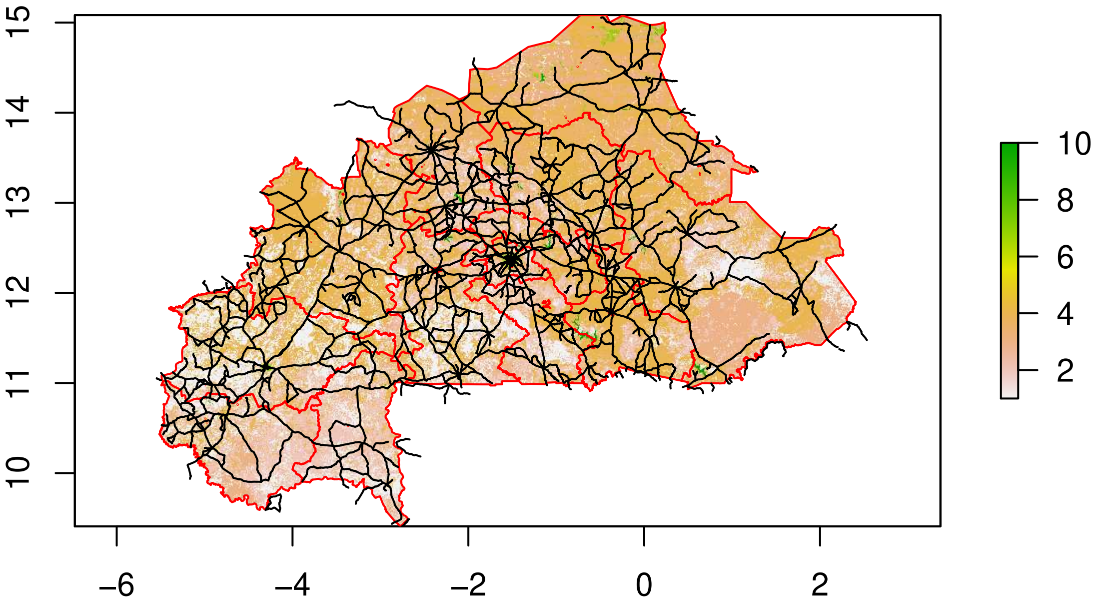
A friction map is a raster layer that shows how much time is required to traverse each raster cell in vertical or horizontal space depending on the underlying surface. Cells that contain roads or navigable rivers on flat surfaces have a very low friction i.e. they are quick to be traversed. In contrast a forest or bush-lands in mountains areas have an extremely high friction. The friction map together with a set of travel destinies are the two main ingredients to create travel time maps.
Let's First load the input-data to create the friction map into *R*:

```
# landcover
r_landcover <-
  raster(
    "input/landuse/esa-copernicus-prototype_0005dergrees.tif"
    )
# digital elevation model (DEM)
r_dem<-
  raster(
    "input/dem/bf_elevation.tif"
    )
# administrational boundaries
spodf_admin<-
  readOGR("input/admin/",
          "gis_osm_places_a_free_1"
          )
# OSM roads
spodf_roads<-
  readOGR(
    "input/roads/",
    "gis_osm_roads_free_1_mainroads"
    )
# OSM places
spodf_sources<-
  readOGR(
```

```
    "input/admin/",
    "gis_osm_places_free_1"
    )
# convert popuplation data from OSM to numeric
spodf_sources@data$population<-
  as.numeric(as.character(spodf_sources@data$population))
```

