Modeling the Interplay between Land Use, Bioenergy Production and Climate Change

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I dedicate this piece of work

entirely to my wife Lucia

who is heroically walking

this long and winding road

by my side.

Summary

Land use has become a force of global importance, considering that 34% of the Earth's ice-free surface was covered by croplands or pastures in 2000. The expected increase in global human population together with eminent climate change and associated search for energy sources other than fossil fuels can, through land-use and land-cover changes (LUCC), increase the pressure on nature's resources, further degrade ecosystem services, and disrupt other planetary systems of key importance to humanity. This thesis presents four modeling studies on the interplay between LUCC, increased production of biofuels and climate change in four selected world regions.

In the first study case two new crop types (sugarcane and jatropha) are parameterized in the LPJ for managed Lands dynamic global vegetation model for calculation of their potential productivity. Country-wide spatial variation in the yields of sugarcane and jatropha incurs into substantially different land requirements to meet the biofuel production targets for 2015 in Brazil and India, depending on the location of plantations. Particularly the average land requirements for jatropha in India are considerably higher than previously estimated. These findings indicate that crop zoning is important to avoid excessive LUCC.

In the second study case the LandSHIFT model of land-use and land-cover changes is combined with life cycle assessments to investigate the occurrence and extent of biofuel-driven indirect land-use changes (ILUC) in Brazil by 2020. The results show that Brazilian biofuels can indeed cause considerable ILUC, especially by pushing the rangeland frontier into the Amazonian forests. The carbon debt caused by such ILUC would result in no carbon savings (from using plant-based ethanol and biodiesel instead of fossil fuels) before 44 years for sugarcane ethanol and 246 years for soybean biodiesel. The intensification of livestock grazing could avoid such ILUC. We argue that such an intensification of livestock should be supported by the Brazilian biofuel sector, based on the sector's own interest in minimizing carbon emissions.

In the third study there is the development of a new method for crop allocation in LandSHIFT, as influenced by the occurrence and capacity of specific infrastructure units. The method is exemplarily applied in a first assessment of the potential availability of land for biogas production in Germany. The results indicate that Germany has enough land to fulfill virtually all (90 to 98%) its current biogas plant capacity with only cultivated feedstocks. Biogas plants located in South and Southwestern (North and Northeastern) Germany might face more (less) difficulties to fulfill their capacities with cultivated feedstocks, considering that feedstock transport distance to plants is a crucial issue for biogas production.

In the fourth study an adapted version of LandSHIFT is used to assess the impacts of contrasting scenarios of climate change and conservation targets on land use in the Brazilian Amazon. Model results show that severe climate change in some regions by 2050 can shift the deforestation frontier to areas that would experience low levels of human intervention under mild climate change (such as the western Amazon forests or parts of the Cerrado savannas). Halting deforestation of the Amazon and of the Brazilian Cerrado would require either a reduction in the production of meat or an intensification of livestock grazing in the region. Such findings point out the need for an integrated/multicisciplinary plan for adaptation to climate change in the Amazon.

The overall conclusions of this thesis are that (*i*) biofuels must be analyzed and planned carefully in order to effectively reduce carbon emissions; (*ii*) climate change can have considerable impacts on the location and extent of LUCC; and (*iii*) intensification of grazing livestock represents a promising venue for minimizing the impacts of future land-use and land-cover changes in Brazil.

Zusammenfassung

Landnutzung ist zu einer Einflussgröße von globaler Bedeutung geworden, betrachtet man die Tatsache, dass im Jahr 2000 bereits 34% der eisfreien Oberfläche unseres Globus durch Acker- und Weideland bedeckt waren. Die erwartete Zunahme der Weltbevölkerung, nebst dem spürbaren Klimawandel und der damit einhergehenden Suche nach alternativen Energiequellen, um fossile Brennstoffen zu substituieren, können durch resultierende Landnutzungs- und Landbedeckungsänderungen (LUCC) den Druck auf die Ressourcen der Natur erhöhen, ferner Ökosystemdienstleistungen abbauen und andere Umweltsysteme mit Schlüsselfunktionen für die Menschheit zerstören. Diese Doktorarbeit legt vier Modellierungsstudien über das Wechselspiel zwischen LUCC, der zunehmenden Produktion von Biotreibstoffen und des Klimawandels für vier definierte Regionen dieser Welt vor.

In der ersten Studie werden zwei neue Feldfrüchte (Zuckerrohr und Jatropha) für das dynamische und global operierende Vegetationsmodell „LPJ for Managed Land" parametrisiert, um die potentiellen Erträge dieser zwei Feldfrüchte zu bestimmen. Für Indien und Brasilien treten jeweils innerhalb des Landes starke Schwankungen der Erträge von Zuckerrohr und Jatropha auf. Diese räumlichen Ertragsschwankungen führen, je nach Standort des Anbaugebiets, zu unterschiedlichen Flächenverbräuchen unter Berücksichtigung der Produktionsziele von Biokraftstoffen für 2015 in Indien und Brasilien. Insbesondere ist der Bedarf an Anbaufläche für Jatropha in Indien höher als vorher geschätzt. Die Forschungsergebnisse weisen darauf hin, dass eine Einschränkung des Biokraftstoffanbaus notwendig ist, um exzessive LUCC zu vermeiden.

Die zweite Studie untersucht anhand der Verknüpfung des Landnutzungsmodells LandSHIFT und einer Ökobilanzierung das Ausmaß und das räumliche Auftreten von indirekten Landnutzungsänderungen (ILUC) für Brasilien im Jahr 2020. Es wird gezeigt, dass Biotreibstoffe tatsächlich beträchtliche ILUC verursachen können, v.a. durch das Eindringen von Viehweideland in den amazonischen Regenwald. Die Kohlenstoffschuld, die durch ILUC verursacht wird (durch die Substitution von fossilen Energieträgern durch Bioethanol und -diesel), würde erst nach 44 Jahren durch Ethanol aus Zuckerrohr und nach 246 Jahren durch Biodiesel aus Jatropha kompensiert werden. Eine Intensivierung der Viehweidewirtschaft könnte ILUC unterbinden. Wir argumentieren, dass die Erhöhung der Viehbestandsdichte durch den brasilianischen Biotreibstoffsektor unterstützt werden sollte, um auch im eigenen Interesse die Kohlenstoffemissionen zu reduzieren.

In der dritten Studie wurde eine neue Methode auf Basis von Infrastrukturinformationen für die Allokation von Feldfrüchten in LandSHIFT entwickelt. Diese Methode wird exemplarisch für Deutschland angewandt, um die potentielle Verfügbarkeit von Agrarland für die Biogasproduktion abzuschätzen. Die erzielten Ergebnisse machen deutlich, dass in Deutschland ausreichend Land zur Verfügung steht, um nahezu alle (90-98%) Kapazitäten von Biogasanlagen mit angebauten Rohstoffen zu versorgen. Biogasanlagen in Süd- und Südwestdeutschland (Nord- und Nordostdeutschland) könnten größeren (geringeren) Schwierigkeiten begegnen ihre Kapazitäten mit nachwachsenden Rohstoffen zu erfüllen, in Anbetracht der Tatsache, dass der Transport ein kritischer Kernpunkt für die Biogasproduktion ist.

Die vierte Studie nutzt eine adaptierte Version von LandSHIFT mit dem Ziel den Einfluss verschiedener Klimaszenarien und Naturschutzmassnahmen auf die Landnutzung im brasilianischen Amazonas zu bemessen. Die Modellergebnisse belegen, dass bis zum Jahr 2050 durch schwerwiegende Klimaveränderungen die Entwaldungssgrenze tiefer in den Amazonas eindringen kann als unter milderen Klimaveränderungen (z.B. im westlichen Amazonas und in Teilen der Cerrado-Savannen). Der Versuch die Entwaldung im Amazonas und den Cerrado-Savannen aufzuhalten, würde entweder eine Verminderung der Fleischproduktion oder die Aufstockung der Viehbestandsdichte voraussetzen. Diese Ergebnisse machen darauf

aufmerksam, dass eine Notwendigkeit für einen integrierten/multidisziplinaren Plan besteht, um Klimaanpassungsstrategien im Amazonas zu entwickeln.

Die allumfassenden Schlussfolgerungen dieser Doktorarbeit zeigen, dass (i) eine sorgfältige Analyse und Planung für den Anbau von Biokraftstoffen erfolgen muss, um den Kohlenstoffausstoß wirksam zu reduzieren; (ii) der Klimawandel einen beträchtlichen Einfluss auf das räumliche Auftreten und das Ausmaß of LUCC haben kann; und (iii) die Intensivierung der Viehweidewirtschaft ein vielversprechenden Ansatz repräsentiert, um die Auswirkungen von zukünftiger Landnutzung und Landnutzungsänderungen in Brasilen zu minimieren.

Contents

CHAPTER 1

General introduction

One of the first acts of the Portuguese after arriving in Brazil in 1500 was to cut down a tree to make a cross for the first mass. “Primeira Missa no Brasil” (1860) by Victor Meirelles, © Museu Nacional de Belas Artes, Rio de Janeiro.

1.1 Land Use Changing the World

The mastering of fire and tools by humans ~400,000 yr BP probably represented the beginning of effective anthropogenic alteration of the Earth's surface through land-use and land-cover[1] changes (hereafter LUCC) [*Ramankutty et al*. 2006; *Bowman et al*. 2009]. Later on, beginning approximately 10,000 yr BP, the domestication of plants and animals led to the development of agriculture in a few small regions of the globe, such as Mesopotamia, China, eastern USA, Sahel, Mesoamerica and the Andes [*Ramankutty et al*. 2006; *Pongratz et al*. 2008] and represented another milestone in the relationship between men and the surrounding landscape. However, by the 16th century both the magnitude and pace of LUCC started to increase rapidly [*Pongratz et al*. 2008; *Ramankutty and Foley* 1999], and by the late 20th century the conversion of forests, grasslands and other natural vegetation into farmlands, waterways and settlements became a force of global importance [*Foley et al*. 2005; *Haberl et al*. 2007], driven - in the global perspective - by the unprecedented increase in human population and supported by the rise of industrial production of fertilizers [*Steffen et al*. 2005; *Tilman* 1998]. But, of course, a myriad of other regional and local factors such as economic opportunities, land tenure, political and cultural factors also influenced (and still influence) these LUCC [*Lambin et al*. 2000; *Geist and Lambin* 2002; *Lambin and Geist* 2006]. Thus, in the year 2000 roughly 34% of the Earth's ice-free surface was covered by croplands (15 x10^6 km^2) or pastures (28 x10^6 km^2), 23.8% of the Earth's potential net primary productivity had been appropriated (reduced) by humans, and nearly 30% of the world's forests and woodlands had been cut down [*Ramankutty et al*. 2008; *Haberl et al*. 2007].

It is undeniable that such an appropriation of the Earth's matter and energy flows helped civilizations to develop, improved the amount, quality and security of food, freshwater, fiber and medical products, provided shelter, and ultimately allowed the increase in human population to 6.8 billion people today [*Foley et al*. 2005; *Ramankutty et al*. 2006]. However, in most cases these gains were not obtained in the most rational manner, i.e., aiming for a continued use of resources throughout time. On the contrary, humans changed their surrounding landscape generally at the expense of ecosystem services (and even planetary systems such as the global climate), which, ultimately, can jeopardize human existence itself [*Steffen et al*. 2005; *Moran* 2006; *Beddoe et al*. 2009]. LUCC can, for example, cause loss of native habitat and pollinators, affect freshwater availability (through irrigation) and quality (through fertilizers misuse), influence regional climate and air quality, facilitate the spread of diseases, etc [reviewed by *Foley et al*. 2005]. In an attempt to define a so-called "safe operating space" for humanity on Earth, *Rockström et al*. [2009] identified thresholds for nine key planetary systems beyond which humanity might face disastrous consequences. They argue that humanity have already crossed

[1] Shorthand definition of (a) land cover: the land surface biophysical attributes; and (b) land use: the purposes for which humans exploit the land cover and the manner which they do that [see *Lambin et al*. 2006].

three of the nine thresholds: climate change, biodiversity loss, and nitrogen cycle changes. The crossing of two of them - biodiversity loss and nitrogen cycle changes – were majorly caused by land use and LUCC [*Rockström et al*. 2009].

An increase in global human population, which is expected to reach from 7.5 to 14 billion people by 2050 [*Lutz* 1996], together with eminent deleterious climate change and the associated search for energy sources other than fossil fuels [*IPCC* 2007; *Smith et al*. 2009] can, through LUCC, increase the pressure on nature's resources, further degrade ecosystem services, and disrupt other planetary systems of key importance to humanity (e.g., freshwater use, phosphorus cycle) [*Steffen et al*. 2005; *Rockström et al*. 2009]. Science, together with other sectors of our society, has a pivotal role in supporting decision-makers and people in general towards more rational trade-offs between nature exploitation and long-term human welfare in the future. In that sense, computer-based modeling combined with descriptive scenario analysis is currently the method used most by science to understand present and future LUCC dynamics, and consequently to support society [*Verburg et al*. 2006; *Alcamo* 2008].

1.2 Modeling Land-Use and Land-Cover Changes

Because of the multidisciplinary aspect that permeates the LUCC subject, LUCC models must also be conceptually multidisciplinary. Therefore, besides considering biophysical aspects of land use, such as potential crop yields and topography, LUCC models also always take into account socio-economical factors, such as population dynamics and commodities prices, in order to better simulate the driving forces of LUCC [*Verburg et al*. 2006]. Nevertheless, LUCC models differ greatly in the way they treat this biophysical and socio-economical information, and also in the type of results they yield, depending mostly on the research question to be pursued through them [reviewed by *Verburg et al*. 2006; and *Heistermann et al*. 2006].

For example, spatial LUCC models aim at the spatially explicit representation of LUCC at some level of detail, such as a district or a continent (e.g., the LandSHIFT model used in this thesis [*Schaldach and Koch* 2009]). Non-spatial LUCC models focus on the rate and magnitude of LUCC, without tackling its spatial distribution [e.g., *Stéphene and Lambin* 2001]. In agent-based LUCC models the analysis is done in a small-scale unit such as a farm or a plot, and this unit of analysis coincides with the level of decision-making [e.g., *Parker et al*. 2003]. On the other hand, in pixel-based models the unit of analysis is the pixel, which is generally embedded in a raster of several units to millions of pixels [e.g., *Schaldach and Koch* 2009]. LUCC models can still be dynamic or static. In the former more importance is given to the temporal aspect, feedbacks and path dependencies (e.g., *Koch et al*. [2008]), whereas in static models the assumptions or driving forces leading to LUCC remain constant through time [e.g., *Overmars and Verburg* 2005].

By combining only the characteristics cited above we could have more than half a dozen different types of LUCC models (e.g., a spatial agent-based dynamic model, or a spatial pixel-based static model), which illustrates the reason behind the high diversity of LUCC modeling approaches developed in the last couple of decades [*Verburg et al*. 2006; *Heistermann et al*. 2006].

One could also classify a LUCC model by the dominant land-use change processes it addresses, as exemplified by models developed specifically to simulate deforestation [*Soares-Filho et al*. 2006], urbanization [reviewed by *Miller et al*. 1999], or agriculture intensification [reviewed by *Lambin et al*. 2000]. Some LUCC models consider the interplay between all these land-use processes at once [e.g., *Schaldach and Koch* 2009; *Verburg et al*. 1999; reviewed by *Schaldach and Priess* 2008]. And others consider these LUCC processes in a more simplified form, but also simulate other components of the Earth System (climate, natural vegetation, etc), as is the case of integrated models [e.g., *Alcamo et al*. 1998].

Finally, the use of computer-based LUCC models ensue major beneficial outcomes, among which are:

- The structured characteristic of LUCC models facilitates our understanding of LUCC temporal and spatial dynamics.
- Model validation requires collection of "reality" data, which *per se* increases our knowledge on LUCC in present and past times.
- It allows us to project future possible LUCC trajectories and patterns.

The results of the modeling studies presented in this thesis touch all of these three major outcomes in selected regions of the world.

1.3 Overview and Major Objective

Four study cases using computer-based modeling are presented in this thesis:

i. Modeling the land requirements and potential productivity of biofuel crops in Brazil and India (**chapter 2**).

Main question addressed:

Would the spatial variation in the potential yields of sugarcane and jatropha lead to considerably different land requirements to fulfill the biofuel production targets planned by the Brazilian and Indian governments in 2015?

ii. Indirect land-use changes can overcome carbon savings from biofuels in Brazil (**chapter 3**).

Main questions addressed:

What would be the location and extension of the indirect LUCC generated by the fulfillment of Brazil's biofuel production targets for 2020? Would the carbon emissions from such indirect LUCC impair the carbon savings from the use of these biofuels instead of fossil fuels?

iii. Implementing a new land use allocation mode in LandSHIFT: spatial analysis of biogas crop production in Germany (**chapter 4**).

Main questions addressed:

How can the location and capacity of specific infrastructure units (such as biogas plants) be integrated in the LandSHIFT model?

iv. Impacts of climate change and the end of deforestation on land use in the Brazilian Amazon (**chapter 5**).

Main questions addressed:

What are the impacts of different climate change scenarios on LUCC in the Amazon? What are the impacts of halting deforestation of the Amazon on the production of food and land-use intensity?

The common objective underlying all these four study cases is to identify the impacts and adaptation or mitigation strategies related to the LUCC problematic. The four studies are independent, in the sense that each one stands on its own, with its own scientific questions and own conclusions. Nevertheless they are interrelated, since dealing with the same topic (e.g., increased production of biofuels in chapters 2-4), or looking at the same region (e.g., Brazil in chapters 2, 3 and 5). The major topics investigated in this thesis are then: modeling of LUCC and crop/pasture productivity associated with increased production of biofuels and climate change. Livestock, scenario analysis, and policy analysis are other related topics dealt throughout the thesis.

In chapter 2 the main modeling tool used is the Lund-Potsdam-Jena for managed Lands (LPJmL) dynamic global vegetation model [*Bondeau et al*. 2007], whereas in subsequent chapters (3-5) the main modeling tool used is the LandSHIFT (Land Simulation to Harmonize and Integrate Freshwater availability and the Terrestrial environment) model of LUCC [*Schaldach and Koch* 2009]. The studies presented here are comprised mostly of model application, considering that the most substantial developments on the LandSHIFT and LPJmL versions used here were done previously in other studies [*Schaldach and Koch* 2009 (LandSHIFT); *Sitch et al*. 2003; *Gerten et al*. 2004; *Bondeau et al*. 2007 (LPJmL)].

However, in this thesis there has also been some complementary model development (for example the parameterization of two new crop types in LPJmL or the new crop allocation mode for LandSHIFT), as well as intensive gathering of data that served either as model inputs, or for model evaluation or calibration.

At the moment this thesis was finished two of the four studies (chapters 2 and 3) have already been published in peer-reviewed journals. Therefore they are reproduced here mostly in the same way they appear in their published form. For that reason there is some repetition in the description of the modeling tools used. Nevertheless, there are some important methodological differences between the four studies which justify describing the modeling approaches separately in each chapter.

CHAPTER 2

Modeling the land requirements and potential productivity of biofuel crops in Brazil and India

Summary

The governments of Brazil and India are planning a large expansion of bioethanol and biodiesel production in the next decade. Considering that limitation of suitable land and/or competition with other land uses might occur in both countries, assessments of potential crop productivity can contribute to an improved planning of land requirements for biofuels under high productivity or marginal conditions. In this paper we model the potential productivity of sugarcane and jatropha in both countries. Land requirements for such expansions are calculated according to policy scenarios based on government targets for biofuel production in 2015. Spatial variations in the potential productivity lead to rather different land requirements, depending on where plantations are located. If jatropha is not irrigated, land requirements to fulfill the Indian government plans in 2015 would be of 410,000 to 95,000 km^2 if grown in low or high productivity areas respectively (mean of 212,000 km^2). In Brazil land requirements, are of 18,000-89,000 km^2 (mean of 29,000 km^2), what makes jatropha a promising substitute to soybean biodiesel. Although future demand for sugarcane ethanol in Brazil is approximately ten times larger than in India, land requirements are comparable in both countries due to large differences in ethanol production systems. In Brazil this requirement ranges from 25,000 to 211,000 km^2 (mean of 33,000 km^2) and in India from 7,000 to 161,000 km^2 (mean 17,000 km^2). Irrigation could reduce the land requirements by 63% and 41% (24% and 15%) in India (Brazil) for jatropha and sugarcane respectively.

2.1 Introduction

Biofuels are increasingly seen as an alternative to petroleum derivates in order to reduce carbon emissions to the atmosphere, but also due to recent escalating petroleum prices [*EIA* 2007]. The two main biofuel options presently considered are ethanol (from fermentation of carbohydrates) as a substitute to gasoline, and vegetable oils (biodiesel) to replace diesel fuel. Besides the uncertain reduction of emissions their use would represent per se [i.e., not considering the emissions arising from land use/cover change, fertilizers, machinery, processing], great concerns exist about the extension of land necessary to cultivate these crops in order to produce a significant substitution of fossil fuels [*Righelato and Spracklen* 2007; *Scharlemann and Laurance* 2008]. The land use change resulting from increase of biofuels could lead to conversion of food croplands to biofuel croplands, extension of biofuel croplands into (tropical) forests, and increased fertilizer use, with the latter two decreasing the beneficial effect of biofuels replacing fossil fuels with respect to greenhouse-gas (GHG) emissions [*Righelato and Spracklen* 2007; *Zah et al*. 2007; *Crutzen et al*. 2008].

The Brazilian Government, for example, is planning an increase in ethanol production of more than 100% up to 2015 [*MME and EPE* 2007]. Increases of the same magnitude are also expected in the USA, India and the European Union [*EIA* 2007; *Bharadwaj et al. 2007a*; *European Commission* 2006]. It is estimated that a 10% substitution of petrol and diesel fuel would require about 40% and 38% of current cropland area in the United States and Europe respectively [*IEA* 2004]. Similar estimates on land requirements for biofuels are also found for Brazil [*MME and EPE* 2007; *UNICA* 2008] and India [*Planning Commission* 2003]. However, such estimates are highly uncertain, because they are strongly dependent on where these biofuel plantations will be located, especially in large countries with diverse agro-ecological conditions. Spatial variability in natural edaphic and climatic conditions in large countries as Brazil and India might lead to differences in crop productivity from one region to another. Furthermore, any regional planning for biofuel crop expansion or intensification in these countries should take into account the spatial variability of these crops' productivity, enabling planners to optimize energy return per land area.

In this study we assess the potential productivity of *Saccharum officinarum* (hereafter sugarcane) for ethanol and *Jatropha curcas* (hereafter jatropha) for biodiesel in Brazil and India. Both countries are subject to increasing concerns on whether biofuels would cause disappearance of natural habitats or replace food production [*UNICA* 2008; *Schaldach et al*. 2010a]. However, spatially explicit large-scale assessments of these crops' potential productivity are barely used in the debate. For example, is sugarcane as productive in the Amazon region as it is in southeast Brazil? Or, what are the most productive areas to grow jatropha in India? We show that the land area needed to fulfill next decade's demands for biofuels in both countries varies significantly depending on where these crops will be grown. Furthermore, based on the simulated potential productivity we are able to ponder whether existing plantations are located in the most productive areas and where future biofuel plantations should be allocated if one wants to optimize energy return per land area.

2.2 Biofuels in Brazil

Brazil is the largest producer and user of sugarcane, with about 30% of the global production in the last decade [*FAO* 2010]. In 2006/2007 these plantations occupied an area of 62,000-78,000 km^2 (depending on data source [*UNICA* 2008; *IBGE* 2010]), which is equivalent to approximately 12% of Brazil's cultivated land, excluding pastures. Most of the sugarcane fields (60%), as well as most sugarcane mills, transport infrastructures, and consuming markets are located in the state of São Paulo [*IBGE* 2010]. Nearly half of the national yield is used for ethanol production from sugarcane juice (which yields ~85 L of ethanol per Mg of sugarcane in Brazil) [*UNICA* 2008], with 20% of this ethanol being exported, and the remaining amount domestically used as fuel for vehicles. Current law enforces a 20–25% blend of ethanol to gasoline used in vehicles. However, flex-fuel cars, which can run on gasoline or ethanol on any proportion, are becoming increasingly common, and represented 73% of light vehicles sold in 2007 [*ANFAVEA* 2007]. Therefore the Brazilian government [*MME and EPE* 2007] and the sugarcane industry [*UNICA* 2008] are projecting an increase of more than 100% in the production of ethanol by 2015 (Table 2.1), owing to the steady growing transport sector in Brazil, and also to the rising external demand for the cheap Brazilian ethanol [*MME and EPE* 2007; *ANFAVEA* 2007; *Goldemberg* 2007].

In 2008 the Brazilian government launched the National Biodiesel Production and Use Program, which in January 2008 resulted in the implementation of a mandatory 2% blending of biodiesel to fossil diesel, with projection of increase to 5% by 2010 [*Pousa et al*. 2007]. In principle this program aims at stimulating the cultivation of a variety of oil crops for biodiesel production, including castor bean and palm oil, which can be grown in the northeastern and northern parts of the country. However, in practice 80–90% of the national biodiesel production (which was nearly 1 $x10^9$ L in 2006–2007) is being produced from soybean oil, planted in Centre-South States, at a cost that was higher than conventional diesel's in the last years [*MME and EPE* 2007]. Although the government projects an increase of 310% in the biodiesel production up to 2015, it is still unclear which oil crops will be used. It is questionable whether soybean will continue to be the main crop to cover such a large increase or not, due to its large land requirements (which is often related to deforestation in the Amazon), low energy return per area of land, low mitigation of GHG emissions, its competition for food markets, and high production costs of soybean biodiesel [*Scharlemann and Laurance* 2008; *Fairless* 2007; *Hill et al*. 2006]. Amongst the other most cited options is castor bean (*Ricinus communis*, Euphorbiaceae; the same family as jatropha), which can be planted on wastelands and supposedly has the potential to include small scale farmers in the biodiesel production process [*Pousa et al. 2007*], even though it has not been subject to substantial research and development (R&D) yet, and current castor bean biodiesel prices are even higher than soybean biodiesel [*MME and EPE* 2007]. Following India's commitment to jatropha, the Brazilian Agricultural Research Corporation (EMBRAPA), has recently started an

Table 2.1: Government targets for increment in biofuel production by 2015 compared to 2006-2007. Sources: *MME and EPE* [2007]; *UNICA* [2008]; *Planning Commission* [2003]. Energy contents of sugarcane ethanol and jatropha biodiesel are 21.3 MJ L^{-1} and 36.2 MJ L^{-1} respectively [*Bharadwaj et al*. 2007a; *Achten et al.* 2008].

	Brazil			**India**		
	Volume $\times10^9$ **L**	**Energy PJ**	**Production Tg**	**Volume** $\times10^9$ **L**	**Energy PJ**	**Production Tg**
Ethanol	+18.55	+395	+215 (+103%)	+1.37	+32	+82 (+249%)
Biodiesel	+3.10	+112	+11 (+310%)	+13.16	+476	+47 (+502%)

R&D program on jatropha as a promising source for biodiesel production. In this study we assume that all future increment in the Brazilian biodiesel production will come from jatropha.

2.3 Biofuels in India

India is the second largest sugarcane producer with roughly 17% of the world production in the last decade [*FAO* 2010]. Plantations are located mainly in Uttar Pradesh (north India) and Tamil Nadu (south India), and more than 90% of them are irrigated. However, just a minimal fraction (~8%) of the Indian yield is dedicated to ethanol production [*Ministry of Agriculture* 2009], and ethanol in India is almost entirely produced in the molasses route, which yields 11 L of ethanol per Mg of sugarcane in comparison to 75 L of ethanol per Mg of sugarcane when made of cane juice in India [*Bharadwaj et al. 2007a*]. This latter number is different from Brazil's 85 L Mg^{-1} due to different technologies used in India. In 2002 the Indian government established a (quasi) obligatory 5% blending of ethanol to gasoline, which was, in practice, applied just in part of the country due to a decline in sugar production in subsequent years. In November 2006 the 5% blending rate was not just reinforced but had to be adopted by 20 states and all union territories [*Bharadwaj et al. 2007a*]. The government projects that a 10% blend will be adopted until 2012 (Table 2.1) [*Planning Commission* 2003], even though current production is not sufficient to fulfill the internal demand for gasoline blending [*Bharadwaj et al. 2007a*].

India's National Mission on Biodiesel aims to meet 20% of the country's diesel requirement by 2012, namely from biodiesel derived from jatropha [*Planning Commission* 2003; *Bharadwaj et al*. 2007b]. The program's first phase (2003-2007) established the current 5% blending of biodiesel to fossil diesel in the whole country. The second phase (2007-2012) envisages self-sustaining expansion of jatropha plantations on up to 112,000 km^2, and the installation of more transesterification plants (needed for jatropha oil transformation) to meet the 20% target by 2012. Jatropha is at the highest rank on India's biodiesel program mainly owing to its high oil content seeds, low water requirements, fast growth, potential value of jatropha biodiesel by-products and the potential benefits its large-scale application would represent for small scale farmers [*Bharadwaj et al*. 2007b; *Openshaw* 2000]. Nevertheless, the utility of jatropha remains debatable due to the lack of in-depth research, for example, on the selection of more productive varieties and its behavior in different climatic regions of India [*Fairless* 2007; *Dange et al*. 2006]. Although 5% of the country's diesel demand is being somehow covered by biodiesel (some by imported palm oil [*Coley 2007*]), it is rather difficult to determine the actual extension and location of jatropha (or other oil crops like *Pongamia pinnata*) plantations, since no reliable statistics are available.

2.4 Materials and Methods

2.4.1 Simulating the Potential Productivity of Sugarcane and Jatropha

Sugarcane and jatropha are modeled within the well established process-based LPJmL model, using the concept of plant (natural) or crop (agriculture) functional types (PFTs or CFTs) [*Bondeau et al*. 2007]. LPJmL simulates natural vegetation dynamics and agricultural productivity based on physiological processes as photosynthesis, autotrophic respiration, evapotranspiration, effects of soil moisture and drought stress, as well as plant's functional and allometric rules, phenology and growth parameterizations in 0.5° grid cells [*Bondeau et al*. 2007; *Sitch et al*. 2003; *Gerten et al*. 2004]. The CFTs previously implemented in LPJmL did not include sugarcane or jatropha, and these two crop types were parameterized for this study.

