The future of global land use

Model-based assessment of policy options to reverse forest loss, mitigate climate change and safeguard biodiversity

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Summary

Global land-use change is a main driver of two of the biggest environmental alterations threatening human well-being: climate change and the decline in biodiversity. Over recent decades, a growing and wealthier population has requested more agricultural products, and this demand has been satisfied by expanding agricultural areas and intensifying production. However, this has posed consequences. As new fields and pastures replaced tropical forests, some of the most biodiverse habitats have been destroyed, and as tropical forests are among the ecosystems with the highest carbon content, their loss has also significantly contributed to global climate change.

The pathway of global land use towards the future remains unclear, but with the projected rise in the global population and the consequent greater demand for food in addition to newly emerging requests for agricultural energy production and land-based climate change mitigation, the pressure on land will likely increase. This thesis conceptualises scenarios of future land use, identifies probable future developments and analyses policies that might help to steer land use in a more environmentally friendly direction.

Forests and how their future extent influences climate and biodiversity are central to this thesis. It assesses policies of a price on greenhouse gas emissions from land-use change and an expansion of protected areas, as well as the effects of agricultural trade liberalisation and intensification. This paper seeks to more concretely determine the expected amount of tropical deforestation until the middle of the century and measures to reduce it. It additionally focuses specifically on tropical biodiversity hotspots and methods to conserve them through the designation of protected areas and agricultural intensification. Furthermore, it considers the carbon sequestration potential of large-scale afforestation and its potential effect on food prices.

The study employs a global economic land-use model to assess potential future developments. The Model of Agricultural Production and its Impact on the Environment (MAgPIE) produces patterns of global land use for a given demand for agricultural products by minimising costs of agricultural production. To analyse the research questions at hand, the model was modified and amended in several aspects.

This thesis shows that the assumed increase in demand for agricultural products will lead to an expansion of croplands, at least until the middle of the century. More than 400 million hectares (Mha) of cropland may be newly established between 2015 and 2050. Continuous loss of forests and other natural vegetation is likely to accompany this if no adequate policies come into force. In a study focusing on the tropical deforestation, deforestation amounts to 140 Mha in Latin America, 64 Mha in Sub-Saharan Africa and 24 Mha in Pacific Asia between 2010 and 2050 in the reference case. Without pasture intensification and further trade liberalisation, land-use change could cause cumulative emissions of more than 100 Gt CO_2 until the middle of the century, and it will also affect areas that are globally most important for the conservation of biodiversity.

Protected areas and emission pricing are promising strategies to abate the loss of forests and land-use change emissions. The results indicate that even a relatively low CO_2 price could render deforestation economically unattractive and could transform the land-use

sector from a net source of greenhouse gas emissions into a net sink. Analysis of the potential of afforestation shows that a CO_2 price that starts at 30 USD and increases by 5% per year could lead to a forest area expansion of almost 2,600 Mha by the end of the century and a sequestration of about 860 Gt of CO_2. The findings of this thesis also underpin the importance of expanding protected areas to reduce the overall amount of deforestation, especially in tropical biodiversity hotspots.

Fulfilling the growing demand for food while maintaining or even expanding forests as a climate mitigation strategy requires substantial agricultural intensification. In MAgPIE, investments into yield-increasing research and development (R&D) are modelled endogenously, and all simulation relied on this option. Especially when afforestation competed with agricultural production for the same areas, high yield increases were necessary in tropical developing countries. These findings highlight the importance of the intensification of pasture areas, which has often been neglected. African pasture intensification seems particularly key to limit the conversion of forests and other natural vegetation.

This thesis also shows that besides offering a large CO_2 removal potential, afforestation can limit the regional self-sufficiency in food production and could have severe consequences for food prices. Global afforestation efforts following the introduction of a price on carbon emissions, as assessed in one of the studies, could result in a four-fold global food price level.

The results also highlight the importance of agricultural trade, which is a main determinant of the future of land use and of the effectiveness of land-use policies. This thesis demonstrates that further trade liberalisation may boost deforestation, especially in Latin America. In this region, it also lowers the land-saving effect of agricultural intensification since higher productivity results in higher exports. On the positive side, trade liberalisation may be essential to curb food price hikes that are associated with regional afforestation.

Altogether, this thesis shows that the future of land use is open. Failure to implement strict policies bears the risk of continued agricultural expansion and tropical deforestation with severe consequences for biodiversity and climate. However, the results also suggests that effective political instruments are available that could reduce certain adverse environmental and social impacts of agricultural production.

Zusammenfassung

Globaler Landnutzungswandel ist ein Haupttreiber von zwei der größten Umweltveränderungen die das menschliche Wohlergehen bedrohen: Klimawandel und der Verlust der Biodiversität. Während den letzten Jahrzehnten hat eine wachsende und wohlhabendere Bevölkerung nach mehr landwirtschaftlichen Produkten verlangt, und diese Nachfrage wurde durch Expansion landwirtschaftlicher Flächen und Intensivierung der Produktion bedient. Das hatte jedoch Konsequenzen. Indem Felder und Weiden tropische Wälder ersetzten, wurden einige der artenreichsten Habitate zerstört, und da tropische Wälder zu den Ökosystemen mit der höchsten Kohlenstoffdichte gehören, trug ihr Verlust auch erheblich zum globalen Klimawandel bei.

Der Pfad den die Landnutzung in Zukunft einschlagen wird bleibt ungewiss, aber mit dem vorhergesagten Anstieg der globalen Bevölkerung und der damit einhergehenden verstärkten Nachfrage nach Lebensmitteln, zusätzlich zu neu auftauchenden Ansprüchen nach landwirtschaftlicher Energieproduktion und landbasierter Eindämmung des Klimawandels, wird der Druck auf nutzbares Land wahrscheinlich steigen. Diese Doktorarbeit entwirft Szenarien zukünftiger Landnutzung, identifiziert wahrscheinliche zukünftige Entwicklungen, und analysiert politische Strategien die helfen könnten Landnutzung in eine umweltfreundlichere Richtung zu lenken.