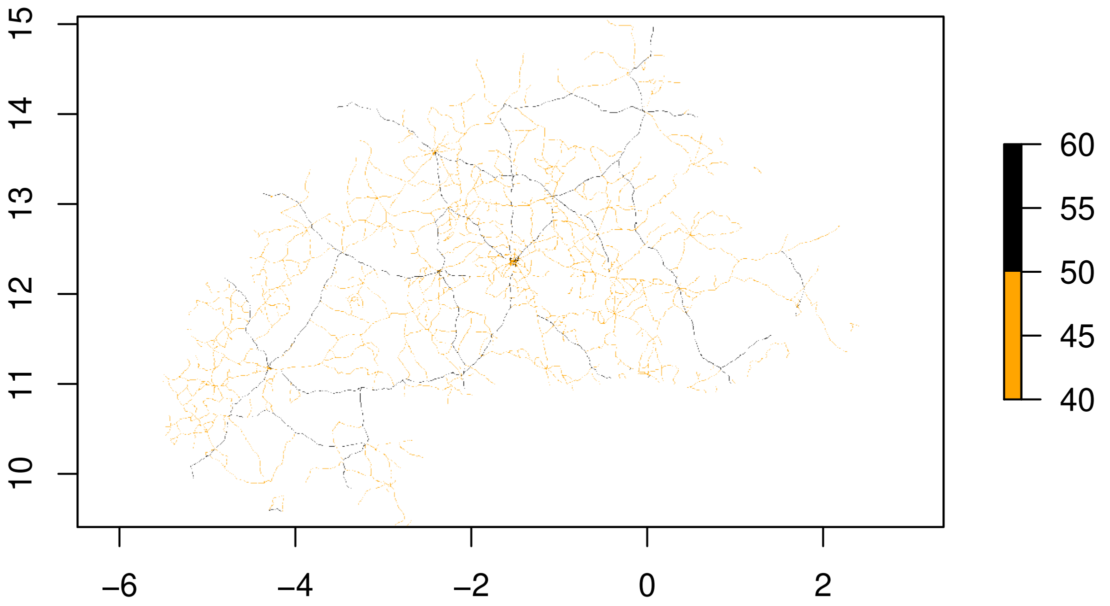
Let's plot some of the input-data to create a map with land-cover classes, the road network and nonadministrative boundaries of Burkina Faso:

```
plot(
  r_landcover
  )
plot(
  spodf_admin,
  border="red",
  add=T)
plot(
  spodf_roads,
  col="black",
  add=T
  )
```



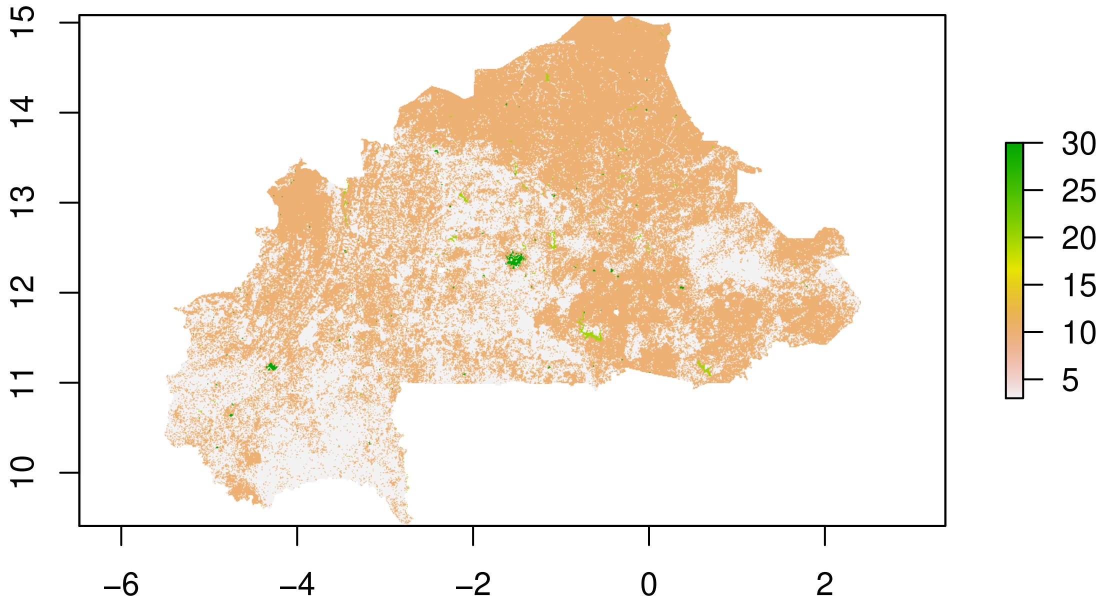
We will start now to process the layers that will be later used to create the friction map. To prepare our road layer for processing we first need to create a column in the attribute table that contains travel speeds in km/h based on the surface type of the road. If it is a road with the category *primary* or *primary_link* (for details see the OSM documentation) we assume it is paved and assign it an average travel speed of 60 km/h. Otherwise we assume that the roads are not paved and assign it a travel speed of 40 km/h in the dry season- and only 10km/h in the rainy season. We use our land-cover map as a base-layer to define the projection system, extent and resolution of the layers to be created. We use the function *acc_vec2fric* to process the vector input-data.

```
# 3.1 roads
# create columns with travelspeeds for dry and wetseason
spodf_roads$s_dry <-
  ifelse(spodf_roads$fclass %in% c("primary", "primary_link"),
         60,
         40)
  spodf_roads$s_wet <-
  ifelse(spodf_roads$fclass %in% c("primary", "primary_link"),
         60,
         10)

# create friction input layers from roads
r_roads_dry <-
  acc_vec2fric(my_input = spodf_roads,
  my_baselayer = r_landcover,
  my_speedfield = "s_dry")

r_roads_wet <-
  acc_vec2fric(my_input = spodf_roads,
  my_baselayer = r_landcover,
  my_speedfield = "s_wet")

# plot the rasterized roads
plot(r_roads_dry,
     col=c("orange","black"))
```