The sugarcane CFT was implemented in LPJmL using the same approach as for the other CTFs previously implemented in the model [*Bondeau et al*. 2007]. The main differences lie in its distinct phenological development curve (Figure A.1 of Appendix A) and a much higher harvest index (fraction of aboveground biomass that is harvested) when compared to other LPJmL CFTs. Detailed description of input data, parameters and management practices (irrigation, fertilization, sowing date) applied to sugarcane CFT are presented in Appendix A. Global actual country level sugarcane modeled yields are in good agreement (r = 0.76) with FAO reported data [*FAO* 2010], especially in Brazil and India. Potential production is calculated just by assigning sugarcane plantations to the whole considered area (in this case Brazil and India), a procedure repeated for both rainfed and irrigated sugarcane fields.

J. curcas is a perennial deciduous shrub native from Central America, nowadays widespread throughout the tropics [*Openshaw* 2000; *Achten et al*. 2008]. Its seeds bear high oil content, which has been used for biodiesel production especially in India [*Achten et al*. 2008]. The LPJmL model accounts only for annual crops, which are sown, grown and harvested within a year, after which the plants die and their residual biomass (including roots) is incorporated in the soil [*Bondeau et al.* 2007]. For that reason jatropha was implemented in LPJmL within a natural PFT framework. With that, important parts (e.g., roots and sapwood) of the plant biomass last for many years, as is appropriate for a permanent (i.e., non-annual) crop. We changed some morpho-physiological parameters to adapt the tree-like framework of LPJmL PFTs to the characteristics of a deciduous drought-resistant shrub. Harvested parts are a constant fraction of annual net primary productivity (see Appendix A). Unlike the sugarcane calculation, jatropha simulations are entirely potential, since there is no global data on the location of jatropha plantations, nor country level production statistics. Comparisons of modeled jatropha yields to observed data available for some countries or regions show a good correlation (r = 0.71), especially for observed data on adult jatropha trees. In India most of the data points are in good match (e.g., Karnataka, Orissa, India irrigated), although there are some outliers like West Bengal, where LPJmL overestimates yields. No data on jatropha yields in Brazil was found. The nearest available observed data, which is also in good correlation to LPJmL results, is in Paraguay. Appendix A shows the details of jatropha parameterization and the performance of modeled results against reported data of jatropha yields and biometrics.

For both sugarcane and jatropha, potential yields were averaged for the 1971-2000 climate period, and no future climate change was considered due to the short term (7 years) analysis on land requirements for biofuels, which is the main focus of this study. Fertilizer use in 2015 is here considered to be the same as in the 1990's (for sugarcane), based on data by the International Fertilizer Industry Association [*IFA* 2002]. The way this information translates into increased plant development is described in Appendix A, and also by *Bondeau et al*. 2007. No other management practice than irrigation is considered for jatropha.

2.4.2 Calculation of Land Requirements

In India almost all ethanol comes from molasses, which is locally a more rentable system, yielding, besides sugar for human consumption, 11 L of ethanol per Mg of sugarcane [*Bharadwaj et al*. 2007a]. Here we assume, following Indian government projections [*Planning Commission* 2003], that by 2015 around 40% of the produced ethanol will originate from cane juice (75 L Mg^{-1} in India), and 60% from molasses. Sugarcane stems dry to fresh matter ratio is 0.27 [*Wirsenius* 2000]. Therefore in Brazil (India) the 2015 additional demand on sugarcane production for ethanol is of 215 (82) Tg, which means an increase of 103% (249%) compared to the actual ethanol production (Table 2.1).

As already mentioned, in Brazil most of the biodiesel is currently produced from soybean oil, but here we consider that all future increment in the production will come from jatropha (see section 2.2). For both countries one Mg (fresh matter) of jatropha seeds yields 277.5 L of biodiesel, assuming a 34% seed oil content, 75% extraction efficiency, and 0.94 dry-to-fresh matter ratio of the seeds [*Achten et al*. 2008]. Therefore the 310% (502%) increase in production of biodiesel in Brazil (India) in 2015 would require additional 11 (47) Tg of jatropha seeds (Table 2.1). Since the report by *Planning Commission* [2003] gives demand projections just up to 2012, we assumed the same growing tendency of the 2007-2012 period applies to 2012-2015. This relative increase is comparable to other projections [e.g., *Singh* 2006].

Three pathways were analyzed in order to fulfill the above cited demands: (1) start allocating demand from the grid cells with the lowest productivity to the grid cells with highest productivity in the country; (2) start allocating demand from grid cells with the highest productivity to the grid cells with lowest productivity in the country; (3) fulfill the demand with the country's mean productivity. In all the cases the land requirements are for the year 2015 in relation to 2006-2007 country production of biofuels used solely in the transport sector (i.e., ethanol production for other uses in India is not considered). Only grid cells presenting yields higher than a minimum value (5 Mg ha^{-1} [2] for sugarcane; 0.33 Mg ha^{-1} for jatropha) are considered for calculation of land requirements. Although commercial scale yields for sugarcane are considerably higher than this minimum value, we did not want to exclude lower yields which are typical to small-scale and subsistence agriculture. Moreover, in this study we do not consider competition between land uses and assume that every grid cell (either currently covered by natural vegetation or agriculture) is available for sugarcane or jatropha.

2.5 Results

Figure 2.1 shows sugarcane potential yields in Brazil and India, under rainfed and irrigated conditions for the 1971-2000 climate. In Brazil the most productive areas are located in the southeast, where milder temperatures optimize photosynthesis and thus net primary productivity. Maximum annual yields (hereafter only yields) range from 89 Mg ha^{-1} for rainfed conditions to 103 Mg ha^{-1} when irrigated. Although somewhat 20% less productive, the Amazon region does support a considerable sugarcane productivity. Low yields are found in dry areas, like in Northeast Brazil under rainfed conditions or in cold high altitude areas as in Santa Catarina State. Most productive rainfed sugarcane areas in India are located in the extreme east (e.g. Assam) and southwest states (Karnataka, Madhya Pradesh), with mean yields around 88 Mg ha^{-1}. Yields decrease steadily towards the drier northwest, reaching a mean value of 14 Mg ha^{-1} in Rajasthan. Notwithstanding, potential yields are considerably higher throughout entire India when sugarcane is irrigated. Under irrigated conditions yields of 73 Mg ha^{-1} or more are reached in most of the country, with peaks of 106 Mg ha^{-1} in the Western Ghats. Even so, low yields still occur in elevated desert-like north Kashmir.

The potentially most productive areas for rainfed jatropha in Brazil are located in the Southern states (Figure 2.2). Pronounced seasonality (jatropha is a deciduous plant), though with sufficient precipitation in the rainy season in this region, promotes yields of more than 5 Mg seeds per ha. This value decreases to 4.5 Mg ha^{-1} in the southeast, 2 Mg ha^{-1} in the northeast and 3.5 Mg ha^{-1} in the north. However, when jatropha is irrigated, the high productivity areas increase to most of south and east Brazil, with mean yields higher than 5 Mg ha^{-1}, while outer parts of the Amazon have an increase to 4.5 Mg ha^{-1}. In India there is a more pronounced difference between the potential yields of rainfed and irrigated jatropha. For the rainfed case peaks of up to 5.2 Mg ha^{-1}

[2] ha: hectare, equivalent to hm^2

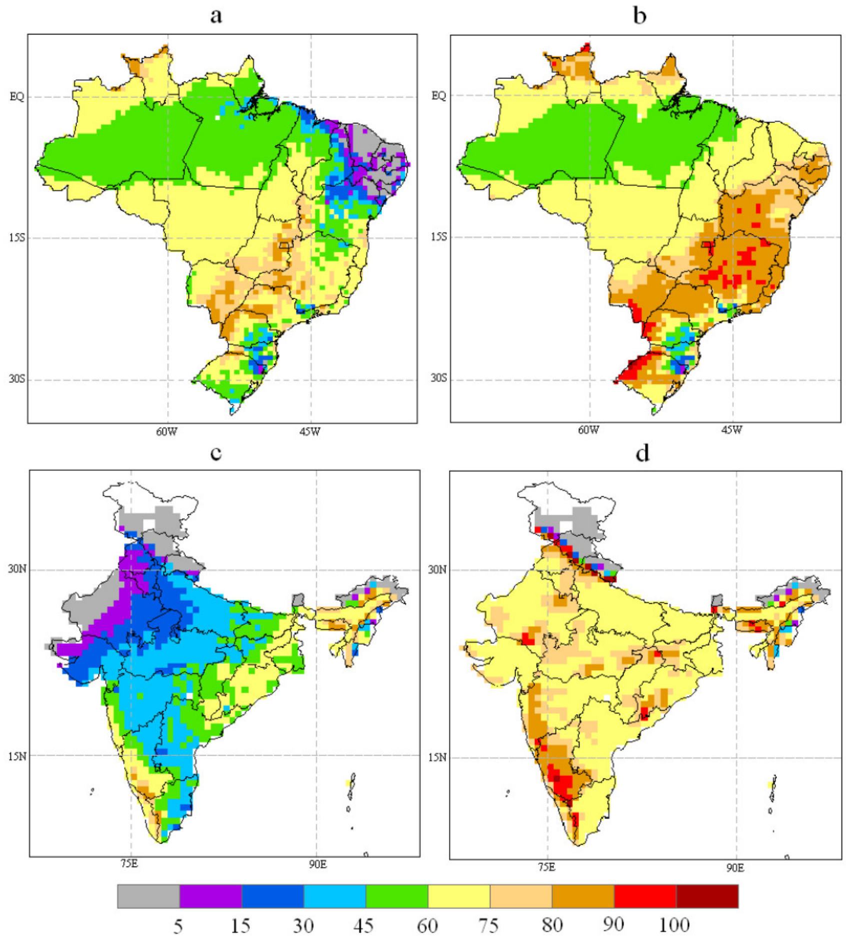


Figure 2.1: Sugarcane potential yields (Mg ha^{-1}) in Brazil (**a**, **b**) and India (**c**, **d**), under rainfed (**a**, **c**) and irrigated (**b**, **d**) conditions averaged for the 1971-2000 climate.

are found in the eastern states and in small areas of the southern states of Tamil Nadu, Karnataka and Kerala. Most of the country has a potential jatropha productivity ranging from 1-3 Mg ha^{-1} with smaller yields towards the northwest. Contrastingly, irrigated jatropha yields around 5.8 Mg ha^{-1} throughout most of India, with 6.9 Mg ha^{-1} peaks in the Western Ghats and other small regions. Country mean, lowest and highest potential yields for both crops are shown in Table 2.2.

Figures 2.3 and 2.4 show the additional land required to fulfill government targets on sugarcane ethanol and jatropha biodiesel respectively, for 2015 in Brazil and India and its variation depending on potential productivity. For rainfed sugarcane in Brazil, the average yield calculation indicates an additional 33,000 km^2 would be needed, representing half of the area actually covered by sugarcane in Brazil. Irrigation of sugarcane plantations could reduce this land demand by approximately 15%, with about 28,000 km^2 (42% of actual Brazilian sugarcane area) being

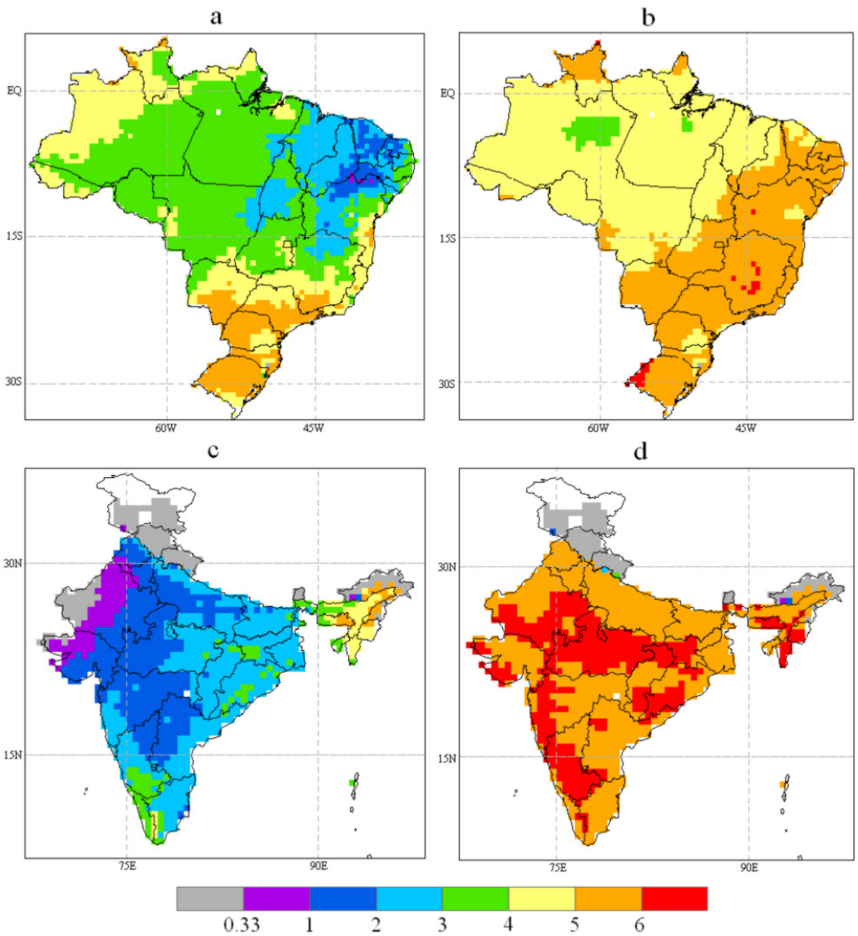


Figure 2.2: Jatropha potential yields (Mg ha^{-1}) in Brazil (**a**, **b**) and India (**c**, **d**), under rainfed (**a**, **c**) and irrigated (**b**, **d**) conditions averaged for the 1971-2000 climate.

required. Nevertheless, this number can be six-fold larger, in the rainfed case, if sugarcane is planted in low productivity areas like northeast Brazil. In India an additional 17,000 km^2 would be required on average, representing an increase of 43% in the area covered by sugarcane plantations in the country. This land requirement is considerably reduced if sugarcane is irrigated, with an additional 10,000 km^2 needed, representing 24% of actual Indian sugarcane fields (and it is likely that, as currently done, most sugarcane plantations in India will continue to be irrigated). However, even when irrigated, there are still large variations in the land requirements in India, tied to spatial variations in productivity, reaching a maximum of 38,000 km^2 if sugarcane is planted in low productivity areas (e.g., Himachal Pradesh State).

Land requirements are large for rainfed jatropha in India, with an average value of 212,000 km^2, representing about 13% of India's total cultivated land that would be needed to fulfill the

Table 2.2: Simulated country level potential yield (Mg ha^{-1}). Sugarcane and jatropha yields lower than 5 and 0.33 Mg ha^{-1} respectively are not considered.

		Brazil		India	
		rainfed	irrigated	rainfed	irrigated
Sugarcane	mean	60.11	68.85	60.11	73.33
	lowest	5.00	12.07	5.00	5.00
	highest	89.19	103.63	87.89	106.15
Jatropha	mean	3.77	4.88	2.20	5.89
	lowest	0.59	3.89	0.33	0.93
	highest	5.64	6.26	5.23	6.93

government planned production in 2015. In case jatropha is planted in high productivity sites, as in the eastern states, then 95,000 km^2 would be sufficient. However, if it is sown in low productivity areas, like northwest India, then 410,000 km^2 would be needed. Contrastingly, an average of just 79,000 km^2 would be required if jatropha plantations are irrigated. Spatial variations would be rather low since most of India is attaining a similar productivity when jatropha is irrigated. In Brazil rainfed jatropha would need about 29,000 km^2, with a maximum (minimum) of 89,000 (19,000) km^2 being required to fulfill the government plans for 2015. There is not much change in the average yield attained (22,000 km^2 required) with irrigated jatropha. But, like in India, spatial variations in productivity would be much lower than under rainfed conditions (18,000-28,000 km^2).

2.6 Discussion

Our results show that there is considerable spatial variability in the potential productivity of sugarcane and jatropha in Brazil and India. These regional differences lead also to significant differences in the extent of land required to cultivate these crops depending on where they are grown. Although not considered in this study, it is clear that land scarcity or competition with other land uses is to be expected with such an increase in the extension of biofuel crops in both countries. Due to similar ecophysiological and climatic requirements, potentially high biofuel crop productivity goes hand in hand with high food crop productivity [see *Monfreda et al.* 2008]. Thus stimulating biofuel plantations in high productivity sites may have the undesirable effect of displacing or replacing food crops. On the other hand, the production of energy crops on marginal lands (as proposed by India's National Mission on Biodiesel [*Planning Commission* 2003]) has a low energy return per area of land, and land occupation can be considerably larger than if crops were grown in high productivity sites. In any case, biofuel crop expansion in India should be planned very carefully since the country already exploits virtually all of its potential agricultural

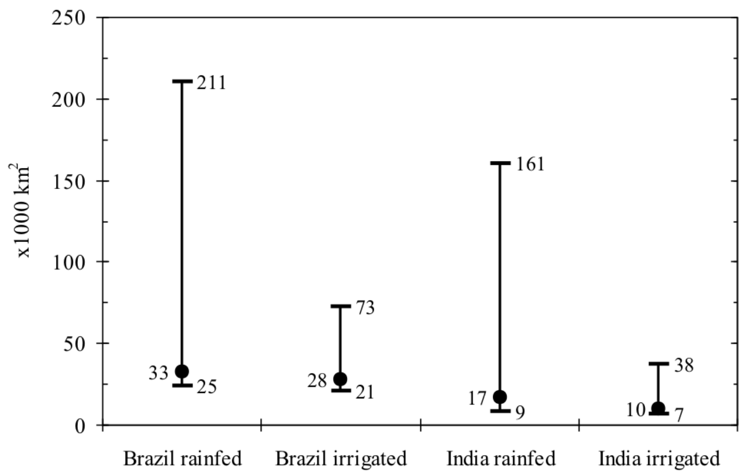


Figure 2.3: Additional land required to fulfill government targets for sugarcane ethanol production by 2015 compared to 2006-2007. Vertical lines indicate maxima and minima depending whether sugarcane is cultivated in low or high productivity areas of the countries respectively (see Table 2.2 for productivity values). Dots show the land required if sugarcane is planted in areas with the country's mean productivity.

land [*Ramankutty et al*. 2008; *Haberl et al*. 2007]. Thus, concurring increased demands for food and biofuels in India could probably push the cropland frontier into the already sparse and threatened natural vegetation areas of the country [*Schaldach et al*. 2010a].

The difference between our projections of land requirements for rainfed jatropha in India and official government numbers is remarkable (212,000 km^2 versus 119,000 km^2), considering that India's National Mission on Biodiesel predicts jatropha cultivation on marginal lands and receiving little management/irrigation inputs [*Planning Commission* 2003]. According to our calculations, the government's target of 119,000 km^2 [*Planning Commission* 2003] could perfectly be achieved if jatropha would be systematically cultivated in high productivity areas (e.g., Eastern States) or would be irrigated (almost anywhere in India). Nevertheless, one should notice that India's official projection was made based on the knowledge available in 2003. Anyhow, the discrepancy between our calculations and India's spatial target is an example of the importance of solid region-specific estimates on crop productivity to support realistic projections by decision makers.

In Brazil, the 22,000-29,000 km^2 (rainfed/irrigated) required for jatropha would represent just 11% of the actual coverage of soybean fields in Brazil. This result questions the benefits of continuing with soybean as the main source for biodiesel in Brazil. The 10,000 km^2 of land required for irrigated sugarcane ethanol in India is more than twice as large as the value calculated by *Schaldach et al*. [2010a] of 4,000 km^2 (the latter is the value equivalent to 2015 and normalized to the additional energetic demand of 32 PJ in Table 2.1), since in their study all

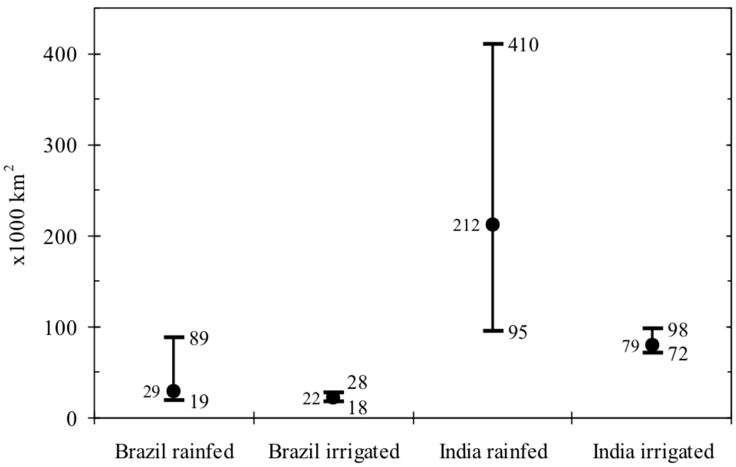


Figure 2.4: Additional land required to fulfill government targets for jatropha biodiesel production by 2015 compared to 2006-2007. Vertical lines indicate maxima and minima depending whether jatropha is cultivated in low or high productivity areas of the countries respectively (see Table 2.2 for productivity values). Dots show the land required if sugarcane is planted in areas with the country's mean productivity.

ethanol is obtained from the cane juice, and not from molasses, in the future. Contrastingly, our projection of 33,000 km^2 average land requirements in Brazil for rainfed sugarcane are well in agreement to what has been projected by the government (31,900 km^2 [*MME and EPE* 2007]) and the Brazilian sugarcane industry (34,400 km^2 [*UNICA* 2008]).

The increase in ethanol production in Brazil is very likely to be allocated close to the current sugarcane plantations in southeast Brazil, due to already existing infrastructure, and proximity to markets or exporting hubs. This region has the highest potential yields for sugarcane, but as well for other food crops too [*Monfreda et al*. 2008]. However, land requirements could be reduced by roughly 15% if sugarcane plantations were irrigated. Thus, unless yields are improved by substantial technological achievements or other management practices, some other land use will be displaced. Recent (last 5 years) expansion of sugarcane fields into pastures support the hypothesis that most of this expansion will displace cattle ranching in southeast Brazil [*Camargo et al*. 2008]. Nevertheless, sugarcane production is also feasible in other regions. In an attempt to separate deforestation of the Amazon from Brazil's growing sugarcane plantations, some [e.g., *UNICA* 2008] have argued that sugarcane production in the Amazon is not feasible from an agronomic point of view, since the plant needs a pronounced dry season to build up its biomass and concentrate sugar in the cane. However, our results and also reported field data [*IBGE* 2010] dismiss this myth. For example, in 2007 the municipality of Presidente Figueiredo in Central Amazon produced 280 Gg of sugarcane with a mean productivity of 70 Mg ha^{-1} [*IBGE* 2010], a value close to what we simulated for this region. For Brazilian biodiesel, it would be a good choice if plantations of jatropha were stimulated in parts of northeast Brazil where, though a semi-arid region, drought tolerant jatropha still provides reasonable seed yields (1-4 Mg ha^{-1}).

Obviously, potential productivity of these two crops and therefore the land requirements might be different from the values presented here if yields are increased by technological achievements (improved genotypes), management practices (increased fertilizer use) or altered by climate change. Additionally, it is possible that by 2015 second generation biofuels from cellulosic biomass could be fully commercially developed, allowing an even more effective use of land for bioenergy production [*Goldemberg* 2007; *Pousa et al*. 2007; *Tilman et al*. 2006]. Anyhow, for the time being, governments are still planning based on first generation biofuels. Therefore, this study is intended as a first quantitative assessment of land requirements for biofuel crops as dependent on site productivity and based on current demand projections.

2.7 Conclusions

Although the 2015 volumetric demand for ethanol in Brazil is about ten times larger than in India, land requirements for ethanol sugarcane are comparable in both countries due to large differences in the methods of producing ethanol from sugarcane. Moreover, land requirements for biodiesel production from jatropha in India can be considerably higher than previously thought, due to the low potential productivity of jatropha in most of India, if it is not irrigated or fertilized. The use of jatropha in Brazil, which has a potentially high productivity in most of the country, should be strongly considered, in order to replace soybean as the main feedstock for biodiesel production. That would considerably increase the energy return per piece of land, and thus reduce the pressure to further expand soybean plantations in the country. Irrigation could reduce the land requirements by 63% and 41% (24% and 15%) in India (Brazil) for jatropha and sugarcane respectively. While constraints on land availability are much larger in India than in Brazil, competition with other land uses is expected to play a key role in both countries. Therefore, scientifically based and fine-scale assessments of potential crop productivity like the present study can contribute to a better planning of land requirements of biofuel crops under high productivity and marginal conditions in face of the eminent expansion of biofuel production.

CHAPTER 3

Indirect land-use changes can overcome carbon savings from biofuels in Brazil

Summary

The planned expansion of biofuel plantations in Brazil could potentially cause both direct and indirect land-use changes (e.g., biofuel plantations replace rangelands, which replace forests). In this study, we use a spatially explicit model to project land-use changes caused by that expansion in 2020, assuming that ethanol (biodiesel) production increases by 35 (4) x10^9 L in the 2003-2020 period. Our simulations show that direct land-use changes will have a small impact on carbon emissions because most biofuel plantations would replace rangeland areas. However, indirect land-use changes, especially those pushing the rangeland frontier into the Amazonian forests, could offset the carbon savings from biofuels. Sugarcane ethanol and soybean biodiesel each contribute to nearly half of the projected indirect deforestation of 121,970 km^2 by 2020, creating a carbon debt that would take about 45 (sugarcane) and 250 (soybean) years to be repaid using these biofuels instead of fossil fuels. We also tested different crops that could serve as feedstock to fulfill Brazil's biodiesel demand and found that oil palm would cause the least land-use changes and associated carbon debt. The modeled livestock density increases by 0.09 livestock units (LU) ha^{-1}. But a higher increase of 0.13 LU ha^{-1} in the average livestock density throughout the country could avoid the indirect land-use changes caused by biofuels (even with soybean as the biodiesel feedstock), while still fulfilling all food and bioenergy demands. We suggest that a closer collaboration or strengthened institutional link between the biofuel and cattle-ranching sectors in the coming years is crucial for effective carbon savings from biofuels in Brazil.

3.1 Introduction

Brazil's government and biofuel industry are planning a large increase in the production of biofuels in the next 10 years. This increase is driven by internal and external market demand (ethanol), as well as by government-enforced blending (biodiesel) [*MME and EPE* 2007; *UNICA* 2008; *Pousa et al.* 2007]. Although Brazilian sugarcane ethanol is often considered to have one of the best production systems with respect to carbon savings [*Goldemberg* 2007; *Goldemberg et al.* 2008; *Scharlemann and Laurance* 2008; *Fargione et al.* 2008; *Leite et al.* 2009], there are concerns about the land-use changes (LUC) that would be incurred by an expansion of biofuel croplands [*Scharlemann and Laurance* 2008; *Fargione et al.* 2008]. Soybean plantations, from which most of the Brazilian biodiesel is produced [*MME and EPE* 2007; *Pousa et al.* 2007], already occupy 35% of the country's cultivated land [*IBGE* 2010]. It is known that biofuels can replace vast areas of farmland and native habitats, driving up food prices and resulting in little reduction of or even increasing greenhouse gas (GHG) emissions [*Scharlemann and Laurance* 2008; *Fargione et al.* 2008; *Righelato and Spracklen* 2007; *Nassar et al.* 2008; *Sawyer* 2008; *Crutzen et al.* 2008; *Hill et al.* 2006; *Searchinger et al.* 2008].

Previous studies focused on the direct land-use changes (DLUC) and the "carbon debt" caused by the replacement of native habitats by biofuel crops in Brazil [*Fargione et al.* 2008; *Leite et al.* 2009; *Righelato and Spracklen* 2007; *Nassar et al.* 2008]. Others pointed to the probable indirect land-use changes (ILUC) in Brazil caused by future expansion of biofuel croplands in the United States [*Hill et al.* 2006; *Searchinger et al.* 2008; *Nepstad et al.* 2006]. Overall, these studies show that potential LUC must be taken into account to assess the efficacy of a given biofuel. However, these studies were neither spatially explicit, nor did they explicitly consider competition between different land uses in view of concurrent food and biofuel demands. *Fargione et al.* [2008], for example, show the LUC carbon debt in terms of rate (e.g., $MgCO_2$ ha^{-1}), since they did not consider the total extent of land dedicated to biofuels or the total area of native habitats affected. Therefore, the net debt in absolute terms (e.g., $MgCO_2$) arising from future biofuel production remains undetermined. Moreover, the cascade effect of biofuel crops pushing the agricultural and cattle ranching frontier is still poorly understood.

Most of Brazil's sugarcane expansion in the last 5 years occurred on land previously used as rangeland in the southeastern states [*Nassar et al.* 2008; *Camargo et al.* 2008]. The same holds true for more than 90% of the soybean plantations in the Amazon region after the 2006 moratorium was implemented [*ABIOVE* 2009]. One of the potential consequences of such LUC is the migration of cattle ranchers to other regions and possible increased deforestation [*Nepstad et al.* 2006; *Fearnside* 2008; *Rigon et al.* 2005; *Morton et al.* 2006]. In light of the role rangeland plays in deforestation in Brazil [*Nepstad et al.* 2006; *Fearnside* 2008; *Rigon et al.* 2006; *Morton et al.* 2006] and the steadily increasing cattle herd (average of 3 million additional head per year in the 1974–2007 period [*IBGE* 2010]), the ILUC to replace rangeland displaced by biofuels are highly important [*Melillo et al*. 2009].