Wälder und die Frage wie ihre zukünftige Ausdehnung Klima und Biodiversität beeinflussen sind zentral in dieser Doktorarbeit. Sie evaluiert die politischen Maßnahmen eines Preises auf Treibhausgasemissionen aus Landnutzungsänderungen und eine Ausweitung von Naturschutzgebieten, sowie die Auswirkungen von Agrarhandelsliberalisierung und landwirtschaftlicher Intensivierung. Die vorliegende Arbeit versucht das voraussichtliche Ausmaß tropischer Entwaldung bis zu Mitte des Jahrhunderts konkreter zu bestimmen, und untersucht Maßnahmen diese zu verringern. Zudem konzentriert sie sich insbesondere auf tropische Biodiversitätshotspots und Methoden diese durch die Ausweisung von Schutzgebieten und landwirtschaftliche Intensivierung zu erhalten. Darüber hinaus berücksichtigt sie das Kohlenstoff-Abscheidungspotential von großräumiger Aufforstung und deren potentiellen Einfluss auf Lebensmittelpreise.

Die Studie nutzt ein globales ökonomisches Landnutzungsmodell um mögliche zukünftige Entwicklungen zu untersuchen. MAgPIE (Model of Agricultural Production and its Impact on the Environment) erzeugt globale Landnutzungsmuster für eine gegebene Nachfrage nach Agrarprodukten durch die Minimierung landwirtschaftlicher Produktionskosten. Um die vorliegenden Forschungsfragen zu untersuchen, wurde das Modell hinsichtlich verschiedener Aspekte modifiziert und erweitert.

Diese Doktorarbeit zeigt, dass die angenommene Steigerung der Nachfrage nach Agrarprodukten mindestens bis zur Mitte des Jahrhunderts zu einer Expansion der Ackerflächen führen wird. Mehr als 400 Mha Ackerfläche könnten zwischen 2015 und 2050 neu entstehen. Dies geht mit großer Wahrscheinlichkeit mit dem fortlaufenden Verlust von Wäldern und anderer natürlicher Vegetation einher, sollten dem keine geeigneten politischen Maßnahmen entgegen gesetzt werden. In einer Studie die sich auf die tropische Entwaldung fokussiert, beläuft sich die Entwaldung zwischen 2010 und 2050 im

Referenzfall auf 140 Mha in Lateinamerika, 64 Mha in Subsahara-Afrika und 24 Mha in Asien-Pazifik. Ohne Intensivierung der Weide und ohne Handelsliberalisierung, könnte Landnutzungswandel bis zur Mitte des Jahrhunderts mehr als 100 Gt CO_2 verursachen, und wird dabei auch jene Gegenden beeinträchtigen die weltweit am wichtigsten für den Erhalt der Biodiversität sind.

Naturschutzgebiete und die Bepreisung von Emissionen sind erfolgversprechende Strategien um den Verlust von Wäldern und Landnutzungsemissionen zu verringern. Die Ergebnisse deuten an, dass bereits ein relativ geringer CO_2-Preis Entwaldungen ökonomisch unattraktiv machen könnte, und den Landnutzungssektor von einer Netto-Quelle zu einer Senke von Treibhausgasemissionen wandeln könnte. Eine Analyse des Aufforstungspotentials zeigt, dass ein CO_2-Preis der bei 30 USD startet und um 5% pro Jahr steigt bis zum Ende des Jahrhunderts zu einer Ausdehnung der Waldfläche um fast 2600 Mha und einer Sequestrierung von etwa 860 Gt CO_2 führen könnte. Die Ergebnisse dieser Dissertation untermauern auch die Bedeutung der Ausweitung von Schutzgebieten um das Ausmaß der Entwaldung, insbesondere in tropischen Biodiversitätshotspots, zu verringern.

Um die steigende Nachfrage nach Lebensmitteln zu bedienen, während Wälder erhalten oder als Klimaschutzmaßnahme sogar ausgeweitet werden, bedarf es erheblicher landwirtschaftlicher Intensivierung. In MAgPIE werden Investitionen in Forschung und Entwicklung endogen modelliert, und alle Simulationen beruhten auf dieser Option. Insbesondere wenn Aufforstung mit landwirtschaftlicher Produktion um die gleichen Flächen konkurrierte waren starke Ertragssteigerungen in tropischen Entwicklungsländern nötig. Diese Ergebnisse unterstreichen die Wichtigkeit einer Intensivierung des Weidelands, welche bisher oft vernachlässigt wurde. Eine Intensivierung afrikanischer Weideflächen erscheint besonders zentral zu sein um die Umwandlung von Wäldern und anderer natürlicher Vegetation einzuschränken.

Diese Arbeit zeigt auch, dass Aufforstungen neben dem zur Verfügung stellen eines großen Kohlendioxid-Abscheidungspotentials, die regionale Selbstversorgung mit Lebensmitteln einschränken und erheblichen Einfluss auf Lebensmittelpreise haben können. Weltweite Aufforstungsbestrebungen die der Einführung eines Preises auf Kohlenstoffemissionen folgen, wie es in einer der Studien untersucht wurde, könnten zu einem viermal so hohen globalen Lebensmittelpreisniveau führen.

Die Ergebnisse verdeutlichen auch die Bedeutung des Agrarhandels, der eine bestimmende Größe für die Zukunft der Landnutzung und die Effektivität von Landnutzungspolitik ist. Diese Arbeit zeigt, dass weitere Handelsliberalisierung insbesondere in Lateinamerika die Entwaldung beschleunigen könnte. In dieser Region verringert sie zudem den flächensparenden Effekt von landwirtschaftlicher Intensivierung, da höhere Produktivität zu höheren Exporten führt. Positiv zu bewerten ist, dass Handelsliberalisierung wesentlich zur Begrenzung von Lebensmittelpreissteigerungen im Zusammenhang mit regionalen Aufforstungen beitragen könnte.