The next step now is to process the land-cover layer. In order to process the raster input data we use the function *acc_ras2fric*. Our original land-cover data has 11 classes from 0 to 10 (for the original classes see the documentation on the link above). Our assumptions on travel speeds on the different land surfaces are listed in the code below. We provide the land-cover input classes as a vector with the sequence 0,1,2,3,4,5,6,7,8,9,10 and another vector containing the corresponding travel speeds in the same order.

```
# reclassification values.
  # no data will be reclassified to 0 km/h,
  # tree cover and shrublands and similar to 3 km/h
  # open landscapes to 10 km/h
  # urban areas to 30 km/h
  # water to 20 km/h (boat transportation)

r_landcover_reclass <-
  acc_ras2fric(
    my_input = r_landcover,
    my_baselayer = r_landcover,
    my_reclass_inputvalues = 0:10,
    my_reclass_outputvalues = c(0, 3, 3, 10, 10, 3, 10, 10, 30, 2, 20)
    )

# plot the output raster
plot(r_landcover_reclass)
```



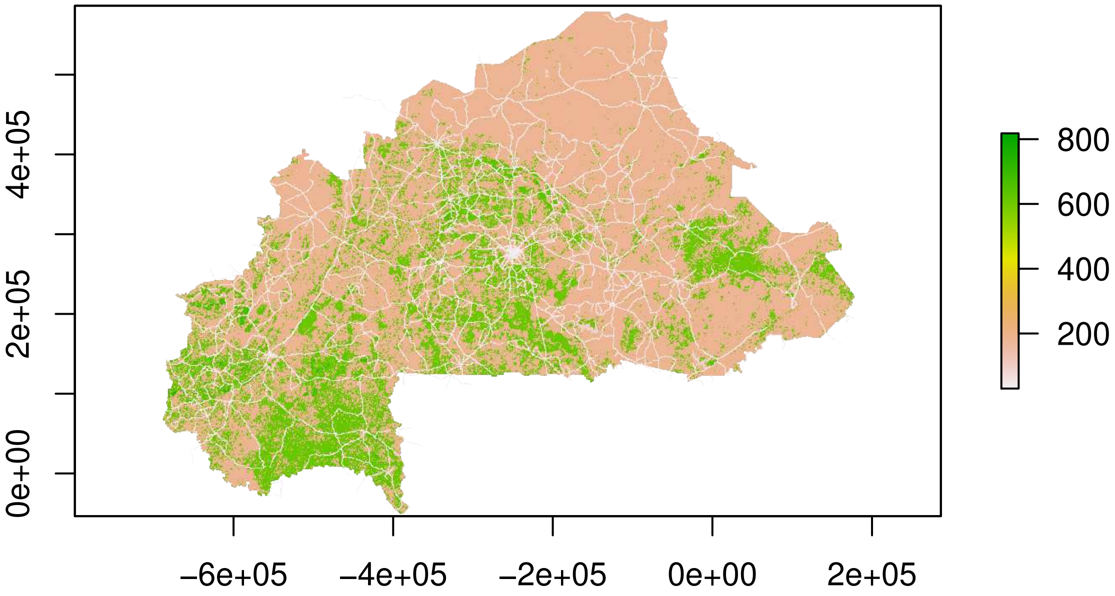
Now we will correct the input layers for the effect that slope exerts on them. Our main assumption is that steep slopes reduce travel speed. In order to do so we first need to convert a *Digital Elevation Model (DEM)* into a map that contains information about the slope measure in radians. We use the function *acc_radians* to create this map assuring that it has the same spatial characteristics as all other layers using the land-cover map as a base-layer. Once we have a radians map we can use the function *acc_slopecorr* to correct the input layers.

```
# Create radiansmap
r_radians <-
  acc_radians(
    my_input = r_dem,
    my_baselayer = r_landcover,
    resampling_method = "near"
```

```
  )

# correct for slope effects
r_roads_dry_corr <-
  acc_slopecorr(my_input = r_roads_dry,
                my_radians = r_radians)

r_roads_wet_corr <-
  acc_slopecorr(my_input = r_roads_wet,
                my_radians = r_radians)

r_landcover_reclass_corr <-
  acc_slopecorr(my_input = r_landcover_reclass,
                my_radians = r_radians)
```

The final step to create the friction map is to use all pre-processed input layers with the function *acc_friction*. We create two distinct friction maps: one for the rainy and one for the dry season. We us an output resolution of 500 meters to facilitate the creation of the travel times maps, which demands quite some computational resources. The processing time will ultimately not only depend on your hardware specification but more importantly on the extent of your research area and the spatial resolution.

```
r_friction_dry <-
  acc_friction(list(r_roads_dry_corr, r_landcover_reclass_corr),
              my_outputresolution = 500)

r_friction_wet <-
  acc_friction(list(r_roads_wet_corr, r_landcover_reclass_corr),
              my_outputresolution = 500)

# plot friction map of dry_season
plot(r_friction_dry)
```



The friction map contains values that define how much time is needed (in seconds) to cross each grid-cell in horizontal or vertical space. Clearly the roads can be identified in white color to be the cells with less friction. The friction map is the main input data for the travel time map together with the travel destinies (sources) which are provided as a vector input layer.