In this study we use a spatially explicit modeling framework to project the DLUC and ILUC arising from the fulfillment of Brazil's biofuel production targets for 2020 concurrent with increasing food and livestock demands. This modeling framework comprises: (*i*) a land-use/land-cover change model for land-use suitability assessment and allocation [*Schaldach and Koch* 2009]; (*ii*) a partial equilibrium model of the economy of the agricultural sector for future food and livestock demands as well as technological improvements of crop yields [*Rosegrant et al*. 2008]; and (*iii*) a dynamic global vegetation model for crop and grassland potential productivity driven by climate [*Bondeau et al*. 2007; see chapter 2]. Competition among land uses (for land resources) is considered based on a multicriteria evaluation of suitability, hierarchical dominance of major land-use activities (settlement, crop cultivation, grazing), and a multiobjective land allocation algorithm which looks for land-use pattern stability. Final outputs of this modeling framework are maps of land use and livestock density (Ld). DLUC and ILUC are determined by comparing land-use maps derived from scenarios with and without biofuel expansion. A number of different scenarios are considered to assess the isolated contribution of ethanol and biodiesel fuel production, as well as their impacts on different native habitats. The carbon debt and payback time from such LUC are calculated by using the average emission values employed by *Fargione et al*. [2008]. We investigate only the effects of ILUC inside Brazil. We do not consider cellulosic biofuels because the technological development of these fuels is unlikely to be fast enough to enable their large scale use in Brazil by 2020 [*Robertson et al*. 2008].

3.2 Methods

3.2.1 Modeling Framework

The central part of our modeling framework is the LandSHIFT model, which simulates land-use and land-cover change in a spatially explicit way at a resolution of 5 arc minutes [*Schaldach and Koch* 2009]. The model relies on a "land-use systems" approach that describes the interplay between anthropogenic and environmental system components as drivers for LUC in three major land-use activities (settlement, crop cultivation, and grazing) and their competition for land resources. It calculates not only the occurrence of grazing but also the intensity at which it occurs. LandSHIFT has been applied and validated in assessments of the impact of grazing management in the Jordan River region [*Koch et al*. 2008], quantification of future LUC and water use by agriculture in Africa [*Alcamo et al*. 2010; *Weiß et al*. 2009], and LUC associated with increased production of biofuels in India [*Schaldach et al*. 2010a]. The framework also comprises other models that, although not coupled to LandSHIFT, provide inputs to the model. The International Model for Policy Analysis of Agricultural Commodities and Trade (IMPACT) [*Rosegrant et al*. 2008] calculates future country-level food demands and technological improvements of crop yields, and the International Futures model [*Hughes* 1999] projects population growth. The LPJ for managed Lands (LPJmL) dynamic global vegetation model is used to calculate crop and grassland potential productivity on a 0.5° resolution grid [*Bondeau et al*. 2007; see chapter 2]. Starting from an initial land-use map, the spatial allocation of different land uses in subsequent time steps is based on a multicriteria-suitability analysis following the equation:

$$\psi_k = \underbrace{\sum_{i=1}^{n} w_i \; p_{i,k}}_{suitability} \times \underbrace{\prod_{j=1}^{m} c_{j,k}}_{constraints} \qquad , with \sum_i w_i = 1, \;\; and \;\; p_{i,k}, c_{j,k} \in [0,1] \qquad (3.1)$$

where the factor-weights w_i determine the importance of each suitability factor p_i at grid cell k, and c_j determines constraints for changing the land-use type at that given cell. In this study, p_i includes potential crop/grassland yield, slope, proximity to settlements, proximity to cropland, road network, and soil fertility (the latter does not apply for grazing). Therefore, n = 6 (5 for grazing).

The weights w_i for cropland were determined with the use of the analytic hierarchy process test [*Saaty* 1980]. The determination of the relative importance of each p_i factor in relation to the others (RI_{AHP}), which is used as an entry to the analytic hierarchy process test followed four steps: (*i*) determination of the coefficient of variation of the given p_i factor over the entire initial land-use map ($CV^1{}_i$); (*ii*) determination of the coefficient of variation of the given p_i factor only over the grid cells covered by cropland in the initial land-use map ($CV^2{}_i$); (*iii*) derivation of an empirical index for the p_i factor (EI_i) by $CV^1{}_i/CV^2{}_i$; and (*iv*) determination of RI_{AHP} with a pair wise comparison of EI_i from all p_i factors. Weights for road network, slope, and soil were fine-tuned from 0.23 to 0.13, from 0.18 to 0.23, and from 0.23 to 0.29, respectively (Table 3.1), to improve spatial distribution of croplands inside the country. The weights w_i for rangeland were assigned all of the same value of 0.2 [*Alcamo et al.* 2010].

Constraints c_j are applied in cells that are designated as conservation areas or according to the land use transition in question (Table 3.2). A third "constraint" was implemented for sugarcane

Table 3.1: Weights w_i for factors p_i used in LandSHIFT's cropland module for this study.

p_i **factor**	w_i **weight**
Potential crop yield	0.23
Proximity to cropland	0.08
Proximity to settlements	0.04
Road network	0.13
Slope	0.23
Soil fertility	0.29

Table 3.2: Land-use transition constraints (c_i) used in this study. Transition to forest or other native habitat is not modeled.

From \ To	Urban	Cropland	Rangeland	Set-aside
Urban	-	0.0	0.0	0.0
Cropland	1.0	-	0.5	1.0
Rangeland	1.0	1.0	-	1.0
Forest	1.0	0.5	0.5	0.0
Other native habitat	1.0	0.5	0.5	0.0
Set-aside	1.0	1.0	1.0	-

and soybean. This constraint represents the preferential occurrence of these crops in places where specific infrastructure is or will be implemented (such as ethanol mills [*Goldemberg et al.* 2008]) or as in the case of soybean, where production costs are lower [*Pousa et al*. 2007] and there is political facilitation for the cultivation of soybean [*Nepstad et al.* 2006; *Fearnside* 2008]. The suitability for sugarcane is increased by 35% in the states of São Paulo, Minas Gerais, Mato Grosso do Sul, Goiás, and Distrito Federal. Suitability for soybean is increased by 35% in the states of Goiás, Tocantins, Mato Grosso do Sul, and Distrito Federal. In Mato Grosso, suitability is increased by 80%. These values were chosen to better reproduce the area of sugarcane and soybean in these states (see full model evaluation in Appendix B). The allocation algorithm assumes that crop cultivation takes place on the most suitable cells for each crop type and calculates a "quasi-optimum" spatial crop distribution. However, the multiobjective land allocation heuristic used here seeks pattern stability and respects previous land use, even if another crop type has a higher suitability in that cell. Besides soybean and sugarcane, nine other major crop types are considered, including maize, pulses, rice, and wheat. LPJmL yields are applied a crop-specific factor to match current crop yields to statistics on the country level (Table 3.3) [*Schaldach et al*. 2010b; *FAO* 2010]. These factors, which are calculated at the first simulation time step, account for uncertainties because of crop management, (e.g., multicropping), or discrepancies because of the aggregation of crop types into the LPJmL crop functional types (e.g., LPJmL pulses refer to extratropical pulses, such as lentils). Crop production of a given grid cell *k* is defined as the potential crop yield at *k* multiplied by the area in *k* that is not covered by settlement.

Allocation of rangeland relies on the potential productivity of grass in the grid cells, based on a livestock feed supply-demand logic. Forage supply is calculated by summing up the grass productivity of every rangeland cell multiplied by the fraction of biomass that can be used by livestock (grazing efficiency g_e = 0.3 [*Pedreira et al*. 2005]). Forage demand is determined by the multiplication of the total livestock herd by the average forage consumption per livestock unit (4.6 Mg yr^{-1} [*Stéphene and Lambin* 2001; *Krausmann et al*. 2008]). We assume that 95% of the livestock demand is fulfilled by forage from pastures [*Krausmann et al*. 2008]. If forage demand

Table 3.3: Average actual crop yields modeled by LPJmL compared to FAO statistics for Brazil. To match FAO yields, a management factor is applied to the LPJmL yields before the yields are fed into the LandSHIFT model. Modeled net yield changes are split into changes due to technological improvements such as increased irrigation and plant breeding [*Rosegrant et al*. 2008; *Rothman et al*. 2007] and changes due to climate change (temperature, precipitation and atmospheric CO_2 concentration) [*Bondeau et al.* 2007] in Brazil. A comparison between projected changes and real yield changes in the last 20 years in Brazil is also shown [*FAO* 2010].

	Actual yields*, Mg/ha			2003-2020 yield changes†, Mg/ha			Annual change rate, kg/ha/yr	
Crop type‡	**LPJmL**	**FAO**	**Management factor**	**Due to technology**	**Due to climate**	**Net**	**LandSHIFT 2003-2020**	**FAO 1986-2006**
Sugarcane	67.9	66.4	1.0	26.9	4.5	31.4	1850	632
Soybean	1.2	2.1	1.8	0.6	0.2	0.8	46	39
Sunflower/Rapeseed	1.4	1.3	0.9	0.4	0.1	0.5	31	46
Oil palm§	13.2	10.0	0.8	4.1	6.1	10.2	600	-26
Jatropha curcas	3.7¶	-	1.0	1.5	0.1	1.6	95	-
Maize	3.1	2.5	0.8	1.5	0.1	1.6	94	78
Pulses	2.5	0.6	0.2	0.2	0.0	0.2	13	21
Rice	5.2	2.6	0.5	1.4	0.4	1.8	108	93

* For the 1991-2000 period [see *Bondeau et al*. 2007].

† All 2020 yields are below maximum theoretical yields [*Luyten* 1995]. See also *Rosegrant et al*. [2008] and *Rothman et al*. [2007] for assumptions used by the IMPACT model for the 'GEO4-Sustainability First' scenario).

‡ More than 90% of the cultivated area in Brazil is comprised by these crop types.

§ Oil palm yields, which are not modeled by the LPJmL model, are derived by applying a factor of 6.0 to the tropical roots crop functional type.

¶ In fact this is the potential yield (average yield of every grid cell) since there is no consistent information on the location and extension of actual *Jatropha curcas* plantations in Brazil.

is higher than the supply, then new rangeland cells are allocated, starting from grid cells with higher suitability and continuing until demand is fulfilled. Average livestock density (Ld) is calculated by dividing the total livestock herd by the rangeland area. Preferential allocation of land-use activities follows the order: settlement, crop cultivation, grazing. Only one dominant land-use type can occur in a grid cell.

3.2.2 Input Data and Modeling Protocol

LandSHIFT is initialized with a combined map of land cover and land use for the year 1992 [*Heistermann* 2006], a map of population density [*Goldewijk* 2005], and national statistics of crop production and livestock herd [*FAO* 2010]. Socioeconomic projections include future demands for food production, technological improvements of crop yields [*Rosegrant et al*. 2008], and population growth [*Hughes* 1999] generated for the United Nations Environment Programme's Global Environmental Outlook 4 (GEO4) report under the Sustainability First scenario [*Rothman et al*. 2007]. We focus our analysis on this scenario because it predicts the highest use of biofuels worldwide by far and the largest increase in food production in Brazil [*Rothman et al*. 2007]. For the sake of scenario consistency, future potential crop and grassland productivity was calculated with the LPJmL model [*Bondeau et al*. 2007; see chapter 2] in 0.5° spatial resolution using as input a climatology of temperature, precipitation, and [CO_2] from the IMAGE model [*NEAA* 2006], which was also generated for the GEO4 report [*Rothman et al*. 2007]. The Sustainability First scenario used here depicts a global mean increase in temperature of 1.1 °C in 2020 in relation to preindustrial times and an atmospheric CO_2 concentration of 426 ppmv [*Rothman et al*. 2007]. There is a national population increase from 177 million people in 2003 to 202 million in 2020, with an average growth rate of 1.62% per year [*Hughes* 1999]. Oil palm yields, which are not modeled by LPJmL, are simulated by applying a factor of 6.0 to the yields of the tropical roots crop functional type. Resulting yields are in accordance with oil palm yields in Brazil from census data for the 1990s [*IBGE* 2010] (Northeastern Pará: simulated = 11.4 Mg ha^{-1}, census = 13.6 Mg ha^{-1}; eastern Bahia: simulated = 3.3 Mg ha^{-1}, census = 4.1 Mg ha^{-1}; roughly 99% of Brazil's oil palm area is located in these two regions). Average potential yields for the 1990s are used as baseline yields in LandSHIFT (Table 3.3). On average, food production increases by 86% in the 2003 to 2020 period (Table 3.4), and yields increase, on average, by 62% because of the combined effects of technological improvements and climate change (Table 3.3). Livestock herd grows from 149 million livestock units (LU)[3] in 2003 to 234 million LU in 2020, with an annual increase of 3.4%. This increase rate is slightly larger than the 3.25% average annual growth rate observed over the last 30 years [*IBGE* 2010].

Biofuel production follows the official projections by the Brazilian government and the biofuel industry [*MME and EPE* 2007; *UNICA* 2008] (Table 3.5). Demands for food and biofuels are fed separately into the model, but they are treated equally inside the model algorithm. No preference

[3] Equivalent to 1.315 cattle heads in Brazil [*FAO* 2010].

Table 3.4: Crop production in 2003 [*FAO* 2010] and 2020 [*Rothman et al.*2007], and changes from 2003 to 2020 in Brazil.

Crop type	2003, Gg	2020, Gg	Δ, %
Wheat	5033	8741	73.7
Other temperate cereals	744	1402	88.6
Rice	11343	17619	55.3
Maize	41928	73640	75.6
Tropical cereals	1560	2831	81.5
Pulses	54831	105172	91.8
Temperate roots	3022	5447	80.3
Tropical roots	23811	33775	41.8
Other annual oil crops	443	717	61.8
Soybeans	47604	77756	63.3
Sugarcane	388184	763493	96.7
Permanent crops / Vegetables	53486	88349	65.2
Total	631989	1178942	86.5

is given to either food or biofuels. The main claims for using *Jatropha curcas* as a biodiesel feedstock are its drought tolerance, the low management inputs needed for its cultivation, and the inclusion of small farmers in the production chain (see chapter 2), which is in accordance to Brazil's National Program on Biodiesel Production [*Pousa et al*. 2007]. For that reason, we restrict the occurrence of *J. curcas* to Northeast Brazil, which is the region targeted by the Brazilian government for inclusion of smallscale farming [*Pousa et al*. 2007]. This restriction is not applied to the other feedstocks.

Four scenario variations are modeled: (*i*) biofuel targets: 2020 food + 2020 biofuel production; (*ii*) no increase in biofuel production: 2020 food + 2003 biofuel production; (*iii*) ethanol targets only: 2020 food + 2020 ethanol + 2003 biodiesel production; and (*iv*) biodiesel targets only: 2020 food + 2003 ethanol + 2020 biodiesel production. DLUC are determined by the changes in the area covered by biofuel crops in variation (*i*) compared to variation (*ii*). ILUC are determined by the difference in the area covered by land uses other than biofuel crops between variations (*i*) and (*ii*). The intensification of livestock needed to avoid ILUC by biofuels is estimated by increasing the grazing efficiency (g_e) factor to the level at which rangeland area is equal to that of variation-scenario (*i*) minus the area of rangeland displaced by biofuels.

3.2.3 Model Evaluation (Short)

LandSHIFT model results for Brazil were evaluated in three aspects (a detailed presentation is given in Appendix B):

3.2.3.1 *Crop and Rangeland Location*

A comparison of LandSHIFT's calculated suitability with a reality land-use map showed a tendency for the occurrence of high suitability values in crop and rangeland, suggesting the procedure used in LandSHIFT is reasonable for allocation of crop and rangeland (Figure B.1). The land-use map used in this comparison is the same as the one used in the determination of the w_i weights, generating a spurious dependency between the datasets used for comparison. However, an analysis where all w_i weights were set to the same value of 0.16 further confirmed the tendency of high-suitability values in crop and rangeland grid cells. A second test using the relative operating characteristics (ROC) method [*Pontius and Schneider* 2001] showed that the spatial pattern computed by LandSHIFT (ROC[cropland] = 0.87; ROC[rangeland] = 0.80) is not random, in which case it would have a value of 0.5 (Figure B.2).

3.2.3.2 *Crop and Rangeland Area*

Modeled cropland and rangeland areas are in very good agreement with country-level reported statistics [*FAO* 2010], suggesting the model is able to convert country-scale crop production mass into cropland area (Figure B.3). Overestimation of rangelands by 8% might be the result of an underestimation of grassland productivity and also because of the assumption of only one land use per grid cell, which leads to overestimation of the rangeland area, especially in regions where Ld is low, as in Northeast Brazil. Crop and rangeland areas within major regions of Brazil are also in good agreement with statistics [*IBGE* 2010], except for the overestimation of rangeland in Northeast Brazil (Figure B.4).

3.2.3.3 *Deforestation Rates*

The modeled annual deforestation rate for the Amazon region in the 1992 to 2003 period compares well with remote sensing data (LandSHIFT: 16,789 $km^2\ yr^{-1}$; INPE-PRODES: 18,266 $km^2\ yr^{-1}$ [*PRODES* 2009]). The shares of this deforestation among states are also comparable with

Table 3.5: Biofuel production in Brazil in 2003 and projections for 2020 [*MME and EPE* 2007; *UNICA* 2008]. Sources for biofuel yields: *UNICA* [2008]; *Jongschaap et al*. [2007]; *Achten et al*. [2008]; *Crutzen et al.* [2008]; *Wirsenius* [2000].

Biofuel	Year	Volume, $x10^9$ liter	Feedstock	Biofuel yield, liter/Mg	Production, Tg
Ethanol	2003	14.50	sugarcane	85	170.59
Ethanol	2020	50.03	sugarcane	85	588.53
Biodiesel	2003	0.50	soybean	200	2.50
Biodiesel	2020	4.47	soybean	200	22.33
Biodiesel	2020	4.47	jatropha	278	16.07
Biodiesel	2020	4.47	sunflower/rapeseed	448	9.97
Biodiesel	2020	4.47	oil palm	490	9.12

PRODES, even though deforestation in Maranhão is overestimated by a factor of 23. However, the land-use map used for model initialization (based on IGBP-DISCover dataset [*Heistermann* 2006]) has 80% more forest in Maranhão compared to PRODES [*PRODES* 2009]. Moreover, LandSHIFT does not consider forestry activities, which might influence deforestation rates. The modeled deforestation rate of Central Brazil Cerrado for the 1992 to 2003 period is 17,753 km^2 yr^{-1}, an amount that lies within the estimated range for the last decade (13,100–26,000 km^2 yr^{-1} [*Sawyer* 2008]).

3.2.4 Carbon Debt and Payback Time

Carbon debt and payback time are calculated following the approach used by *Fargione et al.* [2008], with two major differences. First, the final numbers are absolute values ($MgCO_2$) rather than rates ($MgCO_2$ ha^{-1}) because we calculate total LUC, and second, the annual CO_2 offset by biofuels are calculated on a per ton basis instead of a per hectare basis. All numbers used in the carbon debt and payback time calculations are shown in Tables 3.6 and 3.7.

3.3 Results

3.3.1 Direct Land-Use Changes

Our simulations with increased biofuel production show that the expansion of sugarcane plantations in response to increased ethanol production would take place mostly in the southeastern states (São Paulo, Minas Gerais, Rio de Janeiro, Paraná) and, to a lesser extent, in northeast Brazil (Figures 3.1 and 3.2 and Table 3.8). The expansion of soybean plantations in response to increased biodiesel production would happen mainly in the states of Mato Grosso, Mato Grosso do Sul, Goiás, and Minas Gerais. Sugarcane and soybean have potential yield increases of 31.4 and 0.8 Mg ha^{-1}, respectively (Table 3.3). To fill the biofuel production targets for 2020, sugarcane would require an additional 57,200 km^2 and soybean an additional 108,100 km^2. Roughly 88% of this expansion (145,700 km^2) would take place in areas previously used as rangeland. Food cropland area replaced by biofuels would reach 14,300 km^2. In our simulations, direct deforestation is only caused by soybean biodiesel and amounts to only 1,800 km^2 of forest and 2,000 km^2 of woody savanna. Carbon emissions as a result of DLUC would originate mainly from soil carbon losses when converting rangeland to sugarcane or soybean plantations.

A payback time of 4 years would be necessary to compensate for the sugarcane DLUC emissions with the use of sugarcane ethanol instead of fossil fuels. For soybean biodiesel, DLUC carbon emissions would not be paid back for at least 35 years, primarily because the annual per hectare carbon savings from soybean biodiesel are much smaller than from sugarcane. Despite an increase of 86% in food demand, 4% of the cultivated land (26,000 km^2) is spared in the scenario without the expansion of biofuel croplands (in comparison with 2003) because of higher crop yields driven by technological improvements and climate change (Tables 3.3 and 3.4).

Table 3.6: Carbon debt estimates (CO_2 emissions from soils and aboveground and belowground biomass caused by land-use change) used in this study. Sources: *Fargione et al.* [2008]; *Searchinger et al*. [2008].

Previous land-use (from)	To cropland $MgCO_2e./ha$	To rangeland* $MgCO_2e./ha$	To well-managed rangeland† $MgCO_2e./ha$
Cropland	0	0	0
Rangeland	75	0	0
Other natural vegetation	85	69	13
Woody savanna	165	145	60
Tropical forest	737	690	572

* Soil carbon emissions are 20% lower [see *Cerri et al.* 2007].

† Soil carbon emissions are hypothetically reduced to zero (see section 3.4).

Table 3.7: Proportion of total land-use change carbon debt (see Table 3.6) allocated to biofuel production, and estimates of annual life-cycle GHG reduction from biofuels (including displaced fossil fuels, soil carbon storage and fertilizer use, but not land-use change emissions) used in this study. Sources: *Fargione et al.* [2008]; *Agricultural Marketing Service of the USDA* [2009]; *Center for Jatropha Promotion* [2009]; *Renewable Energy UK* [2009]; *Macedo et al.* [2004]; *Gärtner and Reinhardt* [2003]; *Reinhardt et al*. [2007].

Biofuel	Debt allocated to biofuel* %	Annual GHG offset $MgCO_2e./Gg$ of harvested feedstock
Sugarcane ethanol	100	162
Soybean biodiesel	39	429
Sunflower/Rapeseed biodiesel	82†	935
Jatropha biodiesel	72‡	378
Oil palm biodiesel	87	710

* See *Fargione et al*. [2008] for definition.

† Considering 2007 prices of $1.26 for oil and $0.2 for seed cake

‡ Considering 2007 prices of $0.5 for oil and $0.2 for seed cake

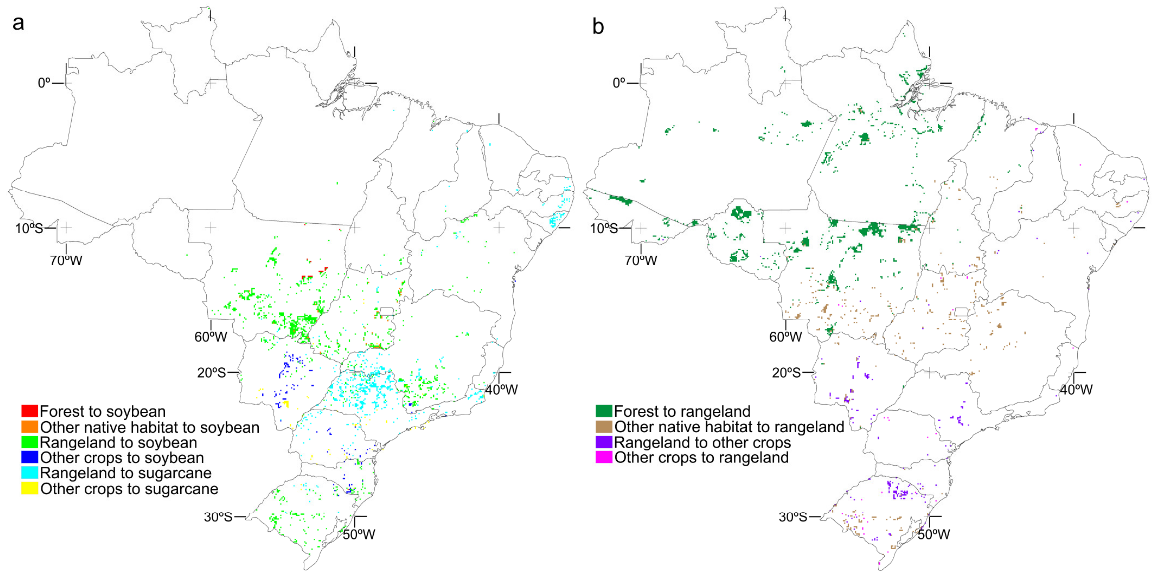


Figure 3.1: Modeled direct (**a**) and indirect (**b**) land-use changes caused by the fulfillment of Brazil's biofuel (sugarcane ethanol and soybean biodiesel) production targets for 2020.

3.3.2 Indirect Land-Use Changes

ILUC could considerably compromise the GHG savings from growing biofuels, mainly by pushing rangeland frontier into the Amazon forest and Brazilian Cerrado savanna. In our simulations, there is an expansion of 121,970 km^2 of rangeland into forest areas, and 46,000 km^2 into other native habitats, due to the expansion of biofuel croplands (Table 3.8). Modeled country-wide average Ld increases by 0.09 LU ha^{-1} in the 2003 to 2020 period if ILUC by biofuels are not avoided, because of the occupation of more (potentially) productive grid cells in the Amazon region. Sugarcane ethanol and soybean biodiesel would be responsible for 41% and 59% of this indirect deforestation, respectively. These percentages were determined by fulfilling only the demand for sugarcane ethanol, while keeping soybean biodiesel production at current levels and vice-versa. Higher potential productivity of grass favors allocation of rangelands in Amazonia instead of in other native habitats. However, when comparing the scenarios with and without increased biofuel production, the displacement of rangelands previously located in high productivity sites in Southeast Brazil to lower productivity sites in Central Brazil causes the newly allocated rangeland area [170,370 km^2 (71.9 by sugarcane + 98.5 by soybean)] to be higher than that displaced by biofuels [145,700 km^2 (52.7 by sugarcane + 93.0 by soybean)]. Ld increase is 0.001 LU ha^{-1} higher in the scenario without increased biofuel production than in the scenario with increased biofuel production. Food croplands displaced by biofuels are not necessarily cultivated in land farther away from cities, and in fact the mean distance of the displaced food croplands to the largest cities is reduced by 17%. It is important to stress that we are not trying to

Table 3.8: Land-use and land-use change (relative to 2003) according to different modeled scenarios for Brazil in 2020. ILUC, indirect land-use changes; LU: livestock units (1.315 cattle heads in Brazil [*FAO* 2010]).

	Soybean	**Sugarcane**	**Other crops**	**Rangeland**	**Forest**	**Other Natural**	**Livestock Density**
Scenario	**x1000 km^2 (Δ03-20, %)**	**x1000 km^2 (Δ03-20, %)**	**x1000 km^2 (Δ03-20, %)**	**x1000 km^2 (Δ03-20, %)**	**x1000 km^2 (Δ03-20, %)**	**x1000 km^2 (Δ03-20, %)**	**LU/ha (Δ03-20, %)**
2003	191 (-)	55 (-)	389 (-)	2133 (-)	4194 (-)	1496 (-)	0.70 (-)
2020 biofuel targets	285 (+49.3)	90 (+63.6)	397 (+2.1)	2972 (+44.3)	3546 (-15.4)	1222 (-21.6)	0.79 (+12.3)
2020 no biofuel expansion	178 (-6.9)	33 (-39.9)	398 (+2.4)	2968 (+44.1)	3668 (-12.5)	1268 (-18.7)	0.79 (+12.5)
2020 ethanol targets only	178 (-6.8)	90 (+62.8)	398 (+2.4)	2973 (+44.3)	3618 (-13.7)	1255 (-19.5)	0.79 (+12.3)
2020 biodiesel targets only	285 (+49.5)	33 (-39.7)	396 (+2.0)	2968 (+44.1)	3598 (-14.2)	1232 (-21.0)	0.79 (+12.5)
2020 biofuel targets no ILUC	285 (+49.3)	90 (+63.6)	397 (+2.1)	2807 (+36.2)	3668 (-12.5)	1268 (-18.7)	0.83 (+18.9)

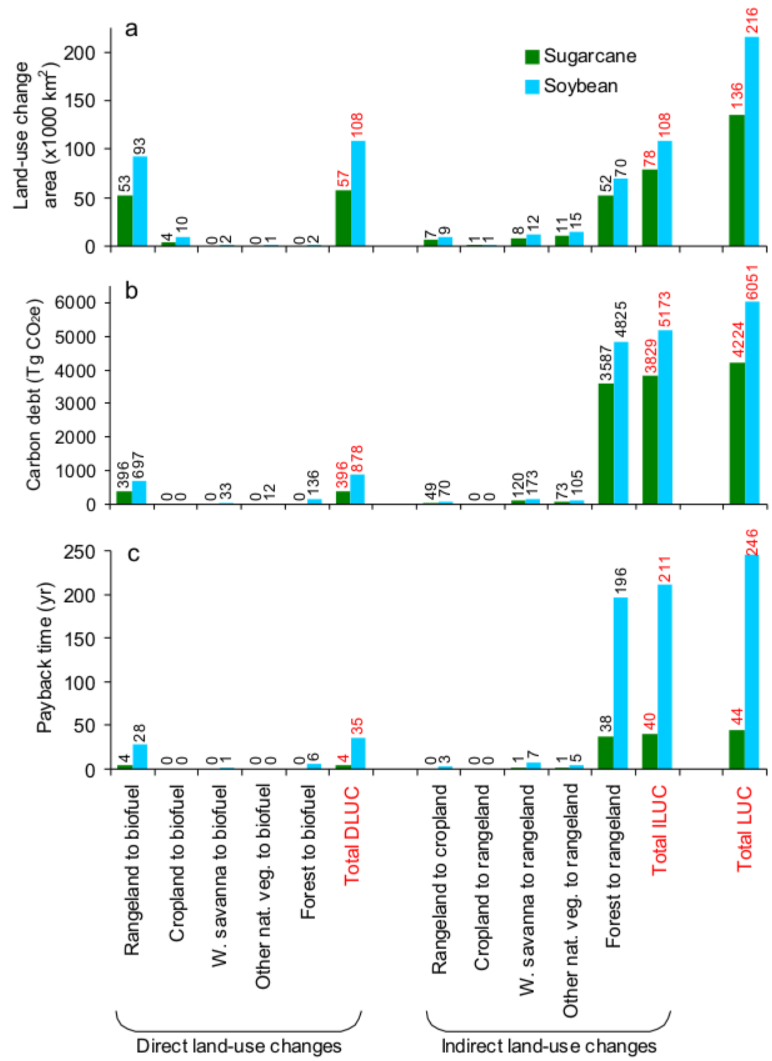


Figure 3.2: Direct, indirect and total land-use changes areas (**a**), carbon debt (**b**), and time to repay debt (**c**) for fulfilling Brazil's biofuel (sugarcane ethanol and soybean biodiesel) production targets for 2020. Here the land-use category "cropland" excludes sugarcane and soybean. Other nat. veg., other natural vegetation; W. savanna, woody savanna.

pinpoint the exact places to be indirectly affected by the expansion of biofuel croplands with Figure 3.1b, as this map is only the difference between the land-use maps with and without biofuels in 2020 (Figure 3.3). Instead, it should be regarded as a spatial evidence of the magnitude that the ILUC might have in the near future because of an expansion of biofuel plantations. The consideration of carbon emissions from ILUC would extend the payback time for sugarcane ethanol by an additional 40 years and for soybean biodiesel by 211 years. Therefore, the payback time for the total LUC (DLUC+ILUC) for sugarcane and soybean would be 44 and 246 years,

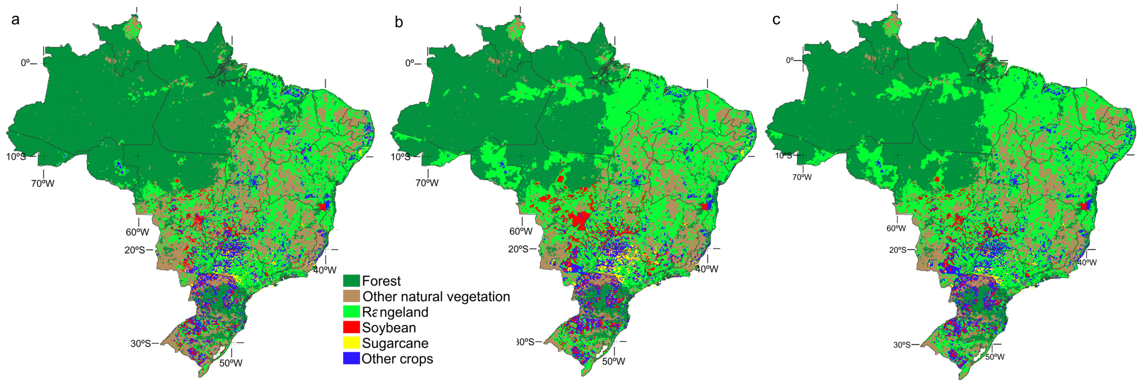


Figure 3.3: Modeled land-use maps for the year 2003 (**a**), 2020 with fulfillment of Brazil's biofuel targets (sugarcane ethanol and soybean biodiesel) for 2020 (**b**), and 2020 with biofuel production at the same level as in 2003 (**c**).

respectively.