Insgesamt zeigt die vorliegende Arbeit, dass die Zukunft der Landnutzung offen ist. Ein Versäumnis konsequente Politik umzusetzen birgt das Risiko fortschreitender landwirtschaftlicher Expansion und tropischer Entwaldung mit einschneidenden Konsequenzen für Biodiversität und Klima. Die Ergebnisse zeigen jedoch auch, dass effektive politische Instrumente zur Verfügung stehen, die bestimmte negative soziale und ökologische Auswirkungen der landwirtschaftlichen Produktion verringern könnten.

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1. Long history of human modification of their environment

Human activity has been altering the environment and shaping the planet for millennia, and it will continue to do so in future. It is now significantly influences many major components of the Earth's system (Vitousek *et al.*, 1997). This strong interference by humans in natural systems began long before the Industrial Revolution. At least since the shift from hunting and gathering to agriculture, humans have strongly modified landscape and climate. Approximately 8,000 years ago, CO_2 concentrations in the atmosphere began to differ from that which natural processes could explain, and the same phenomenon was observed for CH_4 levels at about 5,000 BP. These changes in atmospheric composition respectively coincided with the start of agriculture and forest clearances in Eurasia and the use of irrigated rice (Ruddiman, 2003). Since the onset of the Industrial Revolution, the pace of this change has significantly increased. For example, CO_2 concentration has risen by more than 40% compared to the level in 1750 (Hartmann *et al.*, 2013), and cropland area has increased from about 3 or 4 million km^2 in 1700 (Ramankutty & Foley, 1999a; Pongratz *et al.*, 2008; Klein Goldewijk *et al.*, 2011) to about 14.1 million km^2 today (FAOSTAT, 2016a). The invention of the Haber-Bosch process in the early 20th century revolutionised agricultural fertilisation but has significantly altered the nitrogen cycle. Today, about 113.3 Tg of N are produced as nitrogen fertilizers per year (FAOSTAT, 2017), which is approximately double the estimated pre-industrial rate of biological N fixation (Vitousek *et al.*, 2013).

1.1. The Anthropocene and planetary boundaries

Human modifications of the Earth's system have become so profound that they have prompted the call for a new geological epoch: the Anthropocene (Crutzen & Stoermer, 2000). Irrespective of an on-going debate about when precisely the Anthropocene started (Crutzen & Steffen, 2003; Ruddiman, 2003, 2013; Lewis & Maslin, 2015) – at some point between a few thousand years ago with larger deforestation activities or only in 1964 with a ^{14}C peak of nuclear bomb testing – it is now widely recognised that human-caused changes to the pedosphere, atmosphere, hydrosphere and biosphere are immense. Steffen *et al.* (2007) put is like this: 'Human activities have become so pervasive and profound that they rival the great forces of Nature and are pushing the Earth into planetary *terra incognita*' (emphasis in original). Of course, these human interventions have not been without consequences.

Rising anthropogenic greenhouse gas emissions have increased radiative forcing, led to higher temperatures and triggered a melting of glaciers and a thermal expansion of oceans that have caused a sea level increase (IPCC, 2013). The conversion of natural ecosystems to human-dominated landscapes has benefited many, but it has been mirrored by losses of biodiversity and a degradation of several ecosystem services (MEA, 2005). Rockström *et al.* (2009) have argued that human activities have now reached a level that could shift the Earth's system out of favourable conditions and define 'planetary boundaries' as the safe operating space for humanity. These planetary boundaries have already been exceeded for land-system change and climate change and for biodiversity and the biochemical flows of phosphorus and nitrogen (Steffen *et al.*, 2015).

1.2. Climatic change

Climate change is a major modification of the Earth's system and one of the most substantial challenges of the 21st century. The sustainable development goals (SDGs), for instance, refer to climate change as 'the single biggest threat to development' (UN, 2015). There is now an overwhelming scientific consensus, especially among climatologists publishing on this issue, that anthropogenic climate change is a reality (Oreskes, 2004; Doran & Zimmerman, 2009; Cook *et al.*, 2013, 2016).

Today, there is hardly anything – from the oceans to the highest mountaintops – has been unaffected by higher greenhouse gas concentrations or the changing environmental conditions that accompany them. Higher CO_2 concentrations in the atmosphere have been partly buffered by an uptake of CO_2 by the oceans (Le Quéré *et al.*, 2009), which mitigates the temperature rise but also leads to ocean acidification (Rhein *et al.*, 2014). Globally, mountain glaciers are melting due to increasing temperatures (Roe *et al.*, 2016).

These changed environmental conditions will eventually have severe consequences for human well-being. Ocean acidification together with elevated temperatures leads to coral bleaching (Hughes *et al.*, 2003), which affects coral tourism, fishing and coastal protection against storms (Harvell *et al.*, 2008). Rising sea levels, which are mainly a consequence of thermal ocean expansion and melting glaciers, pose high flooding risks for hundreds of millions of people (Church *et al.*, 2013; Wong *et al.*, 2014). Melting glaciers also affect river run-offs, which many people rely on for drinking water and agricultural irrigation (Chevallier *et al.*, 2011). Climate change also severely impacts human health (McMichael *et al.*, 2006), agricultural production (Rosenzweig *et al.*, 2014; Cohn *et al.*, 2016), biodiversity (Bellard *et al.*, 2012), freshwater ecosystems (Woodward *et al.*, 2010) and economic productivity and supply chains (Kjellstrom *et al.*, 2009; Levermann, 2013).