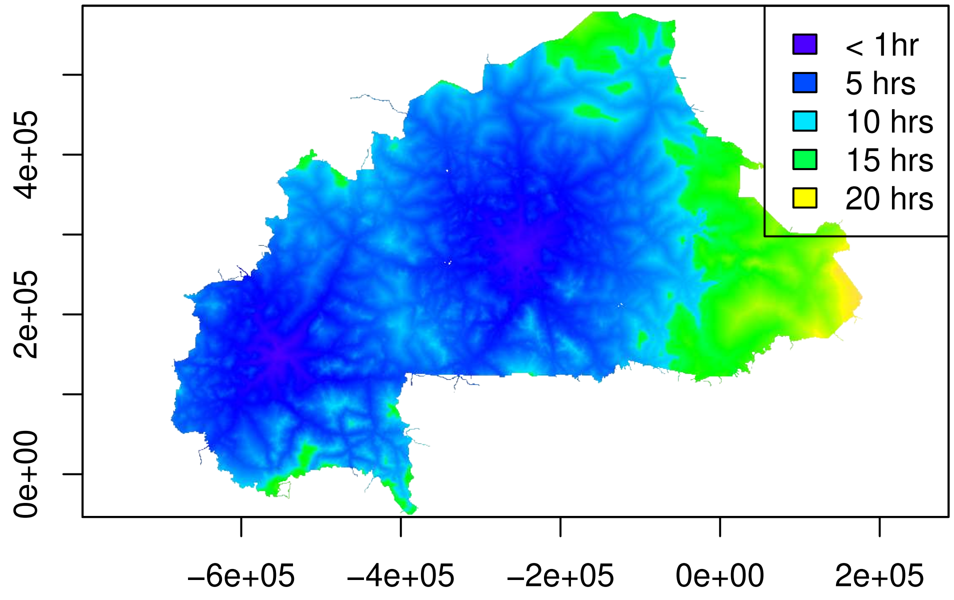
The Travel Time Map

In the last step we will create travel time maps that show the time necessary for any place in Burkina Faso to reach the next city with more than 1 Mio. inhabitants. Once you have the friction maps to your liking you basically can now create travel time maps for any type of source with the code below.

Note, that we use 3 MB of RAM with the *max_ram* parameter. You can adjust this parameter. It is recommended that you do not use much more than around 50-60 % of currently available RAM to not slow down your computer excessively. Also note that you will have to define here the path to the installed *GRASS* binaries. This path varies depending on your local Operation System and the version of GRASS installed. Helpful information about the location of GRASS on your system can be found here: https://grasswiki.osgeo.org/wiki/R_statistics/rgrass7#GRASS_within_R. Another easy way to find the binaries is to use the function *findGRASS* from the the package *link2GI*. Consider installing it if you are unable to find it manually. If you encounter an error regarding *iconv.dll* or other configuration errors for *GRASS* on Windows, this CRAN page might help.

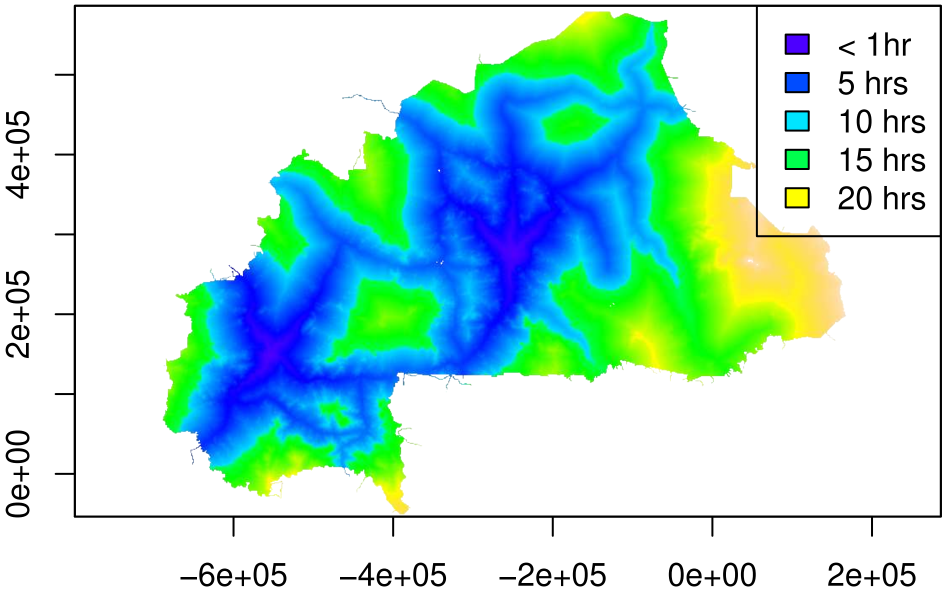
```r