Although the area dedicated to rangeland does not differ greatly between the scenarios with and without increased production of biofuels, the extent of native habitats that are displaced by rangeland is considerably different (Table 3.8). Therefore, avoiding ILUC by biofuels would demand a smaller increase in rangeland area (~8% less rangeland compared to the 2020 scenario with ILUC). To achieve such a reduction in rangeland area but still meet the same livestock demand, Ld would need to be increased by 0.13 LU ha^{-1} in comparison with 2003 values.

3.3.3 Other Biodiesel Feedstocks

It can be argued that soybean is not the most efficient feedstock for biodiesel because it occupies large tracts of land, incurs considerable carbon debt (even without considering ILUC), and has a low annual rate of saved carbon from replacing fossil diesel. Therefore, we tested other feedstock options that could serve to fulfill Brazil's 2020 production demand for biodiesel. Our results show (Figure 3.4 and 3.5) that if the smallest area and carbon debt from LUC are given priority, then oil palm would be the best feedstock for biodiesel by far. Because of its high oil yield, oil palm would need only 4,200 km^2 to fulfill the 2020 demand for biodiesel in Brazil. In comparison, 108,100 km^2 would be needed for soybean, 73,000 km^2 for rapeseed/sunflower, and 31,700 km^2 for *Jatropha curcas*. The payback time for oil palm would be 7 years for DLUC, which is much smaller than the DLUC payback time of 27 years for sunflower/rapeseed. However, if oil palm is strictly planted only in rangeland areas, the DLUC payback time would be reduced to 4 years. Sunflower/rapeseed plantations would be located mainly in south-central states. *J. curcas* plantations, which were forced to occur only in Northeast Brazil in our simulations (section 3.2.2), are concentrated in the coastal area, where potential yields are higher. Oil palm plantations

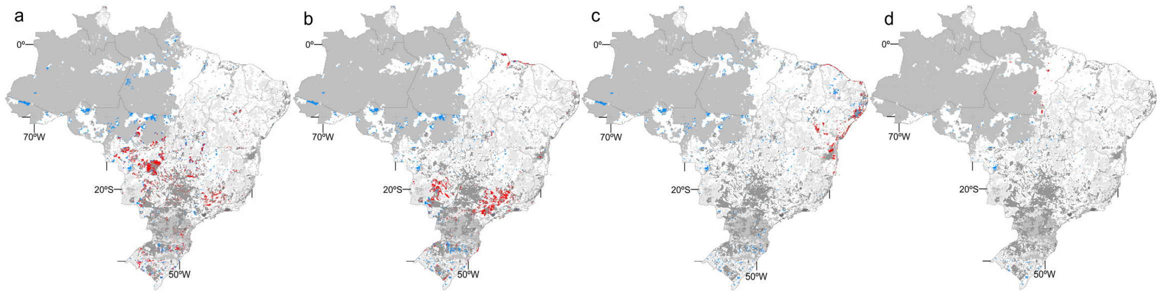


Figure 3.4: Fulfilling Brazil's biodiesel production target for 2020 with different feedstocks: (**a**) soybean, (**b**) sunflower/rapeseed, (**c**) *Jatropha curcas*, (**d**) oil palm. Red, direct land-use changes; blue, indirect land-use changes (see Figure 3.5 for carbon debt and payback time).

would be located entirely in Pará state, close to the Amazon forest and where most current plantations are located [*IBGE* 2010]. Oil palm would incur some direct deforestation (300 km^2), although much less than that directly caused by soybean. If oil palm is used as biodiesel feedstock in conjunction with sugarcane for ethanol, then Ld would need to be increased by only 0.10 LU ha^{-1} from 2003 to 2020 to avoid ILUC, compared to the 0.13 LU ha^{-1} increase needed for the soybean-sugarcane combination.

3.4 Discussion

Our results show that sugarcane-ethanol and oil palm-biodiesel grown in Brazil are the best plant feedstocks in terms of carbon savings for fulfilling the country's demand for biofuels in 2020, assuming that the LUC associated with the increased production are restricted to the DLUC in rangelands. The simulated DLUC, which occur predominantly in rangelands, have already been observed for sugarcane [*IBGE* 2010; *Nassar et al*. 2008; *Camargo et al*. 2008] and soybean [*Nepstad et al*. 2006; *ABIOVE* 2009; *Fearnside* 2008] in recent years. For sugarcane, this trend will probably continue in the next years because of the growing number of standards being imposed on sugarcane plantations [*UNICA* 2008; *Roundtable on Sustainable Biofuels* 2008]. However, the proximity of sugarcane plantations to Atlantic forest remnants in Southeast Brazil is of particular concern, considering that any further deforestation there would have major impacts on the biodiversity and connectivity of this highly threatened forest [*Ribeiro et al*. 2009]. The moratorium on soybean introduced in 2006 has proven to be an efficient way for preventing deforestation directly caused by soybean production in the Amazon region [*ABIOVE* 2009]. Moreover, increasing pressure by the media and non-governmental organizations [*Greenpeace* 2006] suggests that the moratorium will continue to be respected in the coming years. Even though oil palm is strongly associated with deforestation in Southeast Asia [*Koh and Wilcove* 2008], the Brazilian palm oil production is still small and could be expanded into non-forest sites,

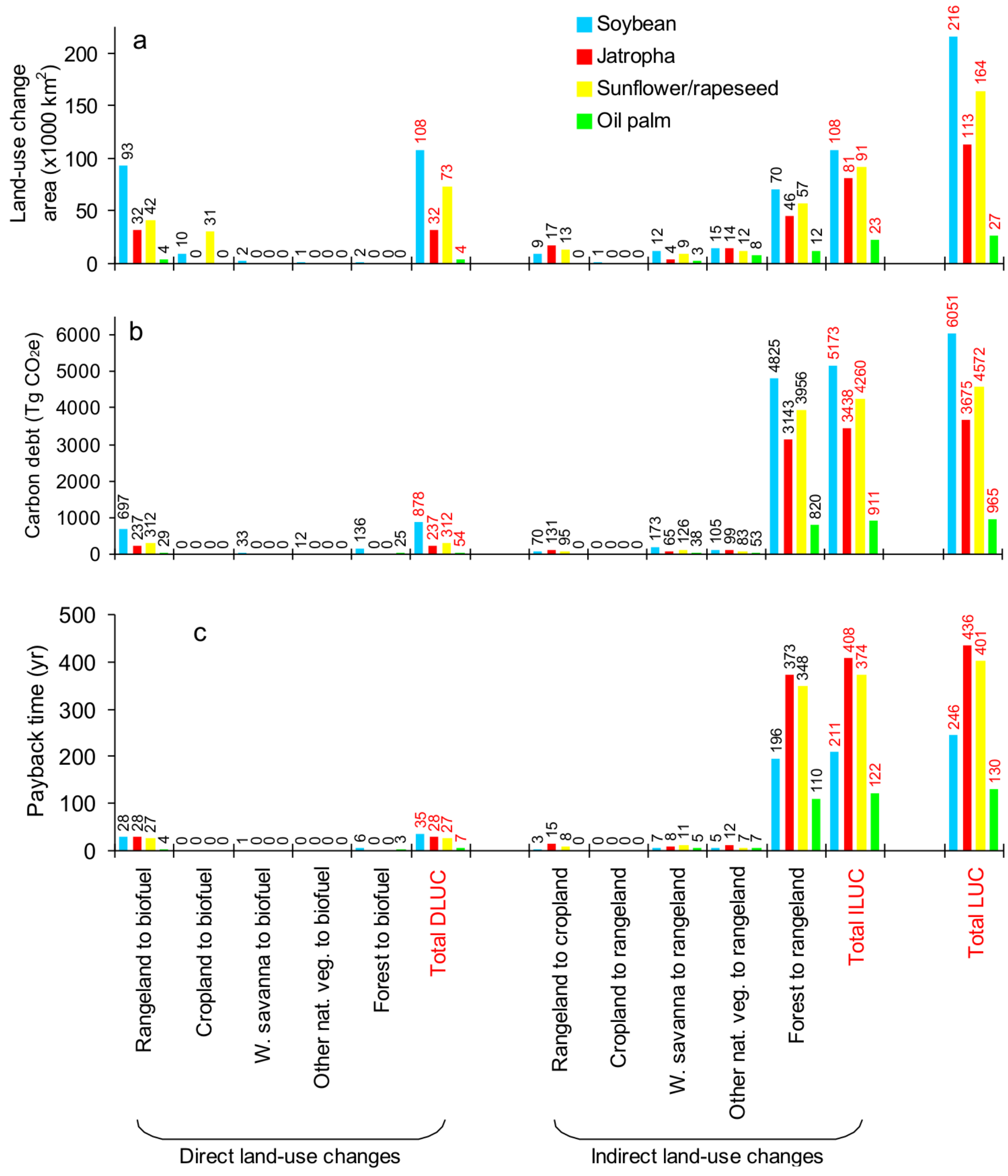


Figure 3.5: Land-use changes, carbon debt and time to repay debt for fulfilling Brazil's demand for biodiesel in 2020 with different feedstocks. W.: woody; nat. veg.: natural vegetation.

assisted by improved governance in the Amazon region [*Butler and Laurance* 2009; *Soares-Filho et al*. 2006].

The efficacy of biofuels in Brazil can be considerably compromised if biofuel-related ILUC, namely moving the rangeland frontier into native habitats, take place as projected here. It has been suggested that ILUC indeed occur in the Amazon region, especially the case where rangeland is shifted by soybean and reestablished elsewhere closer to the deforestation frontier [*Nepstad et al*. 2006; *Fearnside* 2008; *Rigon et al.* 2005; *Morton et al*. 2006]. However, to our knowledge there has been no research quantifying these ILUC and establishing their cause–effect relationships. Therefore, although difficult to validate, the ILUC driven by biofuels projected in our simulations is a hypothesis that cannot be disregarded and may indeed happen in the next years. The question, then, is whether all of the displaced rangeland will need to be reallocated and where this will happen. Roughly 36% of the national cattle herd and rangeland area is currently located in the Brazilian Amazon region, the only region in Brazil that has experienced an increase of rangeland area in the last two decades [*IBGE* 2010; *Barreto et al*. 2008]. In part, this suggests that the expansion of cultivated land in other regions of Brazil is pushing the rangeland frontier into the Amazon forest. Steady annual deforestation rates of the Brazilian Cerrado savanna indicate that this sort of ILUC may also be happening in Central Brazil [*Sawyer* 2008], despite a decrease in the area of rangeland there [*IBGE* 2010].

Animal acquisition is heavily subsidized in Brazilian cattle ranching, especially in the Amazon region, but very few incentives are provided specifically for the recovery of degraded pastures and intensification of grazing [*Nepstad et al*. 2006; *Barreto et al*. 2008; *Fearnside* 2002]. Moreover, land tenure issues do not encourage the intensification of cattle ranching in the region. For example, in many cases Ld is kept at a minimum level only to guarantee ownership over public land [*Nepstad et al*. 2006; *Barreto et al*. 2008; *Fearnside* 2002]. Roughly 290,000 km^2 of land, equivalent to 15% of the currently grazed rangeland, was once grazed in Brazil and is now abandoned [*Campbell et al*. 2008]. Furthermore, up to 60% of the currently grazed rangeland face some form of degradation and could have its productivity improved [*FAO* 2007a]. In that sense, our results (LUC and carbon debt) can be regarded as conservative because rangeland degradation processes, which would increase land requirements for livestock, are not considered in our simulations. If we assume that all rangeland areas will be well managed, meaning that there will be no soil carbon losses [*Cerri et al*. 2007], then the overall carbon debt would be reduced only by 13% because most of the carbon lost in the LUC at forest areas is stored in the vegetation. Still, studies suggest that technological innovation or the intensification of livestock inside the Amazon region may increase the attractiveness of cattle ranching there and further stimulate deforestation [*Fearnside* 2002; *Cattaneo* 2005]. Therefore, an increase in livestock intensity in Brazil by 0.13 LU ha^{-1}, as proposed here, is perfectly possible from a biophysical point of view with the enhancement of grass productivity and introduction of innovative management practices [*FAO* 2007a]. From a socioeconomic point of view, however, increasing Ld in Brazil involves complex interactions between granting the right subsidies [*Barreto et al*. 2008], governance over land ownership [*Fearnside* 2008; *Soares-Filho et al*. 2006], and an increased interconnection between land-use sectors (this latter proposed in this study). We argue that to avoid the undesired ILUC by biofuels presented here, strategies for cooperation between the cattle ranching and biofuel-growing sectors should be implemented by the biofuel sector (based on the sector's own interest in minimizing GHG emissions), and institutional links between these two sectors should be strengthened by the government. For example, biofuel growers should be able to track the amount of displaced cattle when the rangeland-to-biofuel crop transition takes place and guarantee that this demand will be compensated elsewhere in more intensified conditions. In other words, biofuel organizations and the government should support initiatives toward modernization of the cattle ranching sector to guarantee that the production of biofuels is not causing ILUC, which would compromise the efficacy (in terms of carbon savings) of their own product. Such a requirement should also be considered as a standard for the production of sustainable biofuels [*Roundtable on Sustainable Biofuels* 2008].

In fact, our results could be worse in view of the somewhat optimistic increases in potential crop yields projected because of technological improvements compared to the crop yield changes observed in the last 20 years. For example, in our simulations technological improvements increase sugarcane yields by 26.9 Mg ha^{-1} in the 2003 to 2020 period, compared to the 12.6 Mg ha^{-1} increase observed in the last 20 years (Table 3.3). Such optimistic yield increases are bound to the storyline of the scenario used here (section 3.2.2), which, besides predicting a high use of biofuels, also predicts high investments in yield enhancements. If we assume that there will be no enhancements of potential yields until 2020, then the payback time of the DLUC (ILUC) carbon debt would increase to 6 (62) and 50 (301) years for sugarcane and soybean, respectively. In that case, Ld would need to be increased by 0.14 LU ha^{-1} to avoid ILUC, compared to the 0.13 LU ha^{-1} calculated for 2020 with the yield improvements shown in Table 3.3. In addition, we do not account for fertilizer and water requirements associated with these yield improvements [*Melillo et al*. 2009]. Overall, our study should be viewed as the lower limit of the probable effects of biofuels on LUC in Brazil, because we predict substantial ILUC, even with optimistic assumptions (e.g., no rangeland degradation and high yield improvements).

Finally, the efficacy of biofuels is analyzed here in terms of GHG savings and not from the socioeconomical perspective. As a counterpart to the ethanol production chain, Brazil's National Program on Biodiesel Production seems to aim at promoting small-scale farming and shortening dependence on conventional diesel [*Pousa et al*. 2007]. However, between 75 and 95% (depending on the year) of the biodiesel produced in Brazil so far comes from soybean grown on plantations that are owned or controlled by large-scale farmers, and at production costs that are higher than for production of fossil diesel [*MME and EPE* 2007; *Nepstad et al*. 2006; *Fearnside* 2008]. Comprehensive assessments of labor conditions, land division, food prices, and other socioeconomical implications arising from the expansion of biofuels in Brazil are yet to be done. Nevertheless, joining life-cycle assessment figures to spatially explicit LUC projections, like the present study does, allows for a more accurate evaluation of the efficacy of biofuels in terms of carbon savings.

CHAPTER 4

Implementing a new land use allocation mode in LandSHIFT: spatial analysis of biogas production in Germany

Summary

In this study we present a new method of land use allocation for LandSHIFT as influenced by the occurrence of specific infrastructure units (SIU), such as biogas plants or ethanol mills, around which certain land uses have a higher likelihood of occurrence. Therefore in this method both the suitability analysis and land use allocation are performed only in the neighborhood of the SIU; and the extent of feedstock plantations is determined by the SIU's capacity. We exemplarily apply the method to assess the land use changes that might incur with the fulfillment of Germany's current biogas plant capacity. Although 20% to 35% of the arable land of Germany is abandoned and could potentially be used to increase electricity generation out of renewable sources, feedstock transport distance to plants is a crucial issue for biogas production. We make, thus, simulations in which biogas plantations have priority over other crops (biogas crop first) and vice-versa (other crops first). The main outcome of these exemplary runs is that under the 'other crops first' scenario only 10% of the capacity cannot be fulfilled because of lack of available land in the neighborhood of biogas plants. Moreover, our simulations indicate that biogas plants located in South and Southwestern (North and Northeastern) Germany would face more (less) difficulties to fulfill their capacities with cultivated feedstocks, in view of the distance of plantations to biogas plants. The combination of the presented method with refined data on plants/mills will allow for a detailed and more realistic analysis of the expansion of crops that are tightly linked to SIU.

4.1 Introduction

The occurrence and extension of infrastructure such as roads, electrical grids, port hubs, or processing mills have been identified as a major proximate cause of land-use and land-cover changes (LUCC) [*Geist et al.* 2006; *Geist and Lambin* 2002]. Therefore, the elsewhere occurrence of crop and grazing land can be, at least partially, explained by the presence and extent/capacity of infrastructure. Nevertheless, some land uses, especially bioenergy crop cultivation, are more strongly associated with specific infrastructure units (SIU), as is the case for sugarcane plantations in the proximities of sugar/ethanol mills [*Goldemberg et al.* 2008; see also chapter 3] or silage crop fields in the neighborhood of biogas plants [*Walla and Schneeberger* 2008]. The correct simulation of geographical location and pattern of bioenergy crop cultivation allows not only for a more precise estimate of its environmental effects [*Hellmann and Verburg* 2008], but also provides insights on the economic return they yield, considering that the higher the distance to the processing mill/plant, the higher the costs involved in the transport of the feedstock [*Walla and Schneeberger* 2008].

In this study we present the implementation of a new method of land use allocation in the LandSHIFT model of LUCC [*Schaldach and Koch* 2009; *Schaldach et al.* 2010b] as influenced by the occurrence of SIUs. This method is exemplarily applied in a first assessment of the potential availability of land to fulfill Germany's current biogas plant capacity with cultivated feedstocks. Some factors put the use of land for production of biogas in a preferable position in comparison to biofuels such as ethanol: flexibility of feedstocks that can be used for producing biogas [*Amon et al.* 2007a]; the use of the digestate as a high quality fertilizer [*Amon et al.* 2007b]; powering automobiles with biomass-generated electricity leads to greater GHG savings than with biomass fuels [*Campbell et al.* 2009]. However, the prioritization of land uses other than silage crops in the neighborhood of a biogas plant might influence both the availability and transport distance of feedstocks to the plant, which are critical issues for biogas production [*Walla and Schneeberger* 2008; *Gunnarsson et al.* 2008]. Therefore, we also investigate how the prioritization of other land uses over biogas crop cultivation (and vice-versa) could affect the fulfillment of the country-wide biogas plant capacity, and feedstock transport distances.

Biogas plants in Germany have an overall capacity of 2.03 GW_{el} for electricity generation, and represented 2.3% of the total electricity generated in the country in 2005 [*Scholwin et al.* 2008; *Capros and Mantzos* 2008]. Rough estimates account that approximately 50% of this capacity is currently fulfilled with cultivated feedstocks (e.g. silage maize). On the other hand, from 20% to 35% of the arable land in Germany is currently abandoned [*Campbell et al.* 2008; this study], and could be used for growing energy crops, increasing the share of renewable sources in the country's energy matrix.

4.2 Methods

The LandSHIFT model, fully described by *Schaldach et al*. [2010b] (see also chapters 3 or 4), simulates land-use and land-cover change in a spatially explicit way in 5 arc-minutes resolution. Here the spatial allocation of biogas crops is done in a different way compared to other crops, and is based on the occurrence and capacity of a SIU. LandSHIFT's previous allocation method was done with iteration through every grid cell of the analyzed raster (e.g., a country). In the new method presented here both the suitability analysis and land use allocation is performed only in the neighborhood of a SUI (Figure 4.1), which in this study case is represented by a biogas plant. According to the map shown in Figure 4.2, we designate that each German district has, in its geographical center, a "virtual biogas plant" (hereafter VBP), which concentrates the capacity of all the biogas plants in that given district. Therefore, these VBPs have a much higher capacity than "real" biogas plants in Germany, which hardly surpass the capacity of 2500 kW_{el}. Nevertheless, this 'VBP approach' is necessary in light of the interplay between LandSHIFT resolution (5 arc-minutes in the global version), and the total number of biogas plants in Germany (more than 8,000; *Scholwin et al.* [2008]).

The model iterates through all the 310 VBP and, using a previously calculated suitability analysis, allocates biogas crops in the neighborhood of that given VBP before moving to the next VBP. Biogas crops can occupy a maximum of 1/3 of a grid cell's area, to represent a crop rotation scheme, which is typically used in sustainable farming systems [*Amon et al*. 2007b]. The maximum neighborhood searching radius is set relating the VBP power to the maximum distance that silage feedstocks can be cultivated as to make biogas production profitable (Table 4.1) [*Walla and Schneeberger* 2008]. Nevertheless suitability decreases linearly with distance from

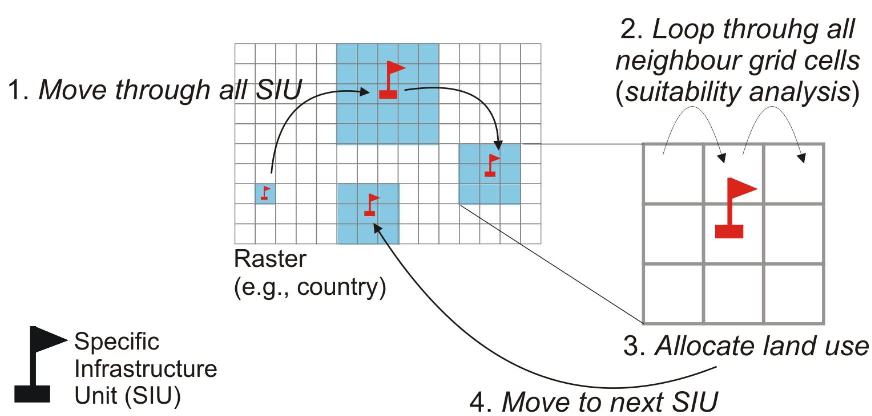


Figure 4.1: Illustrative design of the new method for land use allocation within the LandSHIFT model, as influenced by the occurrence and capacity of specific infrastructure units (SIU). Notice that the radius of analysis in the neighborhood of a SIU (grid cells in blue) is defined by the processing power of the SIU (size of SIU symbols); and that the grid cell in which the SIU is located is also considered in the suitability analysis and land use allocation.

the virtual biogas plant.

In this study we consider five factors for the suitability analysis, all having the same weight of 0.2 (as in the study by *Alcamo et al*. [2010]): proximity to croplands, proximity to settlements, slope, potential crop/grass yield, and road network. Crop/grass yields are calculated with the LPJmL model on 5° resolution [*Bondeau et al*. 2007], and are applied a crop-specific factor to match country level reported yields [*FAO* 2010]. Data source for the other factors are the same as in the study by *Alcamo et al*. [2010]. We assume a corn fraction of 40% in order to derive silage maize yields out of common maize yields. LPJmL rye yields, which are the yields of rye seeds, are applied a ratio of 1.3 to account for other parts of the plant which are used in silage [*Kim and Dale* 2004]. The initial land-use map for the year 2000 was prepared for this study out of the CORINE land cover map [*Keil et al*. 2005] and the approach suggested by *Heistermann* [2006] for agricultural land allocation.

Demands for biogas crops are set according to a district level map of Germany's electrical power of biogas plants in 2007[4] (Figure 4.2). This power is translated into crop demands by considering that plants work 80% of a years's time, and by applying the biogas-electricity yield specific to each feedstock (Table 4.2). Districts with no data available are not considered in the simulations.

Firstly two sets of experiments are carried out to check the availability of land for biogas production in Germany, and the influence that a prioritization of other land uses over biogas crop cultivation (and vice-versa) might have in this availability: (*i*) biogas crops have priority over

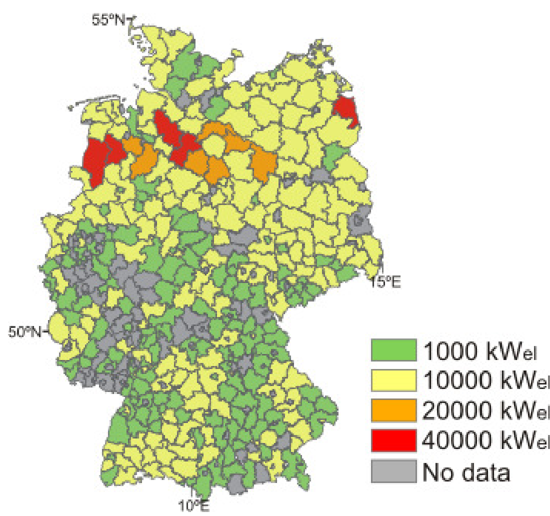


Figure 4.2: Maximum biogas plant capacity in the German districts in 2007 (adapted from *Scholwin et al.* [2008])

[4] Model results presented here are representative for the mid-2000's, considering the land-use map of Germany in the year 2000 and biogas plant capacity in the year 2007.

Table 4.1: Maximum capacity of “virtual biogas plants” (see text for definition) and radius of the supply area (based on *Walla and Schneeberger* [2008]).

Maximum capacity	Radius of the supply area	
kWel	grid cells	km
1,000	1	7.3
10,000	3	21.9
20,000	4	29.2
40,000	5	36.5

other agricultural uses, meaning it is allocated before other crops, and (*ii*) other crops have priority over biogas crops, meaning the latter can be allocated only in abandoned areas, after the allocation of other crops. In both cases urban areas and natural vegetation are not changed. Four biogas (cultivated) feedstock options are considered: silage maize, rye, grass and silage maize plus grass, this latter meaning that grass is used if there is pre-existent rangeland in the neighborhood of the VBP. Secondly, to evaluate the average distance of biogas crop plantations to the closest VBP we perform two additional runs (for each prioritization scenario) with silage maize and silage maize plus grass, assigning the same power (2000 kW_{el}) to all the VBPs, eliminating thus one degree of freedom in our analysis. Distance maps (considering a tortuosity factor[5] of 1.3 [*Walla and Schneeberger* 2008]) resulting from the two prioritization scenarios are submitted to a standard kriging interpolation in ArcGIS software. We derive then a map showing the average of the krigged maps of scenario *i* and *ii*, considering that reality might lie in between the two prioritization scenarios assumed here, since LandSHIFT does not explicitly model competition between land-uses.

4.3 Results

The simulated area covered with biogas crops for each of the feedstock options in the two prioritization scenarios is shown in Figure 4.3. Silage maize is the feedstock that occupies the least area, due to its high yield of electricity per unit of cultivated area. Under the ‘biogas crop first’ scenario 99.9% of Germany’s biogas plant capacity is fulfilled, with any of the feedstocks. Capacity cannot be fulfilled in 2 (out of 310) VBP due to their localization in the middle of natural vegetation patches. On the other hand, under the ‘other crops first’ scenario, roughly 10% of the capacity cannot be fulfilled with cultivated feedstocks. In this case, the area covered with the biogas crop also decreases, though not in 10% since plantations might be allocated now onto less productive cells.

[5] The relationship between actual transport distance (via roads), and the direct (straight line) distance.

Table 4.2: Biogas energy content [*Becker et al.* 2007] and average yield (whole plant) [*Bondeau et al.* 2007; *FAO* 2010] of different silage feedstocks in Germany.