Climatic change is caused by a number of drivers. CO_2 is the single most important greenhouse gas, and its emissions are responsible for about three-quarters of all 100-year global warming potential (GWP)-weighted anthropogenic greenhouse gas emissions (Blanco *et al.*, 2014). CO_2 emissions mostly result from fossil fuel combustion and cement production (about 9.3 Gt C yr^{-1} during the last decade) and net emissions from land-use change (1 Gt C yr^{-1}) (Le Quéré *et al.*, 2016). Among non-CO_2 greenhouse gases, CH_4 is the most abundant in the atmosphere and mainly stems from agriculture, fossil fuel exploitation and wetlands (Montzka *et al.*, 2011).

1.3. Declining biodiversity

Biodiversity is declining at alarming rates. The extinction of species has now reached a level that is between 100 and 1,000 times higher than the typical background extinction rates in Earth's history (MEA, 2005; De Vos *et al.*, 2015). An interplay of several drivers affects biodiversity, but land-use change and the expansion of agriculture are a main cause of the on-going decline in biodiversity (Oliver & Morecroft, 2014). In addition, biological diversity is affected by climate change, through long-term trends (e.g. annual precipitation) and extreme events (e.g. droughts) (Jentsch *et al.*, 2007).

Land-use and related pressures have already significantly reduced biodiversity around the world (Newbold *et al.*, 2015). The planetary boundary for biodiversity – defined as a reduction in local abundance of less than 10% – has been transgressed on more than 50% of terrestrial land (Newbold *et al.*, 2016). Nevertheless, assessments of future biodiversity change have largely neglected land use and have more strongly focused on the impact of climate change (Titeux *et al.*, 2016). Continuous threats to biodiversity are also foreseen for the future. Without further policy interventions, the combined effects of climate change and land-use change will increase the risk of further species extinctions and may lead to a decline in species abundance (Jetz *et al.*, 2007; Visconti *et al.*, 2016).

Biodiversity is not only a value as such but also profits human well-being and is essential for ecosystem functioning (MEA, 2005; Sandifer *et al.*, 2015). It has positive effects on the provision of ecosystem services (Balvanera *et al.*, 2006) and acts as an insurance for risk-averse ecosystem managers (Baumgärtner, 2007). Higher diversity generally helps to increase ecosystem productivity and stability (Cardinale *et al.*, 2012), while species losses could significantly decrease productivity and represent a major driver of ecosystem change (Hooper *et al.*, 2012). Greater biological diversity also profits human health, and plant diversity offers immense resources for the development of new pharmaceuticals (Fowler, 2006; Herndon & Butler, 2010; Hough, 2014).

2. Land-use change

Land use and land-use change are not innately problematic, but they are a basis for human well-being. Yet today, they often stand for the conversion of pristine, natural ecosystems with high biodiversity into agricultural monocultures, and they are responsible for a significant share of total anthropogenic greenhouse gas emissions.

In 1700, nearly half of the terrestrial biosphere was without substantial land use, but agriculture and urban settlements increased from 5% to 39% of total ice-free land until the year 2000 and reduced wilderness areas to about one-quarter of its former area (Ellis *et al.*, 2010). In the so-called 'developed world' of regions such as Europe and North America, but also in China, most of this land conversion from forests and other natural vegetation to agriculture occurred before 1900 (Ramankutty & Foley, 1999b; Williams, 2000; Smith *et al.*, 2004; He *et al.*, 2008; Kaplan *et al.*, 2009; Liu & Tian, 2010), while the hotspots of current land-use change and deforestation activities are found in tropical, low-income counties (Keenan *et al.*, 2015). This is in line with the assumption that forest area first declines alongside with economic development until it eventually starts to recover (Rudel *et al.*, 2005). However, deforestation in the developing world is also partly driven by demand from the developed world, and traded agricultural products embody parts of the deforestation (European Commission, 2013; Karstensen *et al.*, 2013; Henders *et al.*, 2015).

Today, about 4,800 Mha are covered by some form of agricultural land use; roughly one-third of this is cropland (1,400 Mha), while the remaining two-thirds are permanent pastures and meadows (FAOSTAT, 2016a). This already accounts for most of the 5,400 Mha hectares of land that are suitable for agricultural production outside protected areas and dense forests. So, with the exception of Africa, the potential for further agricultural expansion is strongly limited (Zabel *et al.*, 2014).