# subset the cities for such that have more than 1 Mio inhabitants.
spodf_sources<-spodf_sources[spodf_sources@data$population >100000, ]

r_accessibility_dry <-
  acc_accessibility(
    my_friction = r_friction_dry,
    my_sources = spodf_sources,
```

```
    knightsmove = T,
    grassbin = "/usr/lib/grass72", # depends on OS
    max_ram = 3000
  )

r_accessibility_wet <-
  acc_accessibility(
    my_friction = r_friction_wet,
    my_sources = spodf_sources,
    knightsmove = T,
    grassbin = "/usr/lib/grass72",
    max_ram = 3000
  )

# convert output (seconds) to minutes
r_accessibility_dry <-
  r_accessibility_dry / 60
r_accessibility_wet <-
  r_accessibility_wet / 60

# plot both maps
plot(r_accessibility_dry,
     breaks=0:1200,
     col = topo.colors(1200),
     legend=F)
legend("topright",
       legend = c("< 1hr","5 hrs","10 hrs","15 hrs","20 hrs"),
       fill = topo.colors(5))
```



```
plot(r_accessibility_wet,
     breaks=0:1200,
     col = topo.colors(1200),
```

```
     legend=F)
legend("topright",
       legend = c("< 1hr","5 hrs","10 hrs","15 hrs","20 hrs"),
       fill = topo.colors(5))
```



```
# export rasters
writeRaster(
  r_accessibility_dry,
  filename = "output/r_accessibility_dry.tif",
  datatype = "INT2U",
  overwrite = T
)

writeRaster(
  r_accessibility_wet,
  filename = "output/r_accessibility_wet.tif",
  datatype = "INT2U",
  overwrite = T
)
```