Silage feedstock	**Biogas energy content**	**Average yield in Germany**
	kWh/MgDM	**MgDM/ha**
Silage maize	922.5	20.5
Rye	1276.6	13.1
Grass	837.2	12.2

Most of the unfulfilled capacity is found in eastern Lower Saxony (32%), western Baden-Wuerttenberg (22%) and southern Bavaria (20%). However, when silage maize is combined with grass from previously existent rangelands, this non-fulfillment drops to 2.5%. In this option with combined feedstocks the area requirement is 700 km^2 (6%) larger than when using only silage maize in the 'biogas crop first' scenario. The higher use of pre-existent rangeland (when comparing 'other crops first' to 'biogas crop first') denotes a somewhat lower availability of land for biogas crop plantations in the neighborhood of VBPs, if other crops are given priority over biogas crops.

Figure 4.4 shows the spatial distribution of biogas crops in both prioritization scenarios with the 'silage maize or grass' feedstock option, besides the base map of land use employed in the simulations. The major differences between the two prioritization scenarios are found especially in eastern Lower Saxony – a region that is mostly covered by (non-silage) croplands and, at the same time, has a high biogas-plant capacity (i.e., the competition between silage and non-silage crops might be high) – and, to a smaller extent in western Baden-Wuerttenberg , and southern Bavaria.

Figure 4.5 shows the average (between the two prioritization scenarios) krigged map of transport distances for feedstock options 'silage maize' and 'silage maize or grass'. North and east Germany are regions where plantations are located closer to VBPs, whereas south and west Germany have plantations located farer from VBPs in our simulations. Distances are reduced when VBPs are allowed to use grass from surrounding rangelands as feedstock to fulfill their capacities (average distances are significantly different between the two feedstock options [silage maize: 11.2 km (± 2.3 SD); silage maize plus grass: 9.1 km (± 2.3 SD); t = 52.5, P <0.0001])[6]. Some VBPs have their capacities not fulfilled with the 'silage maize' feedstock option in western North Rhine-Westphalia, which makes the kriging analysis underestimate distances in that region (compare Figures 4.5a and 4.5b in that region).

[6] The difference in average transport distance is larger between the two prioritization scenarios (biogas crops first: 7.2 km (± 3.2 SD); other crops first: 11.0 km (± 2.4 SD); t = 78.9, P < 0.0001), but here we focus our analysis (regarding transport distances) on the average of the two prioritization scenarios, with the argument that reality might lie in between them.

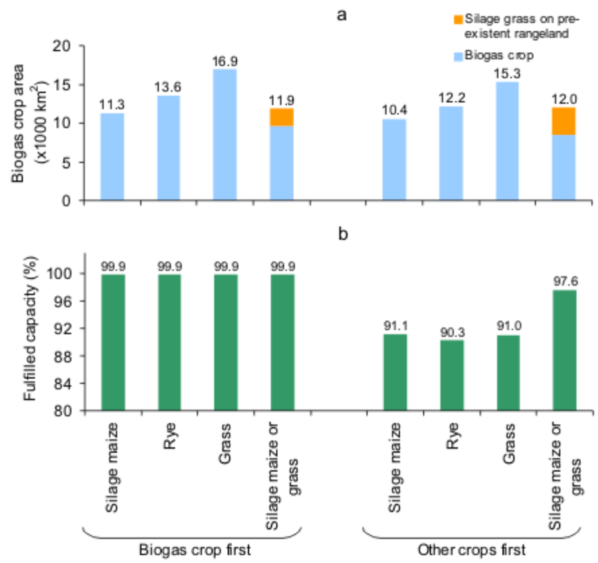


Figure 4.3: Area covered with biogas crops (**a**) and percentage of Germany's biogas plant capacity (2,03 x10^6 kW_{el}) that can be fulfilled (**b**) with different cultivated feedstock options, for two prioritization scenarios.

Transport distances are still large in some regions even with the feedstock option 'silage maize plus grass' (Figure 4.5b) because these regions are covered with natural vegetation (e.g., southern Bavaria and Baden Wuerttenberg) or largely covered with agriculture or urban areas (western North Rhine-Westphalia) (see Figure 4.5a).

4.4 Discussion

The main findings of this first assessment of the land potential for biogas production in Germany are listed and discussed below. But, importantly, one should notice that the results for this study case are preliminary, since it was intended only to test the new land use allocation method of LandSHIFT, was not subject to any validation, and the input data (e.g., Figure 4.2) and assumptions (e.g., tortuosity factor of 1.3) used can be refined in later simulations.

The ~12,000 km^2 required for biogas crops in our simulations corresponds exactly to the current extent of land used by all bioenergy crops in Germany, and three-fold the area of cropland currently dedicated only to biogas production [*Thrän and Kaltschmitt* 2007]. Nevertheless, that represents only 10% of the German agricultural land (cropland and rangeland) area, and, still, only 10% of the abandoned agricultural area [*Campbell et al. 2008*] (Figure 4.4a). It is probable that technological enhancements of crop yields together with the slightly declining population of Germany will cause further abandonment of agricultural areas in the near future [*Schaldach and*

Alcamo 2006; *Rosegrant et al*. 2001]. In light of that, our results suggest that there is far enough land to fulfill Germany's current biogas plant capacity with cultivated feedstocks. Nevertheless, one should ensure that soil carbon emissions resulting from the occupation of abandoned areas are avoided, since these carbon emissions have the potential to offset the carbon savings from the use of biogas instead of fossil fuels [*Gerin et al*. 2008; *Campbell et al*. 2009; *Fargione et al*. 2008; see also chapter 3]. No-tillage practices and other management predicted in sustainable farming directives [e.g. *European Environmental Agency* 2007] have the potential to satisfactorily restrain these soil carbon emissions [*Tebrügge and Düring* 1999], and should be contemplated in certification mechanisms of biogas production.

Our results also show that a "land-use synergy" between biogas crops and pre-existent rangelands increases the chances of fulfilling the biogas plant capacity with cultivated feedstocks. Germany has ~5,000 km^2 of grasslands available for cutting and use as feedstock for the production of bioenergy [*European Environmental Agency* 2007]. The use of these grasslands in our simulations reduces the "friction" (competition between land-uses is not explicitly modeled here) between biogas crops and other crops. Furthermore, simple grass cutting, without revolving the soil, incurs in less soil carbon losses than the occupation of abandoned areas by any biogas crop [*Fargione et al*. 2008]. Therefore, keeping a variety of cultivated feedstocks should be a priority in the biogas production system, as a manner to secure supply and consequently electricity production [*European Environmental Agency* 2007; *Thrän and Kaltschmitt* 2007].

The model runs to evaluate transport distances reveal that feedstock plantations are located closer

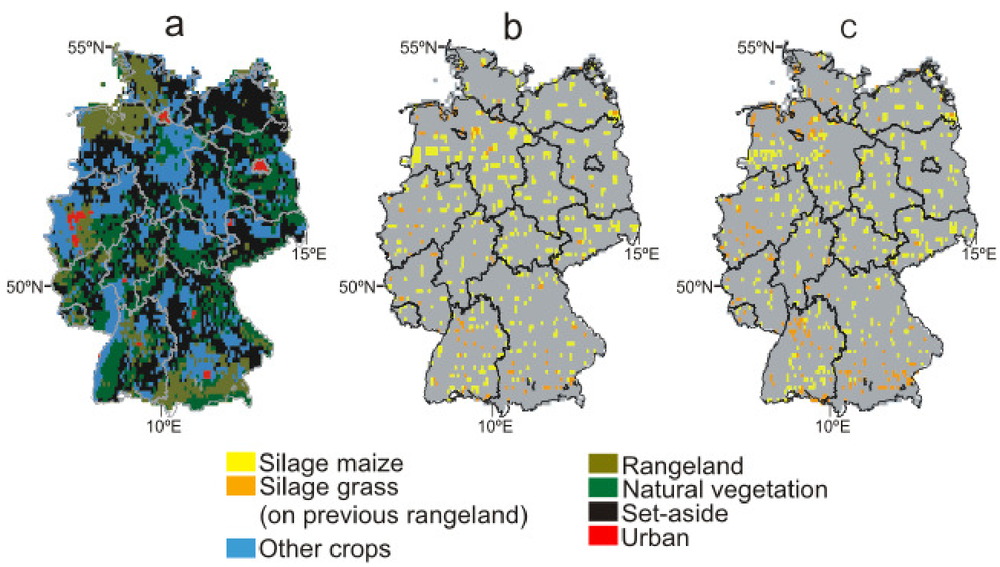


Figure 4.4: Base map of land use for the year 2000 employed in the simulations of this study (**a**), and the spatial distribution of biogas crops under 'biogas first' scenario (**b**) and 'other crops first' scenario (**c**), exemplarily shown here for the feedstock option 'silage maize or grass'.

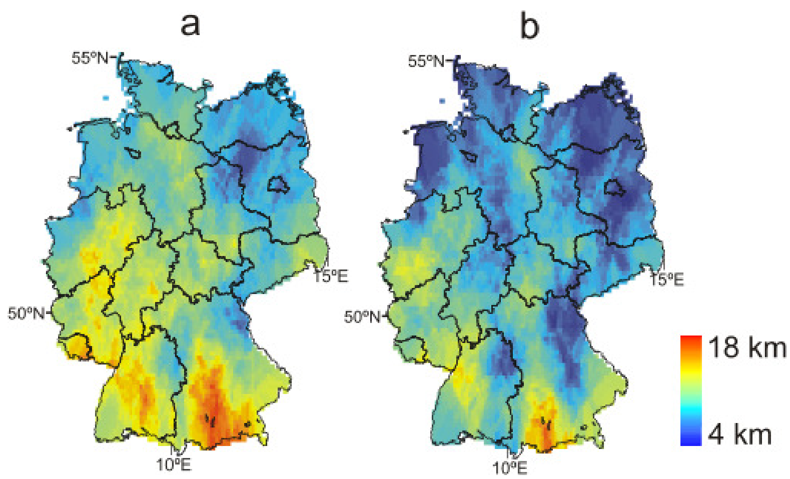


Figure 4.5: Modeled average distance of biogas crop plantations to the nearest virtual biogas plant using silage maize (**a**) and silage maize or grass (from pre-existent rangeland) (**b**) as feedstocks.

to biogas plants in north and east Germany. That means the economical return to biogas producers would probably be higher in these regions [*Walla and Schneeberger* 2008], whereas competition with other crops (as discussed by *Thrän and Kaltschmitt* [2007]) would be smaller. In fact these regions have already been identified as "hotspots" for cultivation of bioenergy crops in Germany by *Hellmann and Verburg* [2008]. Besides being well served with infrastructure and industrial facilities, these regions have large areas of abandoned land that are suitable for biogas crop plantations.

Finally, we believe the land use allocation method presented here is a promising new feature of LandSHIFT that can be applied in studies of LUCC issues involving land uses that are tightly linked to SIUs, such as bioenergy crop cultivation. The conceptual framework implemented for this study allows for more accurate geographical allocation of land use types than previously considered in LandSHIFT, obviously as long as trustworthy inputs and assumptions are provided to the model. With that, we can improve our understanding of the environmental impacts and logistic problems associated with certain land uses. Further refinements of this method should concentrate in the correct prediction of the location of SIU, be it in the sense of gathering state-of-the-art data on the location and capacity of SIUs, or in the sense of explicitly simulating current and future location of SIUs, such as done by *Hellmann and Verburg* [2008].

CHAPTER 5

Impacts of climate change and the end of deforestation on land use in the Brazilian Amazon

Summary

Climate change scenarios vary considerably over the Amazon region, with an extreme scenario projecting a dangerous (from the human perspective) increase of 3.8ºC in temperature and 30% reduction in precipitation by 2050. The impacts of such climate change on Amazonian land-use dynamics, agricultural production and deforestation rates are still to be determined. In this study we make a first attempt to assess these impacts through a systemic approach, using a spatially explicit modeling framework to project crop yield and land-use/cover changes in the Brazilian Amazon by 2050. Our results show that, without any adaptation, climate change may exert a critical impact on the yields of crops commonly cultivated in the Amazon (e.g., soybean yields are reduced by 44% in the worst scenario). Therefore, following baseline projections on crop and livestock production, a scenario of severe regional climate change would cause additional deforestation of 181,000 km^2 (+20%) in the Amazon and 240,000 km^2 (+273%) in the Cerrado compared to a scenario of moderate climate change. Putting an end to deforestation in the Brazilian Amazon forest by 2020 (and of the Cerrado by 2025) would require either a reduction of 26-40% in livestock production until 2050, or a doubling of average livestock density, from 0.74 to 1.46 head/ha. These results suggest (i) that climate change will affect land use in ways not previously explored, such as the reduction of yields entailing further deforestation, and (ii) the need for an integrated/multidisciplinary plan for adaptation to climate change in the Amazon.

5.1 Introduction

The Amazon has been recognized as a region particularly vulnerable to climate change over this century [*Lenton et al*. 2008; *Malhi et al*. 2008]. Although climate change scenarios for the region differ considerably [*Li et al*. 2006], the high end of projections show a temperature increase of 3.8ºC and up to 30% reduction in precipitation by 2050 (Figure 5.1). The impacts of such regional climate change and of the projected "forest dieback" on the vegetation dynamics, water and carbon cycle, as well as feedbacks with the global climate system have been extensively investigated in the last decade [*Cox et al*. 2000, 2004; *Cramer et al*. 2001; *Huntingford et al*. 2004, 2008; *Sitch et al*. 2008; *Lapola et al*. 2009]. In addition, field observations [*Gash and Nobre* 1997] as well as modeling studies [*Nobre et al*. 1991; *Costa and Foley* 2000; *Sampaio et al*. 2007] have shown that there is considerable change in the local and regional climate after the replacement of forest by pasture or crops. On the other hand, considerably less research has been done to assess the effects of future climate on land-use and land-cover dynamics in the Amazon region.

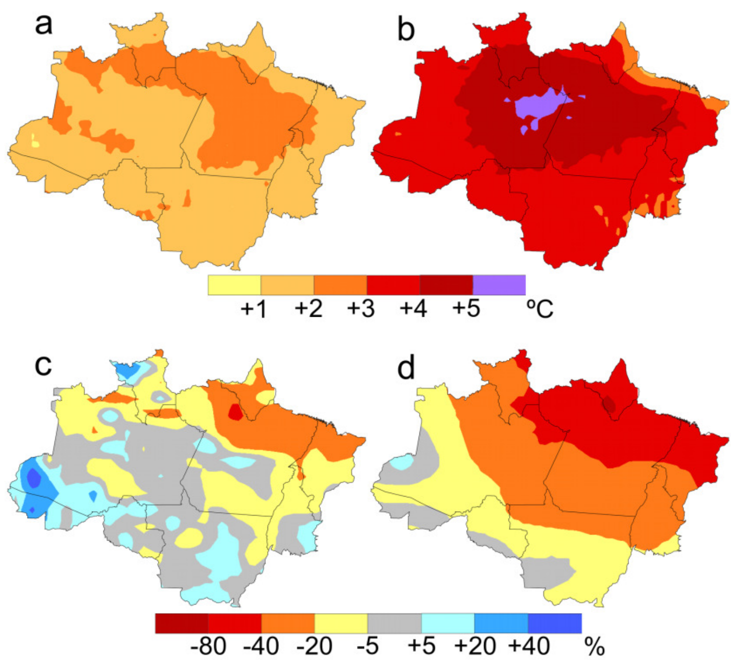


Figure 5.1: Anomalies of temperature (**a**, **b**) and precipitation (**c**, **d**) in a 'moderate' climate change scenario (**a**, **c**; NCAR-CCSM3 model) and in a 'severe' climate change scenario (**b**, **d**; UKMO-HadCM3 model) projected for 2036-2065 under SRES-A2 compared to 1961-1990 in the Brazilian Legal Amazon.

Recent extreme climate events, like the 1997-1998 El Niño drought [*Nepstad et al*. 1999a] or the 2005 drought [*Marengo et al*. 2008], brought considerable reductions in crop/pasture productivity and food shortage, among a variety of other relevant impacts in- and outside the Amazon [*Nepstad et al*. 1999b, 2004; *Moran et al*. 2006; *Brondizio and Moran* 2008; *Lenton et al*. 2009]. Modeling studies by *Cox et al*. [2000, 2004, 2008] project a future in which the Amazon would be exhibited in a permanent El Niño-like climate after 2040, and that events like the 2005 drought will increase in frequency from a 1-in-20 yr event to a 16-in-20 yr event by 2050. Therefore, in light of the impact extreme climate events had on agriculture in the past, and considering that these events might come close to the future "norm" [*Battisti and Naylor* 2009], the impacts of future climate change on land-use and land-cover change are highly relevant [see *Lambin and Geist* 2006, p. 174].

Agricultural activities are now solidly established in the Brazilian Amazon [*Nepstad et al*. 2006], especially the lucrative soybean farming, which had an increase in area from 16,000 km^2 in 1990 to 60,000 km^2 in 2008 [*IGBE* 2010]. Nearly 36% of the Brazilian cattle herd and pasture area is currently located in the Legal Amazon[7], the only region in the country that has experienced an increase in pasture area in the last two decades [*IBGE* 2010; *Barreto et al*. 2008]. Moreover, the Legal Amazon currently contributes for 15% of the national agricultural gross domestic product (GDP) and had a (total) GDP growth of 6.6% yr^{-1} in the 1999-2008 period, compared to the national average of 3.4% yr^{-1} [*Tomazela* 2007; *Salomon* 2008]. On the other hand, this surge of the Amazon economy was accompanied by increasing conservation concerns. For example, more than 75% of the area under strict protection in the Brazilian Amazon has been enacted after 1990 [*ISA* 2007], and since 2002 the protected area network has increased by 6,400 km^2, covering today 51% of the remaining forest [*Soares-Filho et al*. 2009a]. In 2008 the Brazilian Government made a formal announcement within the United Nations climate treaty framework of reducing Amazon deforestation by 80% compared to the historical rate of 19,500 km^2/yr by 2020 [*Government of Brazil* 2008; *Nepstad et al*. 2009]. The interplay between these two apparently antagonistic issues (high growth of agricultural economy and increased environmental concerns) in view of future climate change and growing demands for land (for food, feed and biofuel production) calls for in-depth scientific research to provide a sound foundation for decision making.

In this study we applied a spatially explicit modeling framework to assess the impacts of climate change on land-use and land-cover changes (LUCC) in the Legal Amazon by 2050, taking into account projected levels of crop and livestock production. In this study we focus on climate change effects on LUCC via crop/pasture productivity. Two different scenarios of climate change are used, namely moderate and extreme regional climate change. Additionally, we also investigate how 2050 crop and livestock production demands could be reconciled with the end of

[7] The Brazilian states of Acre, Amapá, Amazonas, Mato Grosso, Rondonia, Roraima, Tocantins and (part of) Maranhão. It comprises 61% of the national territory, roughly 62% of the Amazon forest area [*Soares-Filho et al*. 2006] and has a population of 23 million people [*IBGE* 2010].

deforestation in the Brazilian Amazon forest and Cerrado savanna in the 2020's [*Nepstad et al*. 2009].

5.2 Methods

5.2.1 Initial Land-Use Maps

Land-use maps for the years 2001 and 2006 are used as initial boundary condition in the modeling framework described below, and are also employed in model calibration and evaluation (the latter only applying fractions of the maps). In summary, two land-cover maps of the Legal Amazon (based on *PRODES* [2009] data) for the mentioned years were divided into 32 socio-economical regions, as suggested by *Garcia et al*. [2004] and *Soares-Filho et al*. [2006]. Each subregion had its crop and pasture area determined from IBGE[8] statistics [*IBGE* 2010]. Fourteen different crop types, two types of pastures (well-managed and poorly-managed, hereafter pm-pasture and wm-pasture), urban and abandoned land (see Appendix C) were assigned to grid cells designated either as deforested or as Cerrado savanna in the original land-cover maps (Cerrado was depicted in its pre-Columbian extension in the original maps). Pm-pastures are meant to overcome a probable underestimation of the area of pastures in the Legal Amazon, and represent pastures with a lower intensity of use, with lower livestock density (compared to wm-pasture), mixed with degraded vegetation [see *Ramankutty et al*. 2008; and Appendix C for discussion on this issue]. Spatial distribution of these land-use types was determined based on a map of the year 2000 of the geographical distribution of crops [*Monfreda et al*. 2008] and pasture [*Ramankutty et al*. 2008]. Crops had priority over pasture for occupation of grid cells, while only one dominant land-use type could occur in one grid cell. Urban areas were assigned to those grid cells having a population density higher than 2000 cap km^{-2} [*Erb et al*. 2007], using the HYDE map of population distribution [*Goldewijk* 2005] in both time steps. The land-use map for the year 2006 is shown in Figure 5.2. The whole process for deriving these maps is described in the Appendix C.

5.2.2 Model Description

The central feature of our modeling framework is the LandSHIFT model, which simulates land-use and land-cover change on a 5 arc-minutes spatial resolution [*Schaldach and Koch* 2009]. By using a "land-use systems" approach it describes the interplay between anthropogenic and environmental system components as drivers for land-use change in three major land-use activities (settlement, crop cultivation, and grazing) and their competition for land resources. Moreover, LandSHIFT's livestock module simulates not only the occurrence of pastures, but also the intensity of grazing. The model has been applied and evaluated in assessments of the impact

[8] Brazilian Institute for Geography and Statistics, responsible for the official statistics on agricultural area and production.

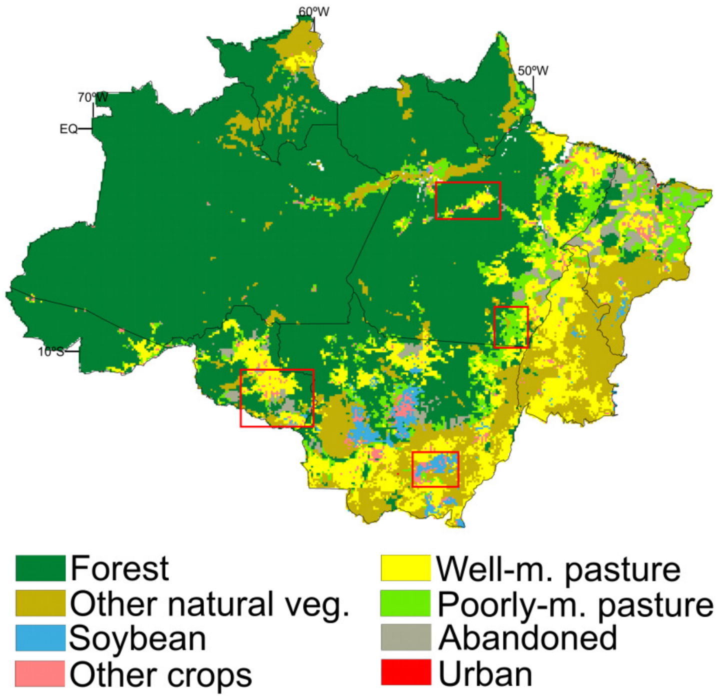


Figure 5.2: Land-use and land-cover map of the Legal Amazon in 2006. Red squares show regions used in the evaluation of model performance. m.: managed.

of grazing management in the Jordan River region [*Koch et al*. 2008], the quantification of future LUCC and water use by agriculture in Africa [*Alcamo et al*. 2010; *Weiß et al*. 2009], and LUCC associated with increased production of biofuels in Brazil (chapter 3) and India [*Schaldach et al*. 2010a].

The International Model for Policy Analysis of Agricultural Commodities and Trade (IMPACT) [*Rosegrant et al*. 2008] calculates future projections of crop/livestock production in the Amazon, and the IFs model [*Hughes* 1999] projects population growth. Because the latter projects population growth on country level, we assumed the Legal Amazon-to-entire Brazil population ratio of 0.12 in 2007 to be constant until 2050. The LPJ dynamic global vegetation model for managed Lands (LPJmL) is used to calculate crop and grassland potential productivity on a 0.5º resolution grid [*Bondeau et al*. 2007]. These three models (IMPACT, IFs, LPJmL) provide inputs to LandSHIFT, even though they are not dynamically coupled to LandSHIFT.

Starting from an initial land-use map (see section 5.2.1), the spatial allocation of different land uses in subsequent time steps is based on a multi-criteria suitability analysis following the equation:

$$\psi_k = \underbrace{\sum_{i=1}^{n} w_i \; p_{i,k}}_{suitability} \times \underbrace{\prod_{j=1}^{m} c_{j,k}}_{constraints} \qquad , \; with \;\; \sum_i w_i = 1, \;\; and \;\;\; p_{i,k}, c_{j,k} \in \; [0, \; 1] \tag{5.1}$$

where the factor-weight w_i determines the importance that each suitability factor p_i has at grid cell k, while c_j represents possible constraints for changing the land-use type at that given cell. In this study p_i includes slope, distance to paved roads, distance to all roads, vegetation type [see *Soares-Filho et al*. 2006 for sources], potential crop/grassland yield (from the LPJmL model), proximity to cropland, attraction to national markets (see below) and distance to deforested land. The latter is used only for grazing, since it has the same effect as proximity to cropland in crop cultivation. Therefore n = 7 for crop cultivation and n = 8 for grazing. Paved roads are updated following the road paving schedule in the study by *Soares-Filho et al*. [2006]. Secondary roads are updated using the outputs of the road constructor submodel of the "SimAmazonia 1" model of land-cover changes [*Soares-Filho et al*. 2006] under a 'business-as-usual' scenario which is consistent with the paving schedule aforementioned. The factor 'attraction to national markets' represents the influence of the Brazilian cities that are the biggest consumers of Amazonian agricultural products, especially meat. These cities are located in Southeast and Northeast Brazil [*Barreto et al*. 2008]. The index is calculated using a unidirectional gravity-type model:

$$NMa_k = \sum_v \frac{Pop_v}{d_k^2} \tag{5.2}$$

Where NMa_k is the national markets attraction exerted in the grid cell k, determined by summing up the population of the five most populous cities in Southeast and Northeast Brazil (v = 5: São Paulo, Rio de Janeiro, Belo Horizonte, Salvador, Fortaleza) weighted by the distance of these cities to the cell k. Soil type is not considered as a p_i factor due to its spatial correlation with the factor 'crop/grass productivity' and 'vegetation type'.

Weights w_i were determined by using the analytic hierarchy process (AHP) test [*Saaty* 1980]. Determination of the relative importance of each p_i factor in relation to the others (RI_{AHP}), used as an entry to the AHP test, was determined by the normalized difference between the average of p_i over areas with and without land-use changes (ε_i):

$$\varepsilon_i = \begin{cases} \dfrac{\alpha_i}{\lambda_i} & \alpha_i > \lambda_i \\ \dfrac{\lambda_i}{\alpha_i} & \alpha_i < \lambda_i \end{cases} \qquad , \; with \; \varepsilon_i \in \; [1, \infty] \tag{5.3}$$

Being α_i the average value of variable p_i in the grid cells where land-use change has occurred in the 2001-2006 period, and λ_i the average value of variable p_i in the grid cells of the 2001 map where land-use change has not occurred (excluding the land-use activity in question, e.g. crops).

Therefore, the higher the ε_i value, the higher the difference between the α_i and λ_i averages and the importance of that p_i factor. RI_{AHP} is then determined with a pairwise comparison of ε_i from all p_i

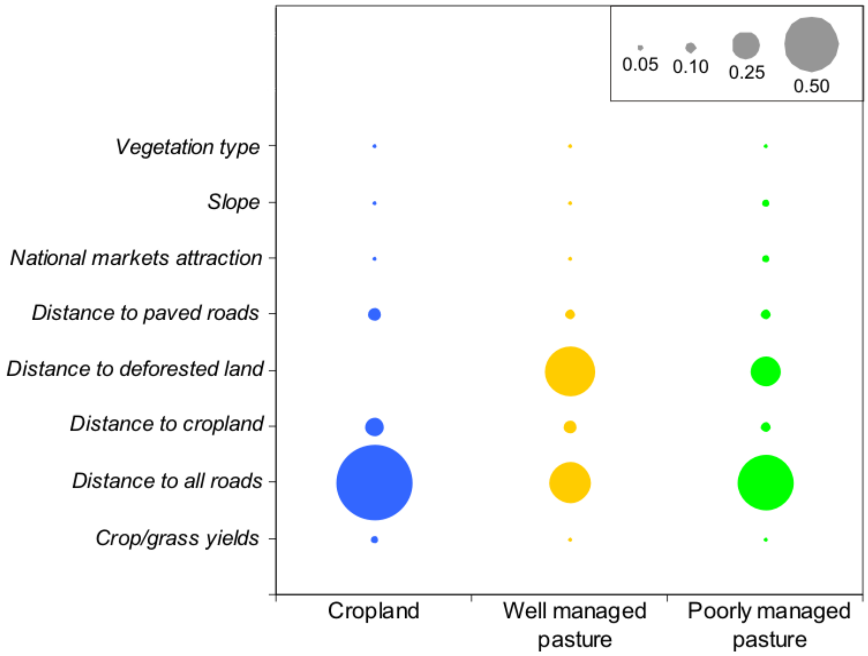


Figure 5.3: Weights w_i given to each p_i factor used in the LandSHIFT model for each of the three major land-use activities in this study. The sum of w_i for all p_i factors of a given land-use activity equals 1. The factor 'distance to deforested land' is not used for cropland allocation.

factors. The procedure was repeated for the three major land-use activities considered here: crop cultivation, well-managed grazing, and poorly-managed grazing (Figure 5.3). Overall, this procedure showed that 'distance to roads' and 'distance to previously deforested areas' are the most important variables for explaining current patterns of land-use change, in agreement with the analysis by *Soares-Filho et al.* [2006]. However, other variables contribute as well to explain the different land-use activities. For example, slope has a higher importance for the location of pm-pastures than for other land-use activities. Interestingly, crop/grass productivity does not play an important role for the location of croplands and pastures.

Constraints c_j comprise conservation areas and land-use transition. The level of constraint for each category of conservation area (strict protection = 0.19, sustainable use = 0.66, indigenous reserve = 0.54, military reserve = 0.01, not protected = 1.0) was derived from the analyses by *Soares-Filho et al.* [2006, 2009a]. In this study the land-use transition constraints all have a value of 1.0 (i.e., no constraint), except the conversion from urban to other land use which has a value of 0.0. The transition from forest to soybean is reduced to 0.1 after 2006 to simulate the soybean moratorium introduced in that year, which almost completely stopped deforestation directly caused by soybean [*ABIOVE* 2009].