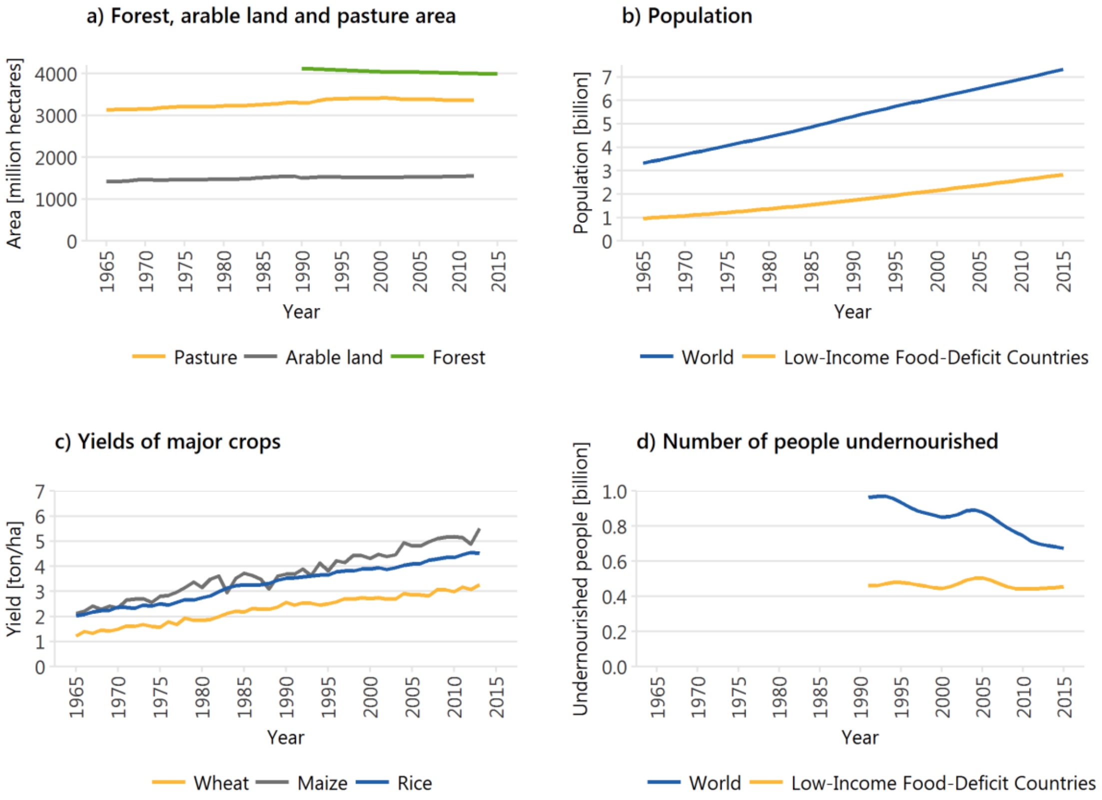

Figure 1 Major trends relevant to land use: a) Forest, arable land and pasture area; b) Population; c) Yields of major crops and d) Number of undernourished people
sources FAOSTAT (2016b) and FRA (2015)

The story of land use, however, is not only one of declining natural vegetation and increasing emissions but also one of successes. While the global population has increased from approximately 3 billion 50 years ago to more than 7 billion today, the number of undernourished people (i.e. those consuming less than their minimum dietary energy requirement) has diminished (Figures 1b and 1d). Data from the period of 1990 to 2015 reveals that numbers have decreased from 960 million to 670 million, which is mainly a result of progress in China and India (FAO, IFAD and WFP, 2015). However, in the 62 countries that are classified as 'low-income food-deficient' (per capita gross national income less than 1,915 USD and gross imports greater than gross exports), the rate of undernourishment remained critically high.

2.1. Demand for food as main driver of land-use change

The growing demand for agricultural products is a main driver of land-use change and is responsible for the expansion of agriculture. It has two main underlying drivers: population growth and a dietary change towards a more livestock product-based consumption as a result of growing incomes. Livestock production was responsible for most of the increase in agricultural production areas in recent decades. The area dedicated to producing feed has increased as much as the area that is utilised for producing vegetal crops for human consumption (140 Mha) since 1961, and an additional 300 Mha of pasturelands were needed to feed growing livestock herds (Alexander *et al.*, 2015). These two drivers – higher livestock shares in consumption and higher population numbers – are also projected to cause continuous increases in produced calories in the future and will increase the pressure on land (Alexandratos & Bruinsma, 2012; Bodirsky *et al.*, 2015).

Additional drivers will likely become more critical in the future and incite an even stronger competition for land (Smith *et al.*, 2010; Haberl, 2014). Many projections of future energy production foresee a strong reliance on bioenergy, especially in combination with carbon capture and storage (CCS) (Edmonds *et al.*, 2013; Klein *et al.*, 2014; Kriegler *et al.*, 2014a; Rose *et al.*, 2014; Bauer *et al.*, 2015). As the production potential from residues and on marginal and abandoned lands is limited (Campbell *et al.*, 2008; Xue *et al.*, 2016; Emery *et al.*, 2017) and unlikely to be fully exploited (Bryngelsson & Lindgren, 2013), the production of bioenergy is going to compete with other uses. Especially under ambitious climate mitigation, bioenergy production could occupy as much as 1,500 Mha, which is more than the current area of cropland (Popp *et al.*, 2017).

Another reason for the increasing pressure is the decrease in agricultural land due to urban sprawl (Hennig *et al.*, 2015), the loss of agricultural land due to soil erosion (Bakker *et al.*, 2005; Rey-Benayas *et al.*, 2007) and desertification (Geist & Lambin, 2004), salinisation (Halvorson *et al.*, 2003) and the submergence of areas due to rising sea levels (Dwarakish *et al.*, 2009; Ramasamy *et al.*, 2010).

2.2. Land use defined by land-use intensity

Land use represents more than the type of activity on one piece of land (e.g. grazing, wheat production) and includes the intensity of production. Yield levels are a main determinant of the area needed to produce a certain amount of produce and thus also of agricultural expansion under growing food demand. Agricultural intensity is usually defined either as yields (production per area and time) or as the amount of inputs, such as labour and capital, that agricultural production uses (Erb *et al.*, 2013).

Another way to look at land-use intensity is through a comparison of the share of net primary production (NPP) that remains after removals through land use to the NPP of potential natural vegetation (Haberl *et al.*, 2014). The global average of this human appropriation of net primary production (HANPP) has been estimated to be about 25% of the NPP of potential vegetation (Haberl *et al.*, 2007; Krausmann *et al.*, 2013). Moreover, the NPP of croplands was approximately 77% of the estimated potential of natural vegetation in 2000. In some locations, cropland productivity exceeded natural productivity by more than a factor of 10 as achieved through the removal of water and nutrient limitations by fertilisation and irrigation (Niedertscheider *et al.*, 2016). Following the definition of HANPP, agricultural intensification can also comprise the transformation of non-agricultural habitats to agricultural habitats (Firbank *et al.*, 2008). Regardless, in this thesis, I refer to a narrower definition of agricultural intensification that focuses on processes in landscapes that are already under agricultural use.