We can clearly detect differences in wet- and dry-season accessibility for several regions in Burkina Faso which is mainly the result of different road qualities. These differences might have an important impact on agricultural value chains or socio-economic welfare for people living in remote rural areas.

Final Note

Please note that it may make also sense to export the layers created during processing to inspect the results in a proper GIS software. If you use *OSM* data please note, that the quality varies greatly from country to country. Depending on your research objective you might want to always complement it with data from governmental agencies. Also it makes sense to include additional infrastructure layers such as waterways or railroads which we did not do in this example for the sake of simplicity. We hope that you can make use of this package and if you encounter any errors make sure to file a bug report here or ask a question on Stackoverflow.

APPENDIX E

The AccessibilityMaps Package Documentation

Package ‘AccessibilityMaps’

January 21, 2020

Type Package

Title Creation of Accessibility Maps in R

Version 0.3.0

Author Johannes Schielein, Gabriel Frey, Javier miranda

Maintainer Johannes Schielein <`johannes.schielein@uni-bonn.de`>

Depends R (>= 2.15.0), raster, rgdal, gdalUtils, rgrass7, sp

Description The AccessibilityMaps package provides comprehensive wrapper function to create accessibility or so called travel-time maps as well as euclidean distance maps in R

License MIT

Encoding UTF-8

LazyData true

R topics documented:

`acc_accessibility` *Create a Travel Time Map from a Friction Surface and a Vector Source Layer*

Description

This is a wrapper function to use rgrass7 to calculate a travel time/accessibility map using the r.cost function from GRASS GIS.

Usage

```
acc_accessibility(
my_friction,
my_sources,
knightsmove = TRUE,
grassbin,
max_ram = 3000
)
```

Arguments

`my_friction`	Friction map for the calculations. Should be of type raster(raster)
`my_sources`	A layer containing the destinies for the accessibility map. Should be of type SpatialPolygons(sp)
`knightsmove`	Use the knightsmove for accessibility calculation? More accurate but also slower. See r.cost document of GRASS for more details. Default value is TRUE.
`max_ram`	Define the maxiumum amount of RAM to be used for the calculation. Should not exeed your available RAM for this. Generally about 30 to 50 % of total RAM should be okay. Default value is 3000 which is suitable for computers with 8GB of RAM
`grassbin`	Define here the path to the grass binaries. An easy way to find the binaries is to use the "findGRASS" function from the the package "link2GI". Consider installing it if you do not know the GRASS location and are unable to find it manually. Additional information can be found here: https://grasswiki.osgeo.org/wiki/R_statistics/rgrass7#GRASS_within_R. If you encounter an error regarding iconv.dll or other configuration errors for GRASS on Windows, this page might help: https://cran.r-project.org/web/packages/openSTARS/vignettes/Warnings_and_Errors.html

Examples

```
# Please refer to https://s5joschi.github.io/accessibility/ to see this usage example.
# notrun
r_accessibility_dry <-
  acc_accessibility(
    my_friction = r_friction_dry,
    my_sources = spodf_sources,
    knightsmove = T,
    grassbin = "/usr/lib/grass72",
    max_ram = 3000
  )
```

`acc_euclid` *Create an Euclidean Distance Raster Map*

Description

This is a wrapper function to use rgrass7 to calculate an euclidean distance map with the r.grow.distance function. This can be considerably faster for raster maps than calculating euclidian distances from raster centroids.