The allocation algorithm assumes that crop cultivation takes place generally, but not always, in the most suitable cells for each crop/pasture type and calculates a "quasi-optimum" spatial crop distribution. The Multi-Objective Land Allocation (MOLA) heuristic used here seeks pattern stability and keeps previous land uses even if another crop/pasture type has a higher suitability in that cell. LPJmL potential yields are applied a crop-specific factor to match current crop yields with statistics from the study area [*Schaldach and Koch* 2009; *IBGE* 2010]. These factors, which are calculated at the first simulation time step, account for uncertainties due to crop management, (e.g., multi-cropping), or discrepancies due to the aggregation of crop types to LPJmL crop functional types (e.g., LPJmL pulses represent extra-tropical pulses such as lentils). Crop production of a given grid cell k is defined as the potential crop yield at k multiplied by the area of k that is not covered by settlement.

Allocation of both types of pasture depends on the potential productivity of grass in the grid cells, based on a livestock feed supply-demand logic. Forage supply is calculated by summing up the grass productivity of every pasture cell multiplied by the fraction of biomass that is utilized by livestock (grazing efficiency g_e) $g_e = 0.37$ in wm-pastures and $g_e = 0.12$ in pm-pastures, meaning that wm-pastures have a higher carrying capacity than pm-pastures. These values of g_e are based on literature [*Rueda et al*. 2003; *Camarão et al*. 2000] and calibration (only in terms of total pasture area) against the initial land-use maps. Forage demand is determined by the multiplication of the total livestock herd by the average forage consumption per livestock unit (10 kg of dry matter d^{-1}; *Krausmann et al*. [2008]). In this study the word livestock refers to bovine species such as cattle and buffaloes, which represent by far the majority of the grazing livestock herd in the Legal Amazon. By overlaying the initial land-use map mentioned above and the map of livestock density by *FAO* [2007b], we estimated that approximately 14% of the Amazonian livestock herd is located in pm-pastures. Therefore, in the simulations in which pm-pastures persist in the future (see next section), we assign a constant value of 14% of the total livestock herd to be allocated in pm-pastures (and 86% in wm-pastures). We assume that 95% of the livestock feed demand is fulfilled by forage from pastures, and the rest from feed grains or crop residues [*Krausmann et al*. 2008]. If forage demand is higher than supply, then new pasture cells are allocated, starting from grid cells with the highest suitability for grazing until demand is fulfilled. Average livestock density (Ld) is calculated by dividing the livestock herd by the pasture area. Allocation of land-use activities follows the hierarchical order: settlement, crop cultivation, well-managed grazing, poorly-managed grazing. Only one land-use type can occur in a grid cell.

5.2.3 Input Data and Modeling Protocol

LandSHIFT is initialized with the land use/cover map for the year 2006 (the map of 2001 is used in a model evaluation run), a map of population density [*Goldewijk* 2005], and national statistics of crop production and livestock herd [*IBGE* 2010]. Socio-economic projections include future demands for food [*Rosegrant et al*. 2008] and population growth [*Hughes* 1999] under a baseline scenario. The human population in the study area increases from 24.2 million people in 2006 to 32.6 million people in 2050, representing an average annual growth of 0.8% per year. Brazil GDP increases from 954 $x10^9$ USD[9] in 2006 to 7,226 $x10^9$ USD in 2050, with an average growth rate of 4.42% yr^{-1}, which is comparable to those projected by the Intergovernmental Panel on Climate Change's (IPCC) Special Report on Emission Scenarios (SRES) A2 and A1 for Latin America (3.8 and 5.5% yr^{-1} respectively) [*Nakicenovic et al.* 2000]. The projections of IMPACT consider price effects that come from dynamics on both the supply and demand side of food and feed commodities. Crop production increases by 93% in total, with soybean production increasing by 11% (Table 5.1). The livestock herd of Legal Amazon grows from 75.7 million head in 2006 to 152.9 million head in 2050, with an average increase of 2.3% per year. This growth rate is far below the average growth of 9.3% yr^{-1} observed in the 1974-2006 period in the region [*IBGE* 2010], and reflects the effect of livestock's own price and the price of competing commodities in the future. Moreover, it would be too difficult to sustain the high growth observed in the last 30 yr in the long-term, considering that recent statistics show that this livestock growth rate is reducing with years (e.g., 6.1% yr^{-1} in the 1990-2006 period) [*Barreto et al.* 2008].

Potential crop/grass yields were calculated with the LPJmL model, which simulates global terrestrial vegetation dynamics, agricultural productivity, and the associated carbon and water cycles in a 0.5º spatial resolution [*Sitch et al.* 2003; *Gerten et al.* 2004; *Bondeau et al.* 2007]. Model calculations are based on physiological processes such as photosynthesis, autotrophic respiration, evapotranspiration, effects of soil moisture and drought stress, as well as on plant's functional and allometric rules, phenology and growth parameterizations. Full model description as well as extensive validation against observed data of sowing dates, fraction of

Table 5.1: Crop production in 2006 [*IBGE* 2010], projection for 2050 [*Rosegrant et al.* 2008] and 2006-2050 changes in the Legal Amazon.

	2006	**2050**	**Δ 2006-2050**
	Gg	**Gg**	**%**
Rice	2392	2138	-11
Maize	5757	9944	+73
Other tropical cereals	294	951	+224
Pulses	213	685	+222
Tropical roots and tubers	9591	18521	+93
Annual oil crops (excl. soybean)	34	62	+84
Soybean	17788	19692	+11
Sugarcane	17146	47130	+175
Other crops*	5567	14070	+153
Total	58781	113196	+93

* Permanent oil crops, fruits, vegetables, fiber crops, coffee, cocoa, and other stimulants

[9] United States Dollar (2000 value)

photosynthetically active absorbed radiation, seasonal CO_2 flux exchanges, and crop yields can be found in *Bondeau et al*. [2007].

Crop yields for the 1990's, used as baseline yields in LandSHIFT, were calculated using CRU-TS2.1 climate data set, a monthly climatology of meteorological variables, and atmospheric CO_2 concentration for the 1901-2003 period [*Österle et al*. 2003; *Keeling and Whorf* 2005]. LPJmL transient simulations are preceded by a 1000-year spin-up period during which the first 30 years of the climate data set are repeated cyclically in order to bring all carbon pools into equilibrium. Future crop yields (2036-2065 mean) were calculated using the outputs from two IPCC-AR4 general circulation models (GCM), both under the SRES-A2 emission scenario: UKMO-HadCM3 and NCAR-CCSM3 [*Meehl et al*. 2007]. Climate anomalies were defined as the differences from the 1961-1990 mean of the CRU-TS2.1 dataset (Figure 5.1). Besides being among the GCMs that best represent the current climate over the Amazon [*Li et al*. 2006], these two GCMs project highly distinct climatic changes for the 21st century in the Amazon. HadCM3 projects an average increase of 3.8°C and 30% decrease in precipitation over the Legal Amazon in the 2036-2065 period (= severe), while CCSM3 projects a smaller temperature increase of 1.8°C and no changes in average precipitation (= moderate). Atmospheric CO_2 concentration increases from an average 333 ppmv in the 1961-1990 period to 537 ppmv in 2036-2065. The effects of CO_2 fertilization on crop productivity are still poorly understood, especially in the tropics, and seem to be overestimated by most vegetation models currently available including LPJmL [*Slingo et al*. 2005; *Ainsworth and Long* 2005]. Therefore, we consider the upper limit of the effect of climate change on crop/grass productivity to be the HadCM3 climate scenario without the effects of CO_2 fertilization. The lower limit is then considered to be the yields calculated with CCSM3 climate and with the CO_2 fertilization effect. Four scenario variations are modelled with LandSHIFT, all of them with road paving and IMPACT projections on crop and livestock production for 2050:

(i) CCSM3 climate + CO_2 fertilization (hereafter 'moderate-BAU', BAU stands for business-as-usual).

(ii) HadCM3 climate, no CO_2 fertilization ('severe-BAU').

(iii) CCSM3 climate + CO_2 fertilization, suppression of pm-pastures, deforestation of the Amazon (Cerrado) gradually reduced to zero until 2020 (2025) (hereafter 'moderate-CONSERV').

(iv) HadCM3 climate, no CO_2 fertilization, suppression of pm-pastures, deforestation of the Amazon (Cerrado) gradually reduced to zero until 2020 (2025) ('severe-CONSERV').

Pm-pastures are gradually replaced by wm-pastures until 2025 in the variations in which a suppression of pm-pastures is assumed. The intensification of grazing needed to meet the feed-demands of future livestock production in variations 'moderate-CONSERV' and 'severe-CONSERV' is determined by increasing the grazing efficiency (g_e) factor to the level at which demands are met, though keeping g_e below the maximum reported value of 0.7 [*Difante et al*. 2009].

5.2.4 Model Evaluation

LandSHIFT has been thoroughly evaluated in terms of the quantity of change in other studies [*Koch et al*. 2008; *Schaldach et al*. 2010a; *Alcamo et al*. 2010], including a study in which the model was applied for entire Brazil (chapter 3). However, the model has not been consistently evaluated in terms of the location of changes mainly due to the lack of independent time series of "observed" land-use maps generated based on the same methodology. Therefore, taking advantage of the two independent maps of land use employed here (see section 5.2.1), a LandSHIFT run from 2001 to 2006 is performed to evaluate the model performance. The model is initialized with the 2001 land-use map of the Legal Amazon and is driven with reported statistics on crop and livestock production for 2006 [*IBGE* 2010]. Since the 2006 map inherits the spatial pattern of the 2001 map we assess the spatial fit only between the maps of changes. Thus, the resulting modeled map of LUCC from 2001 to 2006 is compared with the observed map of changes for that period. In order to reduce the dependency between the datasets used for comparison (the observed maps were used for deriving the w_i weights of LandSHIFT), the evaluation is done only in four selected regions of the Legal Amazon (Figure 5.2). These regions were selected as to cover locations that experienced pronounced deforestation or other LUCC encompassing the three major land-use categories considered here (cropland, wm-pasture, and pm-pasture) in the 2001-2006 period. The four regions and the dominant land-use transitions that were observed from 2001 to 2006 are: Central Pará (forest to pm-pasture, to wm-pasture and to cropland), Southeast Pará (forest to pm-pasture), South Mato Grosso (Cerrado to cropland), South Rondônia (pm-pasture to wm-pasture, forest to cropland). Combined, these four regions represent only 10% of the area that experienced LUCC in the 2001-2006 period, and ~4% of the Legal Amazon.

Both observed and modeled 2001-2006 LUCC maps were reclassified into 3 categories for the comparison: natural vegetation, cropland, pasture. Conversion from any land-use to natural vegetation is excluded from our analysis since LandSHIFT does not simulate natural vegetation regrowth. The maps of changes were subject to the Fuzzy vicinity-based comparison method developed by *Hagen* [2003] (K-Fuzzy method) and modified by *Soares-Filho et al*. [2009b]. This method takes into account the nature of LUCC models to justify a vicinity-based comparison (i.e., LUCC location is "fuzzy"). An exponential decay function is employed to weigh the distance of a cell in one map to its counterpart in the second map. Map comparison is carried out in a two-way manner and at multiple spatial resolutions. However, only the minimum similarity value is used in order to avoid an artificially high fit which is characteristic of univocal comparison of random maps. Figure 5.4 shows the results of this Fuzzy comparison analysis over the four evaluation regions. The model does a reasonable job in capturing the right location of transitions, as the average curve reaches up to 60% of similarity with a search radius of only 2 grid cells, and

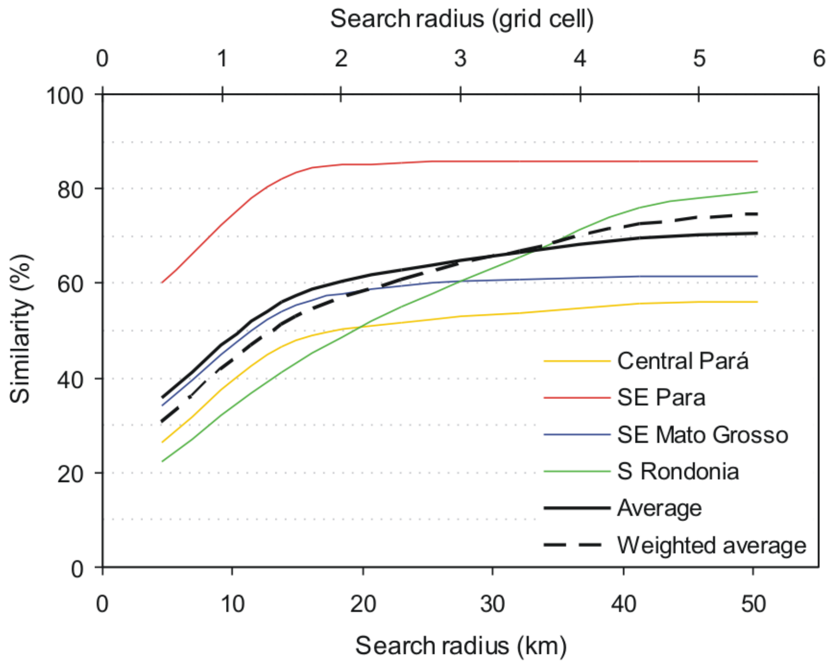


Figure 5.4: Fuzzy similarity [*Hagen* 2003; *Soares-Filho et al*. 2009b] between observed and modeled (LandSHIFT) maps of land-use and land-cover change in 2001-2006 at four selected regions of the Legal Amazon (see Figure 5.2 for location map) and the average of them. The dashed line shows the average weighted by the area of each of the four analyzed regions.

peaks in 71% after 5 grid cells. If the average is weighted by the size of each of the 4 analyzed regions, then similarity reaches the value of 60% after 3 grid cells, but peaks in a higher value of 75% after 5 grid cells. From the original kappa classification [*Monserud and Leemans* 1992], a 60% similarity is classified as a "good" degree of agreement. Lowest similarity is found in Central Pará, because the model does not capture well the transition from forest to wm-pasture. On the other hand, the highest fit is found in Southeast Pará since the model simulates correctly the forest to pm-pasture transition, which, according to the maps presented in section 5.2.1, responded for 56% of the Amazon deforestation in the 2001-2006 period.

Cropland is overestimated by 8%, like in the study presented in chapter 3 for entire Brazil. The area of pastures was calibrated with the g_e factor, therefore its fit to the observed data is nearly perfect. Average livestock density in wm-pastures and pm-pastures in 2006 is 1.11 head ha^{-1} and 0.36 head ha^{-1} respectively (average = 0.74 head ha^{-1}). The modeled rate of Amazon deforestation for 2001-2006 is underestimated by 11%: 20,851 km^2 yr^{-1} versus 18,653 km^2 yr^{-1} [*PRODES* 2009]. This underestimation is because LandSHIFT does not simulate the direct transition from forest to abandoned land, as is the case in some areas of the observed maps (a forestry module is currently being developed in LandSHIFT and could account for this kind of land-use transition in the future). Moreover, one should also consider that 2001-2006 was a period with above-average deforestation rate. For example, average deforestation rate was 18,700 km^2 yr^{-1} in the 1996-2000 period, and 10,833 km^2 yr^{-1} in 2007-2009. Deforestation of Cerrado is underestimated by 18%: 6,366 km^2 yr^{-1} versus 5,206 km^2 yr^{-1}. Nevertheless, there is high uncertainty associated with deforestation rates of the Cerrado since land cover changes in the Cerrado are much more difficult to be detected by remote sensors than in the Amazon (Appendix C).

5.3 Results

5.3.1 Potential Yields

Figure 5.5 shows the simulated changes in crop/grass yields relative to their values in the 1990's. Average (between all crop/grass types) yield changes range from -11% with HadCM3 climate to +14% with CCSM3 climate when the CO_2 fertilization effect is considered. However crop yields are -31% (HadCM3) to -8% (CCSM3) lower compared to the 1990's if we consider that the CO_2 fertilization effect will have no influence on future crop yields. The reductions by 44% and 10% in the yields of soybean and grassland respectively under the 'severe-BAU' scenario are particularly relevant for the Amazon region (besides considerable reduction in the yields of maize, rice and other crops under that scenario). Soybean yield decreases by 1.8% and grass yield increases by 4.5% in 'moderate-BAU' scenario. Tropical roots functional type (cassava) is the only crop that experiences an increase of yields in every scenario since, in LPJmL, this crop type benefits from the increase in temperature. In general, the most pronounced yield reductions are found in the northern portion of the Legal Amazon, since both HadCM3 and CCSM3 climate model project reductions in precipitation in that region (Figure 5.1).

5.3.2 Land Use Change with 'Business-As-Usual'

Under a moderate climate change scenario (and ignoring the target of halting deforestation in the Amazon) deforestation of the Brazilian Amazon would amount to 928,000 km^2 by 2050 (20,173 km^2 yr^{-1} in the 2006-2050 period) in our simulations (Table 5.2). On the other hand, under a severe climate change scenario (severe-BAU) the forest would be reduced by 1,109,000 km^2 (24,108 km^2 yr^{-1}). Therefore, in our simulations deforestation would be 20% higher under severe climate change compared to a scenario of moderate climate change. The Amazon deforestation scenarios simulated by LandSHIFT in the two 'BAU' scenarios lie in between the total deforestation modeled by the "SimAmazonia 1" model of land-cover changes under the scenarios "business-as-usual with creation of new protected areas" and "governance without creation of new protected areas" [*Soares-Filho et al.* 2006]. Deforestation of the Cerrado simulated by LandSHIFT would amount to 88,000 km^2 (1,913 km^2 yr^{-1}) and 328,000 (7,130 km^2 yr^{-1}) under the moderate-BAU and severe-BAU climate change scenarios respectively. Thus in our simulations

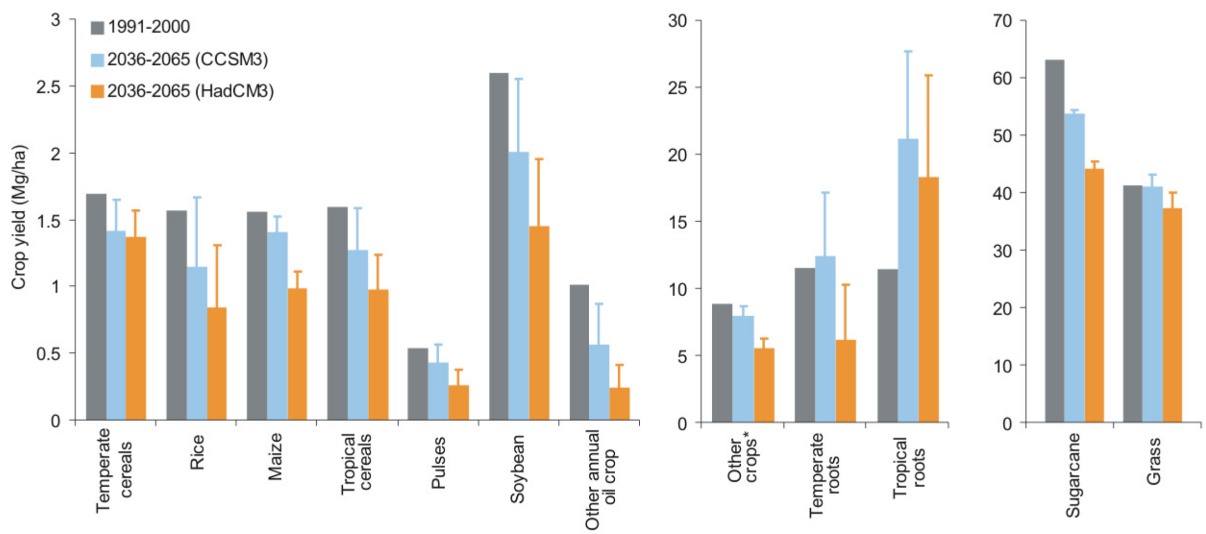


Figure 5.5: Crop and grass average yield (fresh matter) observed in the 1990's and projections for the 2036-2065 period under a "moderate" climate change scenario (NCAR-CCSM3, SRES-A2) and under a "severe" climate change scenario (UKMO-HadCM3, SRES-A2) in the Legal Amazon. Top whiskers denote projections in which the CO_2 fertilization effect influences crop/grass yields. Grass (pasture) comprises predominantly C_4 grass. *Permanent oil crops, fruits, vegetables, stimulants (average yield).

Table 5.2: Land-use and land-use change (relative to 2006) according to different modeled scenarios in the Legal Amazon in 2050. Here “cropland” includes soybean. Urban areas increase from 3,151 km^2 in 2006 to 3,949 km^2 in 2050. nat. veg.: natural vegetation; wm: well managed; pm: poorly managed.

	Forest	Other nat. veg.	Cropland	Soybean	Abandoned	Wm-pasture	Pm-pasture	Livestock density (wm-pasture)	Livestock density (pm-pasture)
Year and scenario*	x1000 km^2	x1000 km^2	x1000 km^2	x1000 km^2	x1000 km^2	x1000 km^2	x1000 km^2	head/ha	head/ha
2006, -	3336	648	135	59	110	585	294	1.11	0.36
2050, moderate-BAU	2408	560	208	63	126	1200	605	1.10	0.35
2050, severe-BAU	2227	320	302	94	119	1423	717	0.92	0.30
2050, moderate-CONSERV	3307	527	209	63	2	1063	0	see Table 3	-
2050, severe-CONSERV	3320	447	303	93	2	1036	0	see Table 3	-

* moderate-BAU: CCSM3 climate + CO_2 fertilization; severe-BAU: HadCM3 climate, no CO_2 fertilization; moderate-CONSERV: CCSM3 climate + CO_2 fertilization, suppression of pm-pastures, deforestation of the Amazon (Cerrado) gradually reduced to zero until 2020 (2025); severe-CONVSERV: HadCM3 climate, no CO_2 fertilization, suppression of pm-pastures, deforestation of the Amazon (Cerrado) gradually reduced to zero until 2020 (2025).

deforestation of the Cerrado would be 273% higher with severe climate change compared to a scenario with moderate climate change.

Altogether, crops would need a 45% larger area under the severe scenario compared to the moderate scenario to meet the 2050 demands projected by the IMPACT model for the Amazon. Soybean alone would occupy a 49% larger area in the severe scenario compared to the moderate scenario (94,000 km^2 versus 63,000 km^2). Wm-pasture (pm-pasture) would have its area increased by 615,000 km^2 (311,000 km^2) in the moderate scenario and by 838,000 km^2 (423,000 km^2) in the severe scenario. Difference in total area of both types of pastures between the climate scenarios would be of 18%. Ld decreases in both climate scenarios, although this decrease is more pronounced in the severe climate change scenario. That is because even though average grass productivity increases from 2006 to 2050 in the moderate-BAU, it decreases punctually in the regions where new pastures are established until 2050, north and northeast Legal Amazon (see Figure 5.1). Abandoned area increases by 16,000 km^2 and 9,000 km^2 in the moderate and severe climate change scenarios respectively (Table 5.2) due to a shift in cropland location driven by local climate change (e.g. reduction in precipitation in western Mato Grosso with CCSM3 causes some soybean fields to shift to southeastern Mato Grosso).

Most of the deforestation would still occur in the southern and eastern Amazon, and along the highways to be paved (Figure 5.6). Cropland expansion would take place mostly in Mato Grosso and Tocantins. Wm-pastures would be widespread along the deforestation arc. Because pm-pasture is the last in the hierarchical allocation of major land-use activities in LandSHIFT, this land-use type is relegated to more remote and less productive areas. Figure 5.7 shows how different climate change scenarios could result in distinct LUCC pattern, impacting both the extent and location of future LUCC. Pastures expand deeper into the western Amazon forest and elsewhere in the Cerrado in the severe scenario compared to the moderate one, owing to the pronounced decrease in precipitation projected by HadCM3 in northern Legal Amazon.

5.3.3 Land Use Change with the End of Deforestation

From 2006 to 2050 Amazon deforestation would amount to 29,000 km^2 (2,230 km^2 yr^{-1}) and 16,000 km^2 (1,230 km^2 yr^{-1}) respectively under moderate-CONSERV and severe-CONSERV scenarios, before it ends by 2020 (Table 5.2). On the other hand, the Cerrado loses 121,000 km^2 (9,307 km^2 yr^{-1}) and 201,000 km2 (15,461 km^2 yr^{-1}) of its native vegetation by 2025 in the moderate-CONSERV and severe-CONSERV scenarios respectively. Halting Amazon deforestation in 2020 but still allowing deforestation of the Cerrado until 2025 explains these highly different deforestation rates when compared to the 'BAU' scenarios.

Future cropland area would be roughly the same as in the 'BAU' scenarios since crops have priority over pastures in LandSHIFT, i.e. crops are allowed to displace pastures. Therefore the area of pasture would increase in approximately 171,000 km^2, by replacing natural vegetation but namely by occupying pm-pastures and abandoned areas (Figure 5.8). Then at this point two

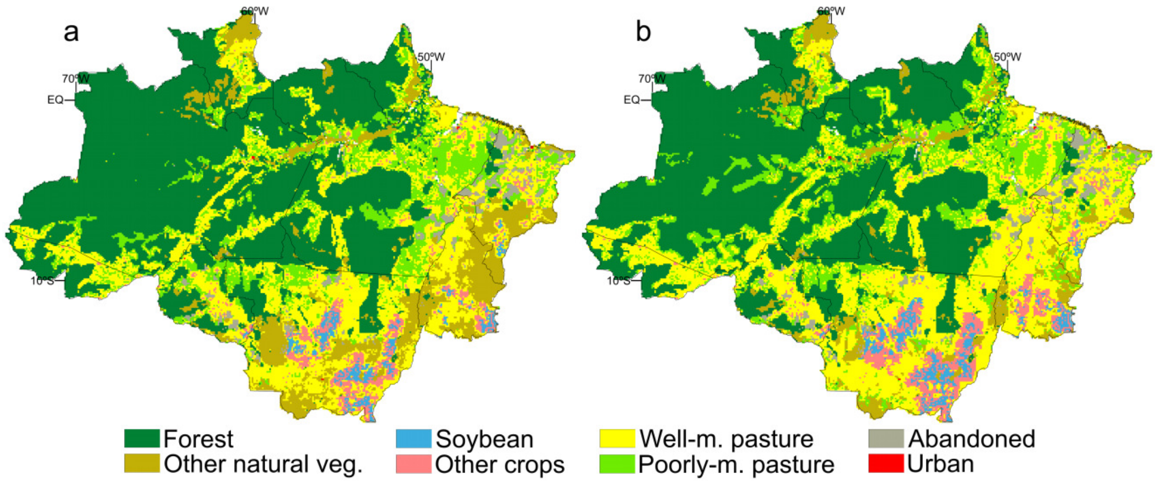


Figure 5.6: Modeled land use and land cover in the Legal Amazon in 2050 under 'moderate' (**a**, NCAR-CCSM3 SRES-A2 with CO_2 fertilization effect) and 'severe' (**b**, UKMO-HadCM3 SRES-A2 without CO_2 fertilization effect) climate change scenarios. m.: managed.

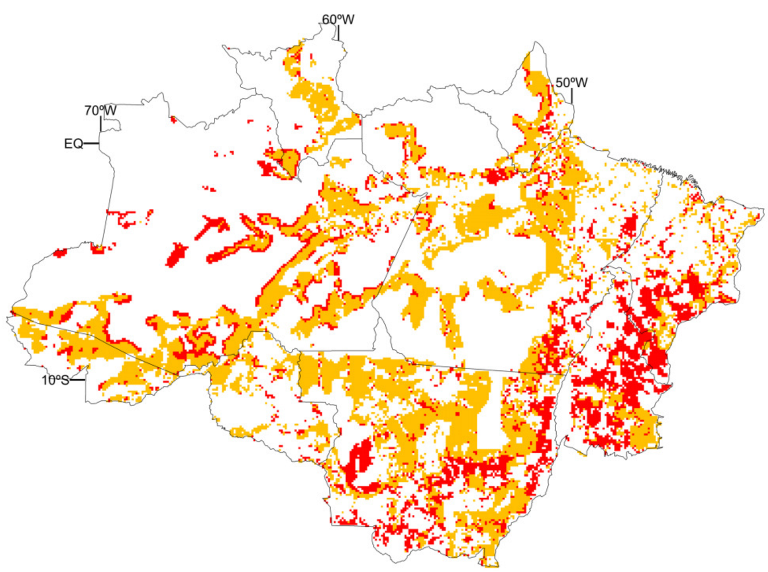


Figure 5.7: Land-use and land-cover change modeled for the 2006-2050 period in the Legal Amazon under 'moderate' (orange; NCAR-CCSM3 SRES-A2 with CO_2 fertilization effect) and 'severe' (orange plus red; UKMO-HadCM3 SRES-A2 without CO_2 fertilization effect) climate change scenarios (see also Figure 5.6).

options are considered here to reconcile land use and conservation targets in the Legal Amazon (Table 5.3): (a) livestock production is reduced, i.e., from 26% (moderate-CONSERV) to 40% (severe-CONSERV) of the Legal Amazon livestock demand for the year 2050 cannot be produced there; (b) livestock production is ensured / kept up with the intensification of livestock in the Legal Amazon. In that case, livestock density needs to roughly double from 0.74 head/ha (average of pm- and wm-pastures) in 2006 to ~1.46 head/ha (1.44 head/ha in moderate-CONSERV and 1.48 head/ha in severe-CONSERV). g_e is increased to 0.47, a value still far from the maximum of 0.7 reported by *Difante et al*. [2009] under rotational stocking management.

Figure 5.8 shows the land-use pattern in 2050 in the 'severe-CONSERV' scenario. Some deforestation of the Amazon is projected in northeast Pará and along the highways BR-163 in Mato Grosso and BR-364 in Acre. Most of the deforestation of the Cerrado takes place in Maranhão and Tocantins in areas that are not specified as conservation units.