Over recent decades, agricultural intensification has been key for production increases. Globally, higher production has mainly been achieved through yield increases much more than through increases in area (sees also Figures 1a and 1c). While total production has increased by 145% since 1960, agricultural area expanded by only 11%, leaving most of the production increase to yield increases (Pretty, 2008). In fact, yields of major crops have increased linearly over the last 20 years at a rate of 1.0% p.a. for wheat and rice and 1.5% for maize (Fischer *et al.*, 2014). These yield increases have resulted in decreasing per-capita land areas in most countries (Alexander *et al.*, 2015).

Further intensification will be critical to feed the growing population and to avoid or limit future agricultural expansion. Erb *et al.* (2016) show that deforestation could be avoided – even under an adoption of Western diets – if crop yields increased significantly. However, this can only be achieved through better management and continuous technological progress, and it will require higher investments in agricultural R&D (Fischer *et al.*, 2014). Furthermore, climate change can potentially increase yields through CO_2 fertilisation, but it can also present negative effects through the gradual change in temperature and precipitation and the frequency of extreme events (Anwar *et al.*, 2013).

2.3. Emissions from land use and agriculture

Land use and agriculture are responsible for a large share of anthropogenic greenhouse gas emissions. Between 1990 and 2010, net emissions of land use and land-cover (LULC) change accounted for 1.1 Pg C yr^{-1} and thus for about 12.5% of anthropogenic carbon emissions (Houghton *et al.*, 2012) and 10% of all greenhouse gas emissions (Tubiello *et al.*, 2015); however, these figures remain highly uncertain (Blanco *et al.*, 2014) and could be substantially underestimated (Arneth *et al.*, 2017). These land-use emissions mainly result from tropical deforestation and evidence a slight decline over the past decades (Houghton, 2013). In contrast, emissions from agriculture have continuously grown by about 1% per year and now represent the larger share of all agriculture, forestry and other land-use (AFOLU) emissions, with about 11.2% of total greenhouse gas emissions (Tubiello *et al.*, 2015). Agriculture is responsible for the bulk of CH_4 (mainly from ruminants and rice production) and N_2O emissions (fertilisation) (Montzka *et al.*, 2011) – two greenhouse gases with a much larger GWP than CO_2 (Myhre *et al.*, 2013).

2.4. Land-based carbon dioxide removal

Although responsible for a large amount of greenhouse gas emissions, and thus contributing to further global warming, terrestrial land could also factor into the solution and mitigation of climatic change. Today, terrestrial ecosystems already represent a sink of carbon and remove CO_2 from the atmosphere, though the precise amount of this flux is uncertain. It is usually calculated as the difference between other known terms of the carbon budget, such as fossil fuel emissions and atmospheric CO_2 concentrations (Le Quéré *et al.*, 2016). It is estimated that the terrestrial land removes more than 3 Pg C yr^{-1}; however, since emissions by land use might be underestimated, this sink could also be considerably larger (Le Quéré *et al.*, 2016; Arneth *et al.*, 2017).

The uptake of carbon by living organisms via photosynthesis limits the theoretical potential of biological carbon dioxide removal (CDR) is limited by the uptake of carbon by living organisms through photosynthesis. Global terrestrial net primary productivity accounts to about 56 Pg C yr^{-1} (Ito, 2011), with a positive trend in the past 150 years as a result of climatic change (Li *et al.*, 2014). This is considerably higher than yearly anthropogenic CO_2 emissions through industrial processes of about 9 Pg C yr^{-1} (Le Quéré *et al.*, 2016), which implies that sequestering and constantly removing only a fraction of the plant biomass production from the atmosphere could offset industrial emissions. In reality, however, the mitigation potential is much lower because humans already harvest around 8 Pg C yr^{-1} of biomass (Haberl *et al.*, 2007). Yet, terrestrial CDR plays a prominent role in many scenarios of a climate-neutral future. For instance, most scenarios of the fifth assessment report (AR5) of the Intergovernmental Panel on Climate Change (IPCC) that reach a warming of below 2°C at the end of the century include the use of plant-based CDR technologies, such as bioenergy with carbon capture and storage (BECCS) or afforestation (Clarke *et al.*, 2014; Fuss *et al.*, 2014).

3. Forests

Forests are central to the Earth's climate system and offer a habitat to many species; however, they are threatened by land-use change which causes the global forest area to shrink (see Figure 1a) (FAO, 2016). Today, forests cover about 4,000 Mha, or approximately 30% of the global land area. Even though rates of deforestation have recently decreased, forest area has declined by 3% over the last 15 years (Keenan *et al.*, 2015). About one-third of the total forest area is primary forest, and the primary forests of the tropics in particular have undergone a substantial loss of 10% in the same period (Morales-Hidalgo *et al.*, 2015).

Tropical deforestation is the result of certain underlying driving factors, such as economic development, institutions and national policies which translate into proximate causes of deforestation (e.g. expansion of agriculture and infrastructure, wood extraction) (Geist & Lambin, 2002). Agriculture is the most prominent cause of deforestation and is responsible for about two-thirds of the total deforested area. In Latin America, commercial agriculture has a dominant role, while in Africa and Asia, subsistence farming is still prevalent (Hosonuma *et al.*, 2012; Kissinger *et al.*, 2012). More specific causes also vary between the regions. While cattle and soy production are predominant reasons for deforestation in Latin America, palm oil is the main driver in Indonesia and Malaysia. Timber production is another reason for deforestation, especially in Southeast Asia, which houses commercially interesting tree species. In Latin America and Africa, land is primarily cleared to make space for other land uses (Boucher *et al.*, 2011).

Notably, forest areas are not decreasing in all parts of the world (Chazdon, 2008). Temperate forest and other wooded land area has constantly increased in the last 25 years, with the highest rates of increase found in China, Chile and the USA (Keenan *et al.*, 2015). These increases are mainly the result of an immense Chinese afforestation and reforestation effort (Zeng *et al.*, 2015) and forest recovery in the northeast USA (Ramankutty *et al.*, 2010).