Usage

```
acc_euclid(
my_sources_raster,
grassbin,
my_metric = "euclidean"
)
```

Arguments

`my_sources_raster`	This should be a layer containing the sources. Should be of type raster(raster). Can be created using the acc_vec2fric function on a vector layer
`my_metric`	Distance metric to be used. Defaults to "euclidean". Can be also squared | maximum | manhattan | geodesic. For details see: https://grass.osgeo.org/grass72/manuals/r.grow.distance.html
`grassbin`	Define here the path to the grass binaries. An easy way to find the binaries is to use the "findGRASS" function from the the package "link2GI". Consider installing it if you do not know the GRASS location and are unable to find it manually. Additional information can be found here: https://grasswiki.osgeo.org/wiki/R_statistics/rgrass7#GRASS_within_R. If you encounter an error regarding iconv.dll or other configuration errors for GRASS on Windows, this page might help: https://cran.r-project.org/web/packages/openSTARS/vignettes/Warnings_and_Errors.html

Examples

```
# Please refer to https://s5joschi.github.io/accessibility/ to see usage examples.
```

`acc_friction` *Create Friction Map from Friction Input Layers*

Description

This function creates a friction map from several friction input layers that contain travelspeeds

Usage

```
acc_friction(
my_friction_layer_list,
my_outputresolution,
getproj = TRUE,
my_proj = NULL,
cropfriction = FALSE,
my_croplayer = NULL
)
```

Arguments

`my_friction_layer_list`	List that contains all friction input layers. Those will be stacked and should therefore be all of class raster(raster) and share the same reference system, resolution as well as extent
`my_outputresolution`	This defines the output resolution for the friction map in meters.
`getproj`	Automatically retrive an appropriate geographic projection system for the region? If set to TRUE a projection system is generated in Lambert Azimutal Equal Area which is appropriate for larger areas.
`my_proj`	If "getproj" is set to FALSE this option enables users to pass a custom geographic projection system for the resarch area in form of a a string (proj4string)
`cropfriction`	should the final map be cropped with another layer? This is usefull to reduce processing time of the accessiblity calculation in the the acc_accessibility function.
`my_croplayer`	if cropfriction is set to TRUE, provide here a vector layer for cropping the friction map. Should be of type SpatialPolygons(sp)

Examples

```
# Please refer to https://s5joschi.github.io/accessibility/ to see this usage example.
# notrun
r_friction_dry <-
  acc_friction(list(r_roads_dry_corr,
  r_landcover_reclass_corr),
  my_outputresolution = 500)
```

`acc_radians` *Create a Radians Map from a DEM for Slopecorrection*

Description

This function creates a slopemap in radians and homogenizes it with the baselayer

Usage

```
acc_radians(
my_input,
my_baselayer
)
```

Arguments

`my_input`	An input layer containing a Digital Elevation model DEM. Should be of type raster(raster)
`my_baselayer`	A baselayer that defines resolution, extent and projection system of the friction map. Should be of class raster(raster). Most commonly a layer containing land-use values. Can be the same layer as the input layer
`resampling_method`	Defines the method used to resample the raster values into lower resolutions. Defaults to maximum. Other options contain ("near" | "bilinear" | "cubic" | "cubicspline" | "lanczos" | "average" | "mode" | "max" | "min" | "med" | "q1" | "q3", see GDAL documentation for more information.