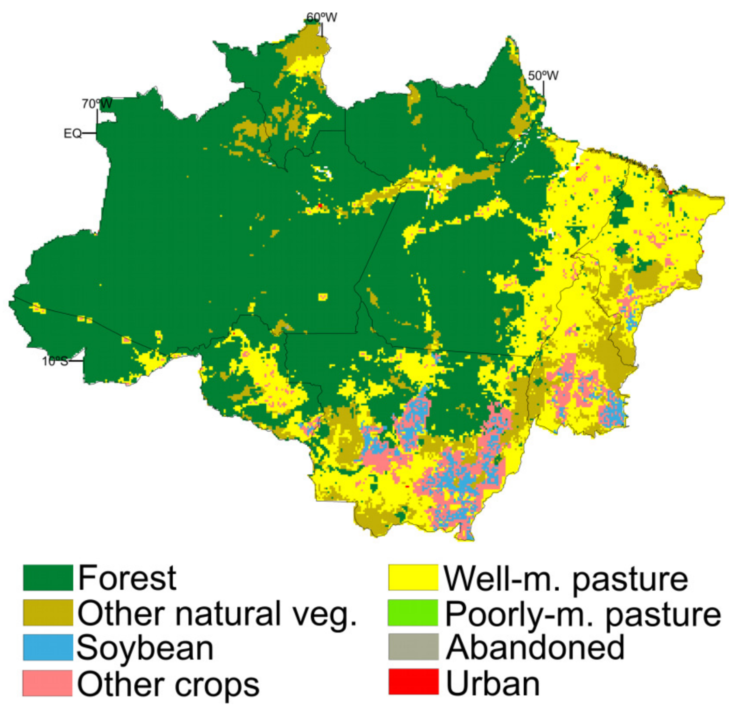


Figure 5.8: Modeled land use and land cover in the Legal Amazon in 2050 under a 'severe' (UKMO-HadCM3 SRES-A2 without CO_2 fertilization effect) climate change scenario, with suppression of poorly-managed pastures and the end of deforestation of the Amazon (Cerrado) by 2020 (2025). m.: managed.

Table 5.3: Livestock production and correspondent livestock density in the Legal Amazon in 2050 with the end of deforestation in the Amazon (Cerrado) by 2020 (2025).

Scenario*	Fulfillment of IMPACT projected livestock production %	Livestock herd 10^6 head	Livestock density head/ha
moderate-CONSERV	74	113.1	1.06
moderate-CONSERV	100	152.9	1.44
severe-CONSERV	60	91.7	0.88
severe-CONSERV	100	152.9	1.48
Average	67	102.4	0.97
Average	100	152.9	1.46

* moderate-CONSERV: CCSM3 climate + CO_2 fertilization, suppression of pm-pastures, deforestation of the Amazon (Cerrado) gradually reduced to zero until 2020 (2025); severe-CONSERV: HadCM3 climate, no CO_2 fertilization, suppression of pm-pastures, deforestation of the Amazon (Cerrado) gradually reduced to zero until 2020 (2025).

5.4 Discussion

From our simulations it can be inferred that in the Legal Amazon: (a) Future climate change may influence LUCC in ways that have previously remained unexplored. Severe climate change in some regions can shift the deforestation frontier. For example, the harsh climate projected by HadCM3 in central and eastern Amazon increases human pressure in the Cerrado and western Amazon; and (b) ambitious conservation targets and increased agricultural production can be reconciled even under a scenario of severe climate change, but it will require either a more intensive use of the land, or a slowdown in the growing production of meat. These two major findings are discussed below.

5.4.1 Climate Change Effects on Land Use

There is now extensive documentation about the impacts of regional or continental extreme climatic events on agriculture and livestock production. Excellent examples are the hot summer of 1972 in the southwest of former Soviet Union and its consequences in world cereal markets [*Dronin and Bellinger* 2005], the record yield drops and livestock stress in Europe during the anomalous heat in the summer of 2003 [*De Bono et al*. 2004], the 2005 drought in the Amazon and associated agricultural losses in many parts of Brazil [*Lenton et al*. 2009], and several studies on the impacts of the El Niño / Southern Oscillation on crop/pasture productivity and food security in the Amazon [*Moran et al*. 2006; *Brondizio and Moran* 2008], in Indonesia [*Keil et al*. 2008], or worldwide [*Ferris* 1999]. Nevertheless, currently such climatic events have a relatively long return interval (El Niño: ~7 yr [*Cobb et al*. 2003]; 2005-like drought: 20 yr [*Cox et al*. 2008]) and are not yet the climatic “norm”, as for example would be the case of the permanent El

Niño events projected by HadCM3. One of the single documented references to recent long-term climate change and its effects on yields and LUCC is the prolonged drier conditions found in the Sahel from the late 1960s until the early 1990s which caused the abandonment of crop and grazing fields, besides massive migration and countless hunger- and battle-related deaths [*Kandji et al*. 2006; *Burke et al*. 2009]. But in fact most of the examples of long-term climate change impacts on LUCC stem from archeological/historical records, as is the case for the theory of the collapse of the Maya in the Yucatán Peninsula in the late 10th century [*Turner II et al*. 2003] or the effects of the onset of the Little Ice Age (16th century) on the agriculture of the Iberian Peninsula [*Puigdefábregas* 1998].

These catastrophic experiences reveal that the impacts of climate change on LUCC are always, though not solely, mediated by changes in crop/grass productivity, which is the way LUCC is affected by climate in this study. In view of that, we can consider the method used here for assessing the impacts of climate change on LUCC as reasonable, even though it does not consider other ways in which climate change could indirectly affect LUCC in the Amazon. Difficulties for navigation if the level of rivers is too low, decrease of fish stocks (which is one of the main sources of protein of the Amazonians), spread of diseases, potable water shortage and higher frequency of floods: all these examples represent pathways through which climate change could affect farmer's and other people's living conditions and consequently LUCC in the region.

The simulated range of changes in crop/grass productivity lies within the range projected in other studies for Brazil [*Assad and Pinto* 2008; *Lobell et al*. 2008] and the whole globe [*Tebaldi and Lobell* 2008]. That is particularly true for the projections in which the CO_2 fertilization effect does affect crop yields in the future. On the other hand, LPJmL yield projections with HadCM3 climate and no CO_2 fertilization are much lower than what has been projected in the studies mentioned above, but should not be considered as less probable, since the uncertainties regarding the effects of rising CO_2 on future crop yields are still large [*Ainsworth and Long* 2005; *Long et al*. 2006; *Lobell and Field* 2008]. The pronounced decrease in the yields of soybean, slight decrease of maize and rice, as well as the increase of cassava yields are particularly in agreement with the projections by *Assad and Pinto* [2008], using a regional climate model for entire Brazil. Nevertheless, the authors of that study point out for a reduction of up to 25% of pasture productivity (for entire Brazil), compared to the 10% projected with LPJmL-HadCM3 for Legal Amazon.

Technological improvements of yields are, on purpose, not considered in our simulations so one can regard the projections shown in Figure 5.5, especially those calculated with HadCM3 climate, as an outlook on the magnitude of adaptation needed by the Amazon agriculture over the next decades. Although in this study we calculate the LUCC resulting from yield changes with the HadCM3 climate (Figure 5.6), we believe it is unlikely that in reality crop cultivation would continue after such a reduction of yields, especially in large-scale farming. So, for example, to avoid the soybean yield reduction caused by an extreme climate change scenario (i.e. to keep soybean yields at their current values at least) a yield increment rate of 23 kg ha^{-1} yr^{-1} would be needed until 2050, which is far lower than the soybean yield enhancement rate of 39 kg ha^{-1} yr^{-1}

observed in the last two decades in Brazil [*FAO* 2010; see also chapter 3]. For maize this yield adaptation would be 11 kg ha^{-1} yr^{-1}, compared to 78 kg ha^{-1} yr^{-1} yield enhancement observed in the last two decades in Brazil [*FAO* 2010; see also chapter 3]. This adaptation of cropping and livestock systems could come in the form of better management of water resources, change in sowing dates, infrastructure to minimize heat-stress-related reductions of livestock productivity, or even altering the location of cropping/livestock activities [*Howden et al*. 2007]. All these actions would obviously demand financial investments. As a consequence, adaptation seems more feasible to large-scale farmers than for smallholder or subsistence farmers due to the former's easier access to credit. A recent survey revealed that although smallholder agriculture occupies only 24% of the total farmed area in Brazil, it is responsible for 87% of the national production of cassava, 70% of dry beans, 46% of maize, 36% of rice and 58% of milk [*IGBE* 2009]. As presumed from our results, this agricultural production (its share in the Legal Amazon) might be compromised in the future assuming no intervention and/or support from the government or other bodies to develop adaptation strategies for the sector [*Morton* 2007]. Such a strategy should take into account the sociocultural and environmental diversity of the Amazonian small-scale farmers and, importantly, institutionalize the translation of large-scale projections, like the one in this study, into local actions [*Brondizio and Moran* 2008].

5.4.2 The End of Deforestation and Land Use

Our results also show that a combination of ambitious conservation targets [*Nepstad et al*. 2009] with increased agricultural production is feasible even under a scenario of severe climate change. But adaptation of agriculture, especially the intensification of cattle ranching, which is the main land use in Legal Amazon, is a *sine qua non* condition to achieve both targets. Brazil's recent economic growth has boosted people's monetary access to meat, and the country today is the fourth biggest consumer of meat per capita in the world [*Barreto et al*. 2008; *Friends of the Earth* 2009]. Considering these current trends of changes in life style, it seems more likely that the mentioned conservation targets might be achieved via intensification of livestock production rather than via reduction of livestock production and consequent meat consumption.

It is well known that the oxisols and ultisols of the Amazon, dominant in over 75% of the basin, make it difficult to keep a high productivity of pastures for more than ~5 years without active management [*Walker et al*. 2000]. But other factors such as land tenure (e.g., in many cases Ld is kept at a minimum level only to guarantee ownership over public land), and ongoing policies of "perverse" subsidies (e.g. animal acquisition is heavily subsidized in Brazilian cattle ranching but nearly no incentives are provided specifically for the recovery of degraded pastures and intensification of grazing) also have a decisive influence on the widespread low Ld across the Legal Amazon [*Fearnside* 2002; *Nepstad et al*. 2006; *Friends of the Earth* 2009]. As discussed in chapter 3, an increase in livestock density in the Legal Amazon, such as the +0.72 head/ha proposed here, is perfectly possible from a biophysical point of view with the enhancement of grass productivity and adoption of some simple management practices [*FAO* 2007a; *Assad and*

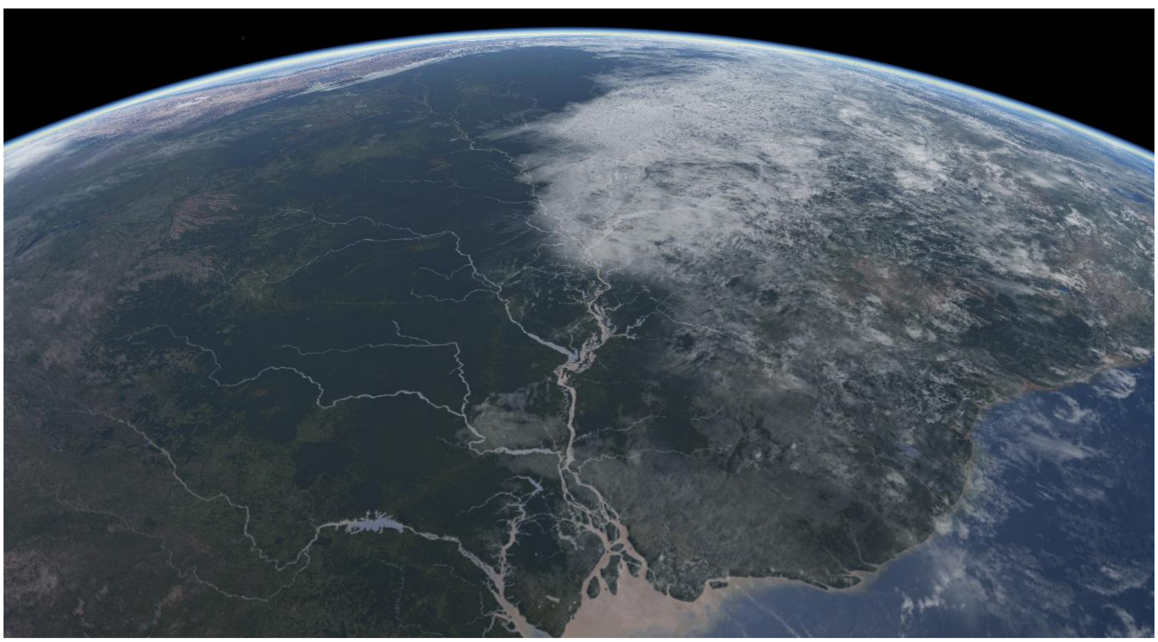
Pinto 2008]. Nevertheless, this intensification seems to be impossible without a concerted effort in terms of providing adequate subsidies [*Friends of the Earth* 2009], increasing land tenure in the region [*Fearnside* 2008], and excluding deforesters from the livestock supply chain [*Nepstad et al*. 2009].

5.4.3 Caveats and Future Research

The main caveat of our simulation is that there are no feedbacks between the models comprised in our framework. For example, it could occur that the agricultural demands projected by the IMPACT model are reduced over time with the establishment of stricter conservation targets (such as those suggested by *Nepstad et al*. [2009]). An improved and fully coupled modeling framework would also help understanding other key questions about the Amazon system. For example, what would be the impacts of a climate-driven forest dieback [*Cox et al*. 2004] on the deforestation rates and land use pattern in the Amazon? How would year-to-year climate variability influence future LUCC and food security? What are the probabilities of the impacts (e.g. assessed with ensemble runs)? These questions remain to be pursued. Overall this study should be regarded as a first indication of the range of impacts that future climate change may potentially have on LUCC, and its relation with conservation strategies. Importantly, it also suggests that both the identification of impacts and the adaptation to them should be tackled in a multidisciplinary and integrated manner, considering conservation strategies, and projections on population growth, changes in life style and agricultural production.

CHAPTER 6

Synthesis



The Amazon as seen from space, with the deforestation frontier evident in its southern border (lefthand side), and along major roads. Source: Celestia.

6.1 Summary of Findings

The general objective of this thesis was to investigate the impacts and mitigation/adaptation strategies related to the interplay between land-use change, increased production of biofuels, and climate change in selected world regions which are expected to largely experience these processes in the near future. Therefore four independent, though related, modeling studies were carried out in Brazil, India, Germany and the Brazilian Amazon. This final chapter summarizes the major findings of these four studies and points out future research needs related to the subjects of this thesis.

The major conclusions that can be drawn from the whole thesis are that:

(i) biofuels must be analyzed and planned carefully in order to effectively reduce carbon emissions and avoid the displacement of other land uses;

(ii) future climate change can have considerable impacts on the location and extent of land-use changes;

(iii) intensification of grazing livestock represents a promising venue for minimizing the impacts of future land-use and land-cover changes in Brazil.

In the next sections each of the main questions presented in chapter 1 are revisited along with a summary of the methods used to pursue them.

6.1.1 Potential Crop Productivity and Land Requirements for Biofuels in Brazil and India

Would the spatial variation in the potential yields of sugarcane and jatropha lead to different land requirements to fulfill the biofuel production targets planned by the Brazilian and Indian governments in the near future?

The country-wide spatial variation in the yields of sugarcane and jatropha incurs into substantially different land requirements to meet the biofuel production targets for 2015 in Brazil and India, depending on the location of plantations. Particularly the average land requirements for jatropha in India are considerably higher than previously estimated. The findings of chapter 2 indicate that crop zoning is important to avoid excessive LUCC.

Summary of methods: simulation of potential crop productivity with the LPJmL model (with the parameterization of two new crop types), and GIS[10] operation.

[10] Geographical Information System

6.1.2 Biofuel-Driven Indirect Land-Use Changes in Brazil

What would be the location and extension of the indirect LUCC generated by the fulfillment of Brazil's biofuel production targets for 2020? Would the carbon emissions from such indirect LUCC impair the carbon savings from the use of these biofuels instead of fossil fuels?

Brazilian biofuels can cause considerable indirect LUCC (ILUC) by 2020, especially by pushing the rangeland frontier into the Amazonian forests, as shown in chapter 3. The carbon debt caused by such ILUC would result in no carbon savings (from using plant-based ethanol and biodiesel instead of fossil fuels) before 44 years for sugarcane-ethanol and 246 years for soybean-biodiesel. Intensification of livestock grazing could avoid such ILUC. We argue that such an intensification of livestock should be supported by the Brazilian biofuel sector, based on the sector's own interest in minimizing carbon emissions.

Summary of methods: modeling of LUCC with the LandSHIFT model, combined with pre-existent life cycle assessments for biofuels.

6.1.3 The Land Potential for Biogas Crops in Germany

How can the location and capacity of specific infrastructure units (such as biogas plants) be integrated in the LandSHIFT model?

In chapter 4 we introduced a new crop allocation method in LandSHIFT to be used in studies of LUCC involving land uses that are tightly linked to specific infrastructure units, such as biogas plants or sugar/ethanol mills. An exemplary application of the method showed that Germany has enough land to fulfill virtually all (90 to 98%) its current biogas plant capacity with only cultivated feedstocks. Biogas plants located in South and Southwestern (North and Northeastern) Germany might face more (less) difficulties to fulfill their capacities with cultivated feedstocks, considering that feedstock transport distance to plants is a crucial issue for biogas production.

Summary of methods: modeling of LUCC with the LandSHIFT model (with implementation of a new crop allocation method).

6.1.4 Climate change, Conservation Targets and Land Use in the Brazilian Amazon

What are the impacts of different climate change scenarios on LUCC in the Brazilian Amazon? What are the impacts of halting Amazon deforestation on the production of food and land-use intensity?

Chapter 5 shows that future climate change can affect both the location and extent of LUCC. For example, in our simulations the deterioration of climate in some regions by 2050 can shift the deforestation frontier to areas that would experience low levels of human intervention with mild climate change (such as the western Amazon forests or parts of the Cerrado savannas). Halting the deforestation of the Amazon and the Cerrado would require either a reduction in the production of meat or an intensification of livestock grazing in the region. Such findings point out the need for an integrated/multidisciplinary plan for adaptation to climate change in the Amazon.

Summary of methods: modeling of future crop/pasture productivity with the LPJmL model, modeling of LUCC with the LandSHIFT model (adapted to the Amazon region), analysis of conservation scenarios.

6.2 Future Research

Future research related to this thesis could concentrate in two major fronts. The first would be tackled with a real coupling between the models used in this thesis (LandSHIFT, LPJmL, and IMPACT models). The dynamic exchange of information between these models would permit, for example, the investigation of feedbacks between future food production demands and climate change or conservation targets. The second would be the implementation of different farming systems in LandSHIFT, separating smallholder agriculture[11] from large-scale technology-intensive agriculture. That would allow investigating, for example, how the LUCC impacts shown in this thesis would specifically affect smallholder farming.

[11] Defined by *Barnett* [1997] as ‘‘farming and associated activities which together form a livelihood strategy where the main output is consumed directly, where there are few if any purchased inputs and where only a minor proportion of output is marketed

APPENDIX A

Supporting information for chapter 2

Contents

- Text: Parameterization of sugarcane and jatropha in the LPJmL model
- Figures A.1-A.3
- Tables A.1-A.2

A.1 Parameterization of sugarcane and jatropha in the LPJmL model

A.1.1 The LPJmL Dynamic Global Vegetation Model

The LPJmL model is based on the Lund-Potsdam-Jena-DGVM [*Sitch et al.* 2003; *Gerten et al.* 2004] a biogeochemical process-based model that simulates global terrestrial vegetation dynamics and the associated carbon and water cycles. The model accounts, on a 0.5º grid base, for the processes of photosynthesis, evapotranspiration, autotrophic and heterotrophic respiration, effects of soil moisture and drought stress, as well as functional and allometric rules defined for 10 plant functional types (PFTs) representing natural vegetation. The new LPJ version, LPJmL (mL stands for managed land), simulates agricultural land use within the same framework, but using the concept of crop functional types (CFTs) [*Bondeau et al.* 2007]. Thirteen CFTs represent the most important field crops, including pastures, either rainfed or irrigated, each one with its own phenology and growth parameterizations. Carbon is allocated daily to four different plant compartments, including a storage organ that represents the harvested part economically explored by humans. For most of the CFTs crop yield for each grid cell is limited by soil moisture and climate only. For two of them (wheat and maize) a simple scaling rule relates reported statistics on fertilizer use per country [*IFA* 2002] to maximum leaf area index. Full model description as well as its extensive validation against observed data on sowing dates, fraction of photosynthetically active absorbed radiation, seasonal CO_2 flux exchanges, and crop yields can be found in the study by *Bondeau et al.* [2007]. Although fully comprehensive in biogeochemical processes, LPJmL does not consider important tropical crop types as sugarcane or *Jatropha curcas* that have very distinct characteristics in relation to the other CFTs already considered in LPJmL. In the next sections we present the implementation of sugarcane and *Jatropha* as new CFTs in LPJ.

A.1.2. Implementation of Sugarcane

Sugarcane CFT was implemented in LPJmL using the same approach as for the other CFTs previously implemented in the model. The main differences, which justify its implementation as a new CFT, lie in its distinct phenological development curve (Figure A.1) and higher harvest index (fraction of aboveground biomass that is harvested) when compared to other LPJmL CFTs. Table A.1 shows the parameters and their values for the new sugarcane CFT. Some of the parameters follow the parameterization in other models like SWAT [*Neitsch et al.* 2002] or DAYCENT [*Parton et al.* 1998; *Stehfest et al.* 2007]. However some values used here differ from those models, mainly due to the differences in the very nature of these models: SWAT for example calculates photosynthesis and net primary productivity on an empirical basis, while in LPJmL they are implemented on a process base. Irrigation priority (see Table 1 in *Bondeau et al.*

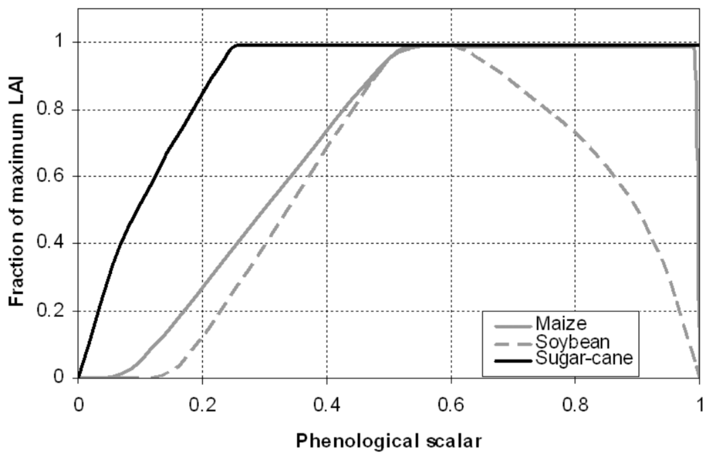


Figure A.1: Leaf area development in LPJmL for tropical crop types as fraction of maximum leaf area index under optimal conditions (non water-stressed).

[2007]) for sugarcane is set to 3, based on Indian agricultural practices, since most of sugarcane plantations in India are irrigated [*Ministry of Agriculture* 2009]. Sowing date is determined by soil water availability (w) in summer/autumn, with a threshold of w = 0.40, as is used for other tropical CFTs in LPJmL [*Bondeau et al.* 2007]. Phenological development is modeled using the heat unit theory [*Boswell* 1926] by accumulating daily mean temperatures above a specific base temperature up to a maturity threshold. In order to better simulate sugarcane yields worldwide, we adopted the fertilization-maximum LAI (LAImax) scaling rule used for wheat and maize also for this CFT, based on *IFA* [2002] fertilizer consumption for sugarcane per country. For countries with low (high) fertilizer input on sugarcane plantations, the model uses the LAImin (LAImax) value shown in Table A.1.

Calculations were made for the 1901-2003 period, driven by University of East Anglia's Climatic Research Unit (CRU05) climate data set, a monthly climatology of meteorological variables, and atmospheric CO_2 concentration as in the work by *Bondeau et al.* [2007]. The transient simulation is preceded by a 1000-year spin-up period during which the first 30 years of the climate data set are repeated cyclically in order to bring all carbon pools into equilibrium. Actual (1990) spatial global distribution of sugarcane plantations was derived from [*Leff et al*. 2004] and scaled to the whole 20th century following the procedure described by *Bondeau et al.* [2007]. The actual crop distribution was used in a first run, to evaluate actual productivity against reported data. Later on we performed a run where all grid cells are assigned to sugarcane plantations in order to assess potential productivity, which was then used for the calculation of land requirements. Country level actual yields were evaluated against FAO data [*FAO* 2010] for the 1991-1995 period, which is the period most of the LPJmL CFT parameterizations rely on. This comparison was made for 38 countries representing more than 95% of world's sugarcane. Figure A.2 shows that the

Table A.1: Parameters and constants used for sugarcane CFT in LPJmL. Other parameters not cited here have the same value as for the other CFTs in the study by *Bondeau et al.* [2007].

Symbol	Value	Units	Description
r	0.005	gC/gN/d	Tissue respiration rate at 10ºC
T_b	11	ºC	Base temperature
E_{max}	7	mm/d	Maximum transpiration rate
init	120; 300	Julian days	Sowing date initialization (default value for N/S Hemisphere)
hlimit	240	days	Maximum lenght of crop cycle
phu	2800	ºC day	Phenological heat units
LAI_{max}	7	m^2/m^2	Maximum leaf area index
LAI_{min}	2	m^2/m^2	Minimum leaf area index at harvest
hi_{opt}	0.8	(0-1)	Optimum harvest index
hi_{min}	0.2	(0-1)	Minimum harvest index

simulated yields of large producers such as Brazil, India, China, Thailand and Pakistan are in good agreement with FAO data. However this first analysis revealed a strong underestimation of sugarcane yields in some of the 38 countries, which we think is due to the irrigation priority list used in LPJmL, which ranks rice and maize before sugarcane for all countries worldwide. This irrigation priority list is derived mostly from European agricultural practices. Thus in South Africa, for example, which has a total area of 11,600 km^2 under irrigation, LPJmL assumes that all irrigation is used in maize plantations, while it should use only ~20% of that value [*FAO* 2009]. This results in an overestimation of maize yields (not shown) and underestimation of sugarcane yields in this country. Thus, to test whether this underestimation is indeed due to the irrigation scheme used or due to the parameterization given, we calculated yields for these countries having sugarcane as the first in the irrigation priority list. That is, instead of rice or maize, all irrigation is directed to sugarcane plantations. This change significantly improved the performance of 15 countries, most of them located in Africa and Latin America (Australia, Bolivia, Colombia, Ecuador, Guatemala, Iran, Kenya, Mexico, Peru, Philippines, South Africa, Sudan, Swaziland, Venezuela, and Zimbabwe). We conclude that most probably in these countries more water is used for irrigating sugarcane plantations than assumed in the original approach of LPJmL (irrigation priority list). Moreover, as stressed by *Bondeau et al.* [2007] some yield-impacting processes are considered in simplified form or not at all in LPJmL, therefore only a first-order qualitative comparison can be made, which is appropriate for the purpose of this study. Nevertheless, the procedure resulted in a better correlation of the modeled data against FAO's (r = 0.54), which is even higher when considering just those countries sharing more than 1% of sugarcane world production (r = 0.76). It should be mentioned that for most sugarcane cultivars the aboveground biomass can be harvested 4 to 5 times since roots are kept in the soil

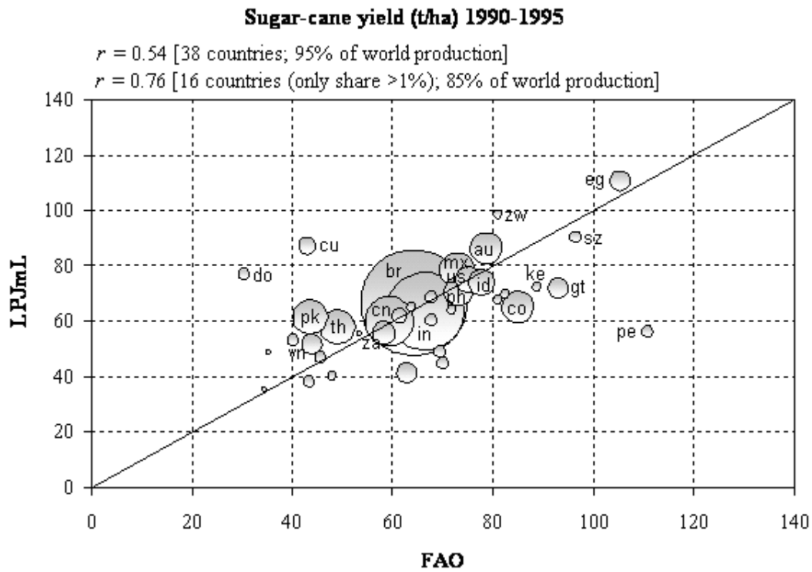


Figure A.2: Country based evaluation of sugarcane yields simulated by LPJmL against FAO observed data. Bubble size depicts share of sugarcane world production. Internet country codes are used: ar: Argentina, au: Australia, br: Brazil, cn: China, co: Colombia, cu: Cuba, do: Dominican Republic, eg: Egypt, gt: Guatemala, id: Indonesia, in: India, ke: Kenya, mx: Mexico, pe: Peru, ph: Philippines, pk: Pakistan, sz: Swaziland, th: Thailand, us: USA, vn: Vietnam, za: South Africa, zw: Zimbabwe.

for regrowth. However, because here we are only interested in crop yields and not in the full carbon cycle, and following the other CFTs in LPJmL, roots are incorporated to soil carbon after harvest in this parameterization of sugarcane.