The value of forests that are managed for the extraction of timber is quite obvious, but forests – including those not under management – additionally provide many ecosystem services that range from water purification and flood protection to carbon sequestration and recreation (Ninan & Inoue, 2013). One of the main ecosystem services of forests is their carbon sequestration.

3.1. Climatic influence of forests

Forests form the largest terrestrial pool of carbon. In total, the terrestrial biosphere contains about 385 to 650 Pg C in vegetation (Houghton *et al.*, 2009; Ciais *et al.*, 2013). Because of their large extent and high carbon density, forests in Latin America, sub-Saharan Africa and Southeast Asia contain about half of vegetation carbon – an estimated 250 Pg C (Saatchi *et al.*, 2011). However, the highest carbon densities are found in moist temperate forests, such as those in southeast Australia (Keith *et al.*, 2009). Deforestation and forest degradation in the tropics has caused net emissions of about 0.8 to 1.1 Pg C yr^{-1} between 2000 and 2005, which is approximately equal to the net global emissions of land-use change (Harris *et al.*, 2012; Houghton, 2013). Forests in temperate and boreal regions, in contrast, have gained aboveground biomass carbon in recent years (Liu *et al.*, 2015). In sum, as a consequence of the fertilising effect of rising CO_2, N deposition and longer growing seasons, the land sink is believed to be substantially bigger than net emissions from land-use change (Le Quéré *et al.*, 2016).

In addition to the biochemical effect of carbon sequestration, forests also have a biophysical influence, which more directly affects the Earth's radiation balance. Forests usually exhibit a lower albedo than grasslands, and in the case of grassland afforestation, they would reflect less short-wave radiation back to space (Jackson *et al.*, 2008). This effect is especially pronounced in boreal regions, where forests masks areas that would otherwise be covered with snow with a very high albedo (Bonan, 2008). On the other hand, forests usually evaporate more water than other land uses; this leads to local cooling and, in combination with the release of cloud condensation nuclei, to cloud formation. These clouds prompt an additional cooling effect (Jackson *et al.*, 2008; Spracklen *et al.*, 2008). Whether the biochemical or the biophysical effects prevails depends on the local conditions, but a general latitudinal trend is observed in which tropical forests reveal a clear cooling while in boreal regions the effect is much lower (Bala *et al.*, 2007; Bonan, 2008; Bathiany *et al.*, 2010; Arora & Montenegro, 2011).

3.2. Importance of forest ecosystems for global biodiversity

Biodiversity is not distributed equally around the globe. Broadly speaking, the number of species follows a latitudinal gradient, with highest numbers observed towards the equator (Gaston, 2000; Ceballos & Ehrlich, 2006). Of terrestrial biomes, tropical and sub-tropical moist broadleaf forests have the highest species richness by far (MEA, 2005; Liang *et al.*, 2016). Furthermore, of 25 biodiversity hotspots that feature a high number of endemic plant species and are under high threat, 15 contain tropical forests (Myers *et al.*, 2000). Land-use conversion of primary vegetation represents a particular threat to biodiversity, as species richness is usually higher in primary vegetation than in other land-use types (Newbold *et al.*, 2015). Lower forest cover in the Amazon has evidently reduced bird and mammal species diversity (Ochoa-Quintero *et al.*, 2015) and has altered bird species composition (Morante-Filho *et al.*, 2015). In addition to higher vertebrate diversity in forests compared to other land uses, ant species richness, is much higher in Brazilian forests than in agricultural production areas (Solar *et al.*, 2016). Especially in tropical and sub-tropical ecosystems with the highest numbers of endemic species (Orme *et al.*, 2005;

Lamoreux *et al.*, 2006; Kier *et al.*, 2009), land-use conversion can directly translate to a worldwide extinction of species.

Regardless of their high value for biodiversity, forest ecosystems in many places lack adequate protection against conversion. Globally, about 16% of all forests are reportedly under protection (Morales-Hidalgo *et al.*, 2015). Yet, protection does not necessarily save them from deforestation (Heino *et al.*, 2015), and even if forests are conserved, anthropogenic disturbances may substantially reduce their biodiversity (Barlow *et al.*, 2016).

4. Policies and options

The future development of land use and its impacts on the environment and on society are not predetermined but are subject to political and private interventions. In order to mitigate climate change and avoid further loss of biodiversity, policymakers can rely on numerous options to regulate agricultural production.

The following sections provide an overview of several policy options that could steer land use in such a direction to reduce greenhouse gas emissions as well as their adverse effects on society (e.g. due to rising food prices) and biodiversity. This thesis focuses on only a limited selection of policies that are relevant on the global scale but acknowledges that local policies also affect land use and emissions.

4.1. Agricultural trade liberalisation

Agricultural trade has a strong influence on the agricultural system. The amount of calories that are traded has more than doubled since 1986, and roughly one-quarter of current global food production for direct human consumption is traded internationally (D'Odorico *et al.*, 2014). Moreover, there are on-going efforts to further liberalise agricultural trade (McCorriston *et al.*, 2013). Yet, agriculture is an exception to the general trade liberalisation trend, as the level of protectionism remains higher in agriculture than in other sectors. Measures of protectionism, such as subsidies and tariffs, are motivated by the need to assure access to affordable food and to safeguard the income of farmers (Trebilcock & Pue, 2015). Nevertheless, trade liberalisation is considered 'the best hope for growth and poverty alleviation' for developing countries in which agriculture remains to be the biggest employer (McCalla & Nash, 2007). Trade liberalisation is generally believed to reduce farm prices in industrialised countries while developing countries face higher world market prices, which benefits exporters but adversely affects importing countries (Lutz, 1992). However, price convergence to world market prices has not been fully transmitted to domestic markets (McCorriston *et al.*, 2013), and empirical research has demonstrated that deeper market integration elevates consumer food prices during global price spikes (Flachsbarth & Garrido, 2014). Most importantly, however, according to the theory of comparative advantages, more liberalised trade would lead to a reallocation of production whereby countries specialise in the production of individual goods, which would ultimately reduce prices.