Examples

```
# Please refer to https://s5joschi.github.io/accessibility/ to see this usage example.
# notrun
r_radians <-
  acc_radians(
    my_input = r_dem,
    my_baselayer = r_landcover,
    resampling_method = "near"
  )
```

`acc_ras2fric` *Process Raster Layer to Friction Input Raster*

Description

This function converts raster data into a raster layer containing travel speeds that can be used to create a friction map.

Usage

```
acc_ras2fric(
my_input,
my_baselayer,
resampling_method="max",
my_reclass_inputvalues = NULL,
my_reclass_outputvalues = NULL,
my_datatype = "UInt16"
)
```

Arguments

`my_input`	An input layer for the conversion. Should be of type raster(raster)
`my_baselayer`	A baselayer that defines resolution, extent and projection system of the friction map. Should be of class raster(raster). Most commonly a layer containing landuse values. Can be the same layer as the input layer
`resampling_method`	Defines the method used to resample the raster values into lower resolutions. Defaults to maximum. Other options contain ("near" | "bilinear" | "cubic" | "cubicspline" | "lanczos" | "average" | "mode" | "max" | "min" | "med" | "q1" | "q3", see GDAL documentation for more information.
`my_reclass_inputvalues`	A vector with all unique values from the input raster that should be reclassified into travel speeds
`my_reclass_outputvalues`	A vector with all travelspeeds for reclassification. Should be of the same length and in the corresponding order to reclassify values in my_reclass_inputvalues
`my_datatype`	GeoTiff datatype for temporary storage of data. Defaults to UInt16 storing integer values. For details see GDAL documentation.

Examples

```
# Please refer to https://s5joschi.github.io/accessibility/ to see this usage example.
# notrun
r_landcover_reclass <-
  acc_ras2fric(
    my_input = r_landcover,
    my_baselayer = r_landcover,
    my_reclass_inputvalues = 0:10,
    my_reclass_outputvalues = c(0, 3, 3, 10, 10, 3, 10, 10, 30, 2, 20)
    )
```

`acc_slopecorr` *Slope Correction for Friction Maps*

Description

This function corrects an input raster containing travel speed for slope effects. The conversion is done with the function my_input * (exp(-3 * tan(my_radians)))

Usage

```
acc_slopecorr(
my_input,
my_radians
)
```

Arguments

`my_input`	An iput layer containing travelspeeds for slope correction. Should be of type raster(raster)
`my_radians`	An input layer containing slope measured in radians for correction. Should be of type raster(raster) and share same extent, resolution and reference system as the travelspeed input layer. Can be created with the acc_radians function
`correctionfactor`	This is the correction factor which defines the strength of slope correction. Default value is 3. see...

Examples

```
# Please refer to https://s5joschi.github.io/accessibility/ to see this usage example.
# notrun
r_roads_dry_corr <-
  acc_slopecorr(my_input = r_roads_dry,
                my_radians = r_radians)
```

`acc_vec2fric` *Process Vector Layer to Friction Input Raster*

Description

This function converts vector data into a raster layer containing travel speeds that can be used to create a friction map.

Usage

```
acc_vec2fric(
my_input,
my_baselayer,
my_speed = NULL,
my_speedfield = NULL,
my_datatype = "UInt16"
)
```

Arguments

`my_input`	An input layer for the conversion. Should be of type SpatialPolygons(sp)
`my_baselayer`	A baselayer that defines resolution, extent and projection system of the friction map. Should be of class raster(raster). Most commonly a layer containing land-use values.
`my_speed`	Travel speed in km/h that applies to all polygons in this layer. If the travelspeed of polygons differs depending on a certain feature (e.g. paved/unpaved roads) use my_speedfield instead
`my_speedfield`	Column name of the input layer that contains travel speeds for the different polygons in km/h
`my_datatype`	GeoTiff datatype for temporary storage of data. Defaults to UInt16 storing integer values. For details see GDAL documentation.

Examples

```
# Please refer to https://s5joschi.github.io/accessibility/ to see this usage example.
# notrun
r_roads_dry <-
  acc_vec2fric(my_input = spodf_roads,
  my_baselayer = r_landcover,
  my_speedfield = "s_dry")
```

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