A.1.3. Implementation of Jatropha

Jatropha curcas (hereafter jatropha) is a perennial deciduous shrub native from Central America, nowadays widespread throughout the tropics [*Openshaw* 2000; *Achten et al.* 2008]. Its seeds bear high oil content, which has been used for biodiesel production especially in India [*Openshaw* 2000; *Achten et al.* 2008]. The LPJmL crop module accounts only for annual crops, which are sown, grown and harvested within a year, after which the plants die and their residual biomass (including roots) is incorporated into the soil or is removed [*Bondeau et al*. 2007]. For that reason jatropha was implemented in LPJmL within a natural PFT framework. With that, important parts (e.g. roots and sapwood) of the plant biomass last for many years, as is appropriate for a permanent (i.e., non-annual) crop. Nevertheless jatropha's yearly leaf phenology is represented with a raingreen phenology scheme, and is dependent on soil water availability. Table A.2 present the values used for jatropha PFT parameterization in LPJmL. Leaf senescence (or irrigation, when prescribed) occurs if ω (the relationship between plat water canopy supply and atmospheric demand for transpiration) falls below 0.2. For irrigated jatropha water is provided as long as $\omega < 0.2$. This value makes jatropha more drought resistant when compared to the other CFTs which all have a ω value of 0.3. Plant maximum height is constrained to 7 meters, which is the maximum height of an adult jatropha shrub [*Openshaw* 2000; *Achten et al.* 2008; *Jongschaap et al*. 2007]. Ecosystem-to-leaf level ratio of absorbed radiation has the same value as for any LPJmL CFT. Maximum fractional projective cover (FPC_{max}) is constrained to 0.4, which associated with a maximum crown area (CA_{max}) of 5 m^2 results in an average number of about 2,500 jatropha individuals per hectare (in agreement with the most recommended spacing of jatropha individuals of 3 to 2 m [*Planning Commission* 2003; *Achten et al*. 2008]. Harvested parts (fruits) are a fixed 20% fraction of annual net primary productivity. This value is ranging between those indicated by *Achten et al.* [2008] and *Jongschaap et al*. [2007]. The reason for deducting the harvest from NPP and not from aboveground biomass, as it is implemented for the other CFTs, is that aboveground biomass, differently from NPP, does not vary considerably from one year to another in mature perennial plantations. For example, if in a certain year there was no significant NPP but there is still a considerable aboveground biomass (remaining from previous years), then it is likely that the plants will not produce fruits this year. In this case, deducting harvest from aboveground biomass would not be adequate. Mortality of trees follow the original rules set in LPJ, and jatropha trees die due to light competition, low growth efficiency, negative annual carbon balance, heat stress or when PFT bioclimatic limits are exceeded for an extended period [*Sitch et al*. 2003]. Even though the current LPJmL version does not simulate age structure, it is assumed that the average simulated jatropha individual is adult and thus more reliable when compared to observed data on adult jatropha trees (older than 5 years).

Table A.2: Parameters and constants used for jatropha PFT in LPJmL. Other parameters not cited here have the same value as for the other tropical PFTs [*Sitch et al*. 2003]. PAR: photosynthetically active radiation.

Symbol	Value	Units	Description
z_1	0.8	-	Fraction of roots in the upper soil layer
ω_{min}	0.2	-	Ratio between water supply and demand below which stomata close
r	0.011	gC/gN/d	Tissue respiration rate at 10ºC
a_{leaf}	0.5	yr	Leaf longevity
lr_{max}	1.0	-	Leaf-to-root ratio under nonwater stressed conditions
α_a	1.0		Ecosystem-to-leaf level ratio of assimilated PAR
T_{photos}	20; 45	ºC	Lower and upper limit of temperature optimum for photosynthesis
$T_{c,min}$	11	ºC	Minimum coldest-month temperature for survival
CA_{max}	5	m^2	Maximum shrub crown area
H_{max}	7	m	Maximum shrub height
FPC_{max}	0.4	-	Maximum fractional percentage cover

Modeling protocol was nearly the same as mentioned in the previous section. However, differently from the sugarcane calculation and since the assessment for jatropha is only potential, the whole world is covered naturally (following LPJmL establishment rules) by jatropha, of course respecting climatic thresholds for this PFT. The general global pattern (not shown) is to have the potential presence of jatropha trees within the 30° latitude belt. Highest yields occur in areas with high precipitation levels and maximum temperature, due to jatropha's higher range of temperature optimum for photosynthesis (20-45ºC). Comparison of modeled results and observed data was possible for selected countries or regions, strongly constrained by the availability of data on jatropha yields and biometric information. For country level data we used the country's mean LPJmL yield, while for municipality or regional data, we used only the value of the respective grid cell. Figure A.3 shows that LPJmL jatropha yields correlate well with the available observed

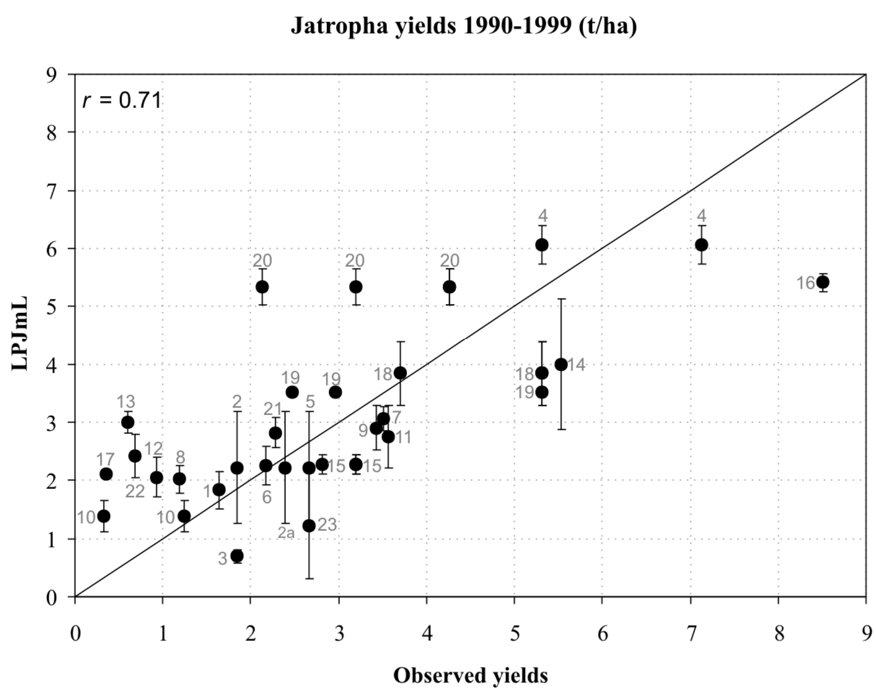


Figure A.3: Evaluation of jatropha yields simulated by LPJmL averaged for 1990-1999 period against observed data in several countries/regions. Vertical bars depict standard deviation. Numbers indicate location: 1: Burkina Faso; 2: India; 3: India, 450mm; 4: India, irrigated; 5: India, wasteland; 6: India, Andhra Pradesh; 7: India, Karnataka; 8: India, Maharashtra; 9: India, Orissa; 10: India, Rajasthan; 11: India, Tamil Nadu; 12: India Uttar Pradesh; 13: India, West Bengal; 14: Madagascar; 15: Mali; 16: Mali, irrigated; 17: Mali, Dijidian; 18: Nicaragua; 19: Nicaragua, Managua; 20: Paraguay (different ages); 21: Thailand; 22: Zimbabwe; 23: non-identified semi-arid areas. Sources of observed data (sorted by point number) are: 9: *Fairless* [2007]; 7: *Bharadwaj et al.* [2007b]; 6, 8, 11, 12, 13: *NOVOD Board* [2008]; 2a: *Kaushik and Kumar* [2006]; all other data points: *Achten et al.* [2008].

data (r = 0.71), especially for data on adult jatropha trees. Examples of that are the three points in Paraguay, which represent three different ages with increasing yields for older trees. Correlation is good in India (e.g. Karnataka, Orissa, India irrigated), although there are some outliers like West Bengal, where LPJmL overestimates yields. Nevertheless, one should take into account the reliability of the observed data on jatropha, which not necessarily always follow the same procedure for yield measurements [see *Achten et al*. 2008]. No data on jatropha yields were found for Brazil, since controlled experiments there have started only very recently.

Modeled mean global NPP value is 557 g m^{-2} y^{-1} (± 373 SD) which is inside the NPP interval of a shrubland biome (400-800 g m^{-2} y^{-1} [*Cramer et al.* 1999]). Biomass per individual is also in agreement with the two unique observed data found (both for seven year old trees). In Egypt, under irrigated condition [*Henning* 2009] reported a value of 50 kg of dry matter per individual, comparable to the LPJmL 40-70 kg of dry matter per individual interval in that country. *Francis et al*. [2005] indicate 13.4 kg of dry matter per individual (± 2.5 SD) in Indian wastelands, comparable to LPJmL 0-15 kg of dry matter per individual interval in whole India (since it is hard to precise what are wastelands and where they are located). Woody biomass mean global value is of 85.8% (± 10.7 SD), compared to the 74 to 92% interval in a generic shrubland in Texas (mean value for 10 shrub species) [*Northup et al*. 2005].

APPENDIX B

Supporting information for chapter 3

Contents

- Text: LandSHIFT model evaluation for Brazil (full)
- Figures B.1-B.4

B.1 LandSHIFT Model Evaluation for Brazil (Full)

B.1.1 Crop/rangeland location

Because crop/rangeland suitability analysis is the central aspect of LandSHIFT, its ability to determine crop/rangeland spatial distribution requires testing. Therefore, we first compared the suitability computed by LandSHIFT against crop and rangeland distribution on an actual land-use map [*Loveland et al.* 2000; *Heistermann* 2006]. Cropland areas tend to be located where crop suitability is higher, assuming that cropland is given priority over other land uses (besides urban areas) [*Schaldach and Koch* 2009]. Figure B.1 shows that suitability for cropland and rangeland indeed tends toward higher values in comparison to suitability for other land uses, suggesting that the suitability analysis used in the model is appropriate for determining cropland/rangeland allocation. The suitability frequency distribution of 'other land uses' is significantly different from that of cropland and rangeland (Kolmogorov-Smirnov test, $P<0.01$). There is no significant difference between the suitability distributions for cropland and rangeland. Overall, this analysis suggests a tendency to allocate crops in places with higher suitability. It can be argued that the estimation of the w_i weights (Table 3.1) using the initial land-use map (which is the same used in this evaluation) may create a spurious dependency between the datasets used for comparison, thus impairing the reliability of this test. Therefore, we performed the same suitability frequency

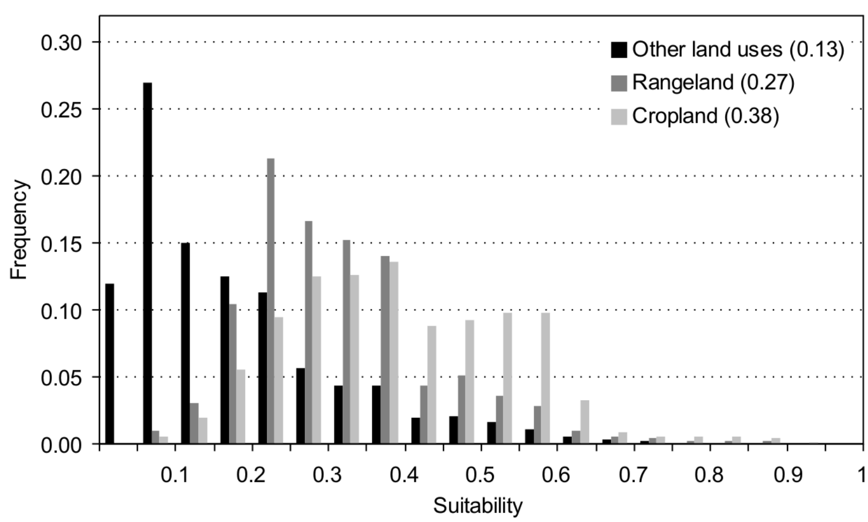


Figure B.1: Frequency distribution of suitability values among different land-use activities: cropland (n = 7436), rangeland (n = 22577), and other land uses (n = 72848). Value in parenthesis indicates the median suitability for the given land use.

distribution test for cropland with the w_i weights all having the same value of 0.16. This analysis further confirmed what is shown in Figure B.1 because the median suitability for cropland (0.56) differs even more from that of other land uses compared to the analysis in which the weights were determined using the initial land-use map. Moreover, the distribution in which all $w_i = 0.16$ is not significantly different from the distribution for cropland using the pre-determined w_i weights in Table 3.1. Despite incurring an overlap with 'other land uses' between suitability values of 0.15 and 0.4 (Figure B.1), the latter distribution is preferred over the one in which w_i have all the same values because it better represents the distribution of croplands throughout the whole country and avoids excessive (and erroneous) concentration of cropland in the southern and southeastern states of Brazil.

A second test using the Relative Operating Characteristics (ROC) method [*Pontius and Schneider* 2001] makes it possible to assess the degree to which the spatial pattern computed by the model is random or not. This ROC also compares computed suitability to the actual land-use map pattern but relates the proportions of correctly (true positives) and incorrectly (false positives) classified spatial predictions in contingency tables. The resulting curves are shown in Figure B.2. The area under the curve (0.87 for cropland; 0.80 for rangeland) reveals that the spatial pattern of suitability computed by LandSHIFT is not random as exemplified by the 1:1 line, which has an area under the curve of 0.5. This result further confirms that higher suitability values tend to be located in grid cells occupied by cropland and rangeland. Therefore, the ROC method test suggests LandSHIFT is able to represent crop location using suitability analysis. A third analysis regarding crop/rangeland distribution inside major regions in Brazil is presented below.

B.1.2 Crop/rangeland area

We compare crop area modeled by LandSHIFT with reported statistics data [*FAO* 2010] for the year 2003. At the country level, modeled crop areas of sugarcane and soybean match FAO data almost perfectly, whereas the area covered by 'other crops' and rangeland (and therefore livestock density, Ld) is overestimated in the model by 13% and 8% respectively. This result suggests the model is able to convert country-scale crop production mass (e.g., Mg) to cropland area (km^2). Model efficiency [*Janseen and Heuberger* 1995] for the data presented in Figure B.3 is 1.06 (1.0 would represent a perfect match). The overestimation of the 'other crops' area is due to some underestimation of crop yields by LPJmL. However, in the case of rangeland, the area overestimation might also be due to the following reasons: (*i*) the assumption of only one land use per grid cell leads to overestimation of rangeland area, especially in regions where Ld is low, as in Northeast Brazil; and (*ii*) rangeland area might not increase in response to increasing livestock herd in all areas of Brazil, as modeled by LandSHIFT. For example in the Amazon region the farmer's interest is often on guaranteeing ownership over the land rather than on allocating the market demand for livestock on his pastures, and the pasture area may increase not because of increasing livestock demand but because of less obvious reasons like population migration and lack of governance in the region [*Nepstad et al*. 2006; *Fearnside* 2008].

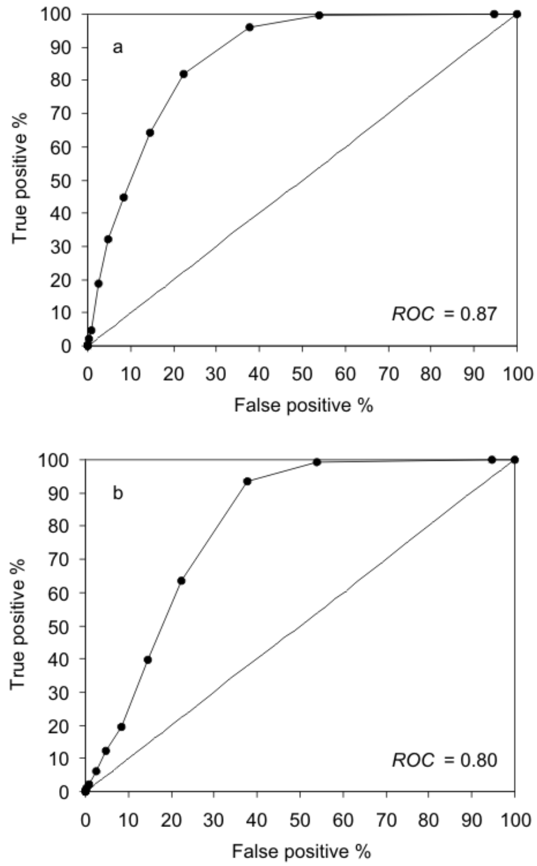


Figure B.2: Relative operating characteristic (ROC) curves for comparison between cropland and other land uses excluding rangeland (**a**), and between rangeland and other land uses excluding cropland (**b**).

Distribution of cropland/rangeland inside major regions in Brazil is in good agreement with statistics on a sub-national level [*IBGE* 2010] weighted by total crop/rangeland area modeled by LandSHIFT (Figure B.4). The underestimation of rangeland area in southern Brazil is corrected if we add 68,000 km^2 of natural grasslands, which are considered in the Brazilian official statistics as 'natural pasture' but are not included in LandSHIFT calculations. The overestimation of rangeland area in Northeast Brazil is explained by two reasons (*i*) the difficulty to deal with the extension of rangeland in areas with low Ld [*FAO* 2007b], and (*ii*) the rangeland area in Northeast Brazil is overestimated by a factor of 2.3 in the initial land-use map used by LandSHIFT [*Loveland et al.* 2000; *Heistermann* 2006]. Estimates by *Campbell et al*. [2008] suggest that roughly 110,000 km^2 of the rangelands in Northeast Brazil are abandoned (not grazed

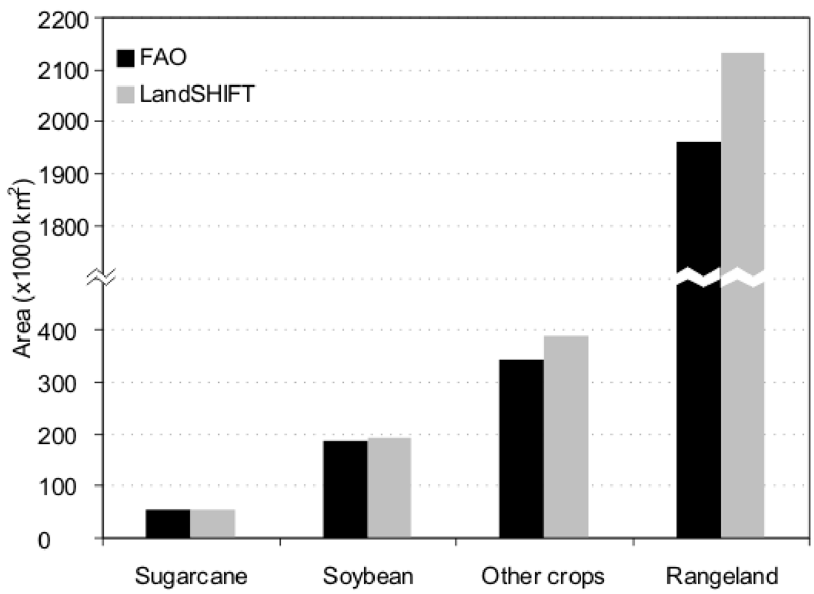


Figure B.3: Comparison of cropland and rangeland area modeled by LandSHIFT against FAO statistics for the year 2003.

anymore). These areas are probably not considered as rangeland in the statistics used here for comparison.

B.1.3 Deforestation rates

The modeled annual deforestation rate for the Amazon region for the 1992-2003 period compares well with remote sensing data (LandSHIFT: 16,789 km^2 yr^{-1}, INPE-PRODES: 18,266 km^2 yr^{-1} [*PRODES* 2009]). The shares of this deforestation among states are also comparable with PRODES, though deforestation in Maranhão is overestimated by a factor of 23. That overestimation is due to the denser road network found in this state compared to Mato Grosso, where deforestation is underestimated by a factor of 5.7. Nevertheless, any comparison between different data sets is biased by the different methods used in the construction of a given map. For example, the initial land-use map for the year 1992 used in LandSHIFT has 80% more forest in the state of Maranhão compared to the dataset used for comparison here [*PRODES* 2009]. Moreover, capturing the exact location of deforestation in the Amazon region, which is not the goal of this study, might involve other factors that are not accounted for in a country-scale simulation program such as LandSHIFT, in which deforestation is mostly caused by increasing crop and/or livestock demand. The deforestation model developed by *Soares-Filho et al*. [2006] is

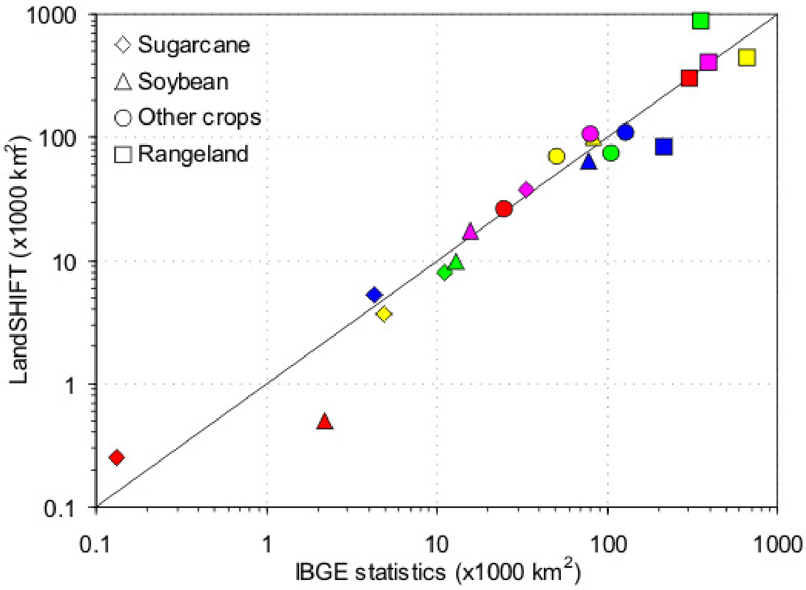


Figure B.4: Comparison of crop/rangeland distribution within major Brazilian regions modeled by LandSHIFT against IBGE subnational statistics [*IBGE* 2010] weighted by modeled total crop/rangeland area (logarithmic scale). Yellow: Centre-West; red: North; green: Northeast; blue: South; purple: Southeast*.

*Centre-West: Distrito Federal, Goiás, Mato Grosso, Mato Grosso do Sul; North: Acre, Amapá, Amazonas, Pará, Rondônia, Roraima, Tocantins; Northeast: Alagoas, Bahia, Ceará, Maranhão, Paraíba, Pernambuco, Piauí, Rio Grande do Norte, Sergipe; South: Paraná, Rio Grande do Sul, Santa Catarina; Southeast: Espírito Santo, Minas Gerais, Rio de Janeiro, São Paulo.

focused on the Amazon basin and considers neither the dynamics of land use occurring at deforested sites, nor the teleconnections between land-use changes in Amazonia and other parts of Brazil. Also, the current version of LandSHIFT does not consider forestry activities, which may contribute to deforestation. The modeled deforestation rate in the Cerrado savanna of Central Brazil for the 1992-2003 period is 17,753 $km^2\ yr^{-1}$. This amount lies within the estimated range (13,100-26,000 $km^2\ yr^{-1}$) of Cerrado deforestation for the last decade [*Sawyer* 2008]. The deforestation of ~5000 km^2 of the Atlantic forest in the 1992-2003 period [*SOS Mata Atlântica and INPE* 2008], approximately 55 grid cells in LandSHIFT's resolution, is not captured by the model.

APPENDIX C

Supporting information for chapter 5

Contents

- Text: Land use maps of the Legal Amazon in 2001 and 2006
- Figure C.1
- Table C.1

C.1 Land use maps of the Legal Amazon in 2001 and 2006

For the production of the two land-use maps used in this study (chapter 5) we used land-cover maps of the legal Amazon in 2001 and 2006, produced out of PRODES satellite data [*PRODES* 2009], a 2000 vegetation map of South America (Eva et al. 2004), and classified MODIS vegetation continuous field [*Hansen et al*. 2002]. These land-cover maps were degraded to the resolution of 5 arc-minute, and divided into 32 socio-economical regions, as suggested by *Garcia et al*. [2004] and *Soares-Filho et al*. [2006]. Each of these subunits had their own crop and pasture area determined from the IBGE municipal agricultural production database for the given years [*IBGE* 2010]. Because data on pasture area is not available for the year 2001, it was estimated, through linear interpolation, from the 1996 and 2006 data. The crop types and other land uses considered in the confection of the land use maps are shown in Table C.1. Only areas depicted as deforested or as Cerrado savanna (since land-cover changes of this latter are not tracked by satellites as the deforestation of the Amazon) could have the assignment of crops or pasture. Crops had priority over pasture for occupation of grid cells, while only one dominant land use type can occur in one grid cell. The allocation procedure followed a preference list of grid cells, which was built based in a 2000 map on the geographical distribution of crop/pasture areas, also on 5 arc-minute resolution [*Monfreda et al*. 2008; *Ramankutty et al*. 2008]. Grid cells with higher fraction of a given crop type in the map by *Monfreda et al*. [2008] had preference for assignment of that crop type in our land-use map. Disambiguation within one crop type (i.e. when the *Monfreda et al*. map for soybeans, for example, had several grid cells with the same area) or between different crop types (i.e. when *Monfreda et al*. maps for two or more different crop types had exactly the same value in a given grid cell), was performed using a multi-criteria analysis (MCA) of slope, potential productivity of the given crop type (or grassland for pasture), distance from settlements, soil type and distance from paved roads [for data sources see *Soares-Filho et al*. 2006]. However, this MCA was needed only in a minor fraction (<1%) of the grid cells that later were assigned as crop or pasture. Therefore the maps of *Monfreda et al*. (for crops) and *Ramankutty et al*. (for pastures) played the major role in the allocation of land uses in our base maps. Urban areas were assigned to those grid cells having a population density higher than 2000 cap km^{-2} [*Erb et al*. 2007], using the HYDE map of population distribution [*Goldewijk* 2005], with no distinction between years 2001 and 2006.

A first assessment of the land-use maps revealed that the area assigned as ‘abandoned’ was too large (350,000 km^2 in 2001), surpassing any estimate on the extent of land currently abandoned in the legal Amazon, which ranges from 61,000 km^2 to 106,000 km^2 (several datasets analyzed by *Campbell et al*. [2008]). In fact the very concept of abandoned land is quite variable, and can, for example, refer to temporal characteristics (e.g. set-aside), soil conditions (e.g. degraded), or management (e.g. poorly-managed) of the land use. Here the land use type “abandoned” is considered to be simply land with no occurrence of any other land-use type. Therefore, considering that PRODES provides trustworthy numbers for the extent of the Amazon forest, and

Table C.1: Land use types considered in the land-use maps presented here.

Land-use type	Identifier	Description
tropical forest	2	-
savanna (Cerrado)	9	-
rice	103	paddy rice
maize	104	maize
other tropical cereals	105	millet, sorghum, quinoa
pulses	106	dry beans, dry peas, chick peas, lentils
tropical roots and tubers	108	cassava, sweet potatoes, yams
annual oil crops (excl. soybeans)	109	groundnuts, rape, sesame, sunflower
soybeans	110	soybeans
permanent oil crops	111	oil palm, coconut, olives
fruits	112	see http://faostat.fao.org for details
sugarcane	114	sugarcane
fiber crops	115	cotton, hemp, flax
coffee and cocoa	116	coffee, cocoa
other stimulants	117	tea, tobacco
poorly-managed pasture	118	lower intensity of use, lower livestock density (compared to well-managed pasture) mixed with degraded vegetation and not accounted in official statistics
well-managed pasture	119	higher intensity of use, higher livestock density (compared to poorly-managed pasture), accounted in official statistics
abandoned	121	no occupation with any land use
urban	160	cities

that the extent of Cerrado in our maps are certainly optimistic [compare with Figure 6 in *Machado et al*. 2004] - besides the fact that most of the geographical subunits with abandoned lands did not have Cerrado within it limits (e.g., Paragominas) - we argue that the IBGE data for pasture area in the legal Amazon might be underestimated, at least in some regions [see *Ramankutty et al*. 2008]. Thus in order to correct this 'discrepancy', after IBGE area requirement for pasture is fulfilled in our maps (i.e., all crop and pasture areas were allocated at this stage), we assign the type 'poorly-managed pasture' to all the remaining grid cells which are covered by pasture in *Ramankutty et al*. [2008] map and were, until this stage, set as 'abandoned' in our land-use map. That reduces the area of the abandoned land-use type to 102,000 km^2, in better agreement with data available for comparison. This type of pasture is meant to represent pastures with a lower intensity of use, with lower livestock density (compared to wm-pastures), and mixed with degraded/secondary vegetation [compare locations with *INPE* 2009]. The other type of pasture is then referred to as 'well-managed pasture'.

In general the methods used to obtain these maps are not as comprehensive as, for example, the one used by *Cardille and Foley* [2003] to produce land-use maps of the Brazilian Amazon for 1980 and 1995 (e.g., our maps have only one land use per grid cell instead of fractional coverage), even though they are in accordance to official statistics (IBGE). But most of all, one should consider that the maps presented here were produced as to fit their use in the LandSHIFT model.

Areas of each major land-use type in 2001 and 2006 are shown in Figure C.1. 36% of all cultivated land in the Legal Amazon in 2006 was covered with soybean fields, located namely in Mato Grosso. Well-managed pastures are concentrated along the deforestation arc but with a general tendency of not occurring at the deforestation frontier, such as in Southeast Pará. Poorly-managed pastures are located mostly in the eastern deforestation frontier, and in interior Maranhão. During the 2001-2006 the Amazon forest lost more than 125,000 km^2, giving place mainly to pastures. Most of the soybean expansion took place in Mato Grosso, replacing the Cerrado savanna (which lost more than 30,000 km^2 in that period) as well as pastures and other crops.

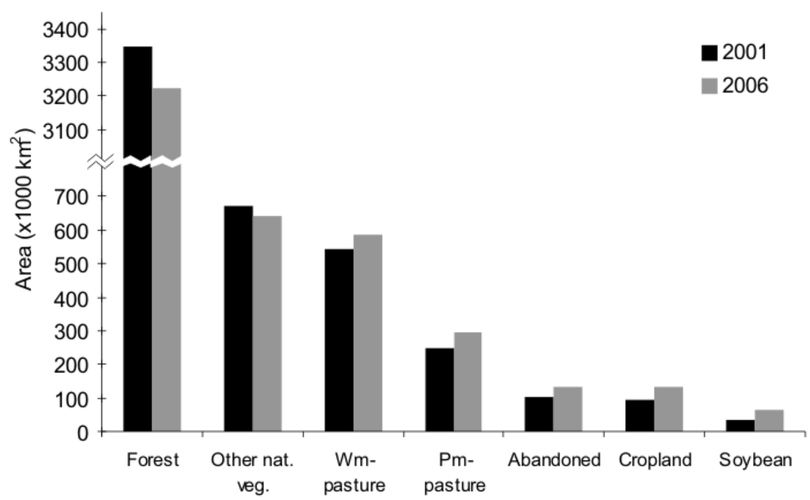


Figure C.1: Area of different land uses in the legal Amazon in 2001 and 2006 as determined from satellite-based maps and agricultural statistics.

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