In addition to price effects, agricultural trade offers an option to produce at locations with higher productivity and lower relative emissions, and it could therefore help to reduce overall greenhouse gas emissions (Kreidenweis *et al.*, 2016). It can also contribute to save agricultural water consumption and therefore alleviate water scarcity (Calzadilla *et al.*, 2011; Biewald *et al.*, 2014). Especially with regard to climate change, free trade can gain importance, as it could buffer the effects of long-term trends and extreme events and may decrease agricultural welfare losses and reduce food price volatility (Baldos & Hertel, 2015; Stevanovic *et al.*, 2016).

On the downside, when not accompanied by suitable policies, further trade liberalisation could heighten tropical deforestation and CO_2 emissions (Schmitz *et al.*, 2012). Trade can also be interpreted as a displacement of land use from high-income countries to lower-income regions (Weinzettel *et al.*, 2013). In line with this view, agricultural trade can be linked to substantial species losses (Chaudhary & Kastner, 2016). More liberalised trade could diminish the effectiveness of national environmental policies when negative externalities are moved to other countries instead. Policymakers need to be aware of the manifold consequences that trade liberalisation can have for the agricultural sector.

4.2. Price on land-use change emission

Pricing greenhouse gas emissions is a common assumption in many integrated assessment models of the future (Edenhofer *et al.*, 2010; Kriegler *et al.*, 2014a; Luderer *et al.*, 2016) and would directly target a root cause of climate change. Such emission pricing could lead to substantial emission cuts in the land-use sector as well.

A price on carbon emissions could, for instance, significantly reduce deforestation (Kindermann *et al.*, 2008). However, only taking into account emissions from deforestation might result in a leakage and shift of land-use conversion to non-forest ecosystems, which would still cause substantial carbon emissions (Popp *et al.*, 2014a). A carbon price could also provide an incentive for forest expansion and large-scale production of bioenergy (Wise *et al.*, 2009; Humpenöder *et al.*, 2014). Pricing of agricultural and land-use change emissions at the same level as in the energy sector is also the shared climate policy assumption of the shared socio-economic pathways (SSPs), whereby it reduces CO_2, CH_4 and N_2O emissions (Kriegler *et al.*, 2014b; Popp *et al.*, 2017).

There is currently no uniform, global carbon tax, and none is in sight. However, there are several multinational, national or sub-national cap-and-trade systems or carbon taxes (e.g. emission trading schemes in the European Union and California and a tax in Sweden) (World Bank Group & ECOFYS, 2016). Some of these emission trading systems are also linked directly or indirectly, and plans to further strengthen these linkages exist (Ranson & Stavins, 2016). However, such systems usually do not cover agriculture and land use, although different proposals have been made to include agriculture into the EU Emission Trading Scheme, for instance (Grosjean *et al.*, 2016). The UNFCCC debate specifically focuses on avoiding emissions from deforestation and forest degradation (REDD). Unlike a tax on carbon emissions, the proposed approach offers payments to countries that reduce their emissions from deforestation to below a pre-defined baseline (Strassburg *et al.*, 2009). Even though emission pricing is not in place on a global level, its inclusion in scenarios of future land use is a useful assumption, as it reveals the potential of such a policy.

4.3. Large-scale afforestation

Many studies have emphasised the need for CDR in order to limit global warming to 2°C or 1.5°C (Edmonds *et al.*, 2013; van Vuuren *et al.*, 2013; Fuss *et al.*, 2014; Kriegler *et al.*, 2014a; Rogelj *et al.*, 2015). However, concerns have been raised that negative emission technologies, such as BECCS, have not achieved maturity and should therefore not be relied on (Fuss *et al.*, 2016; Vaughan & Gough, 2016). Afforestation, in contrast, is a low-cost and low-tech mitigation strategy (Smith *et al.*, 2015). Given the right incentives, it could be relatively easy to implement worldwide – including in developing countries – and has the potential to remove a significant amount of CO_2 from the atmosphere (Humpenöder *et al.*, 2014).

On the downside, the establishment of forests always involves the risk that these forests will be cut down again or lost through natural disturbances, which would incite a sudden release of greenhouse gases (Galik *et al.*, 2016). Since the carbon stocks in old-grown forests take several decades to accumulate, absolute priority should be given to the avoidance of deforestation. Afforestation will increase the competition for land, and could thereby prompt an increase in food prices (Calvin *et al.*, 2014). Lastly, due to its effect on surface albedo, afforestation of northern latitudes could even increase global warming (Bala *et al.*, 2007; Bonan, 2008; Bathiany *et al.*, 2010; Arora & Montenegro, 2011).

Afforestation is achievable through emission pricing but could also result from other policy instruments. Currently – and justifiably –international policy concentrates far more on reducing emissions from deforestation and forest degradation (REDD) than on afforestation and reforestation. However, certain afforestation projects are in progress within the Clean Development Mechanism (CDM) of the Kyoto protocol, whereby developed countries can offset a limited amount of their emissions through afforestation projects in countries that are not bound to emission targets (Pearson *et al.*, 2006; Schlamadinger *et al.*, 2007; Corradini *et al.*, 2016). Carbon credits from Land Use, Land-Use Change and Forestry (LULUCF), and therefore also from afforestation, are explicitly excluded from the EU Emission Trading Scheme, so that the potential for afforestation is not fully exploited (Ellison *et al.*, 2014).

4.4. Designation of protected areas

Protected areas are a cornerstone of biodiversity conservation (Gaston *et al.*, 2008). Today, more than 155,000 protected areas cover 12.5% of the terrestrial realm (Watson *et al.*, 2014). Many studies have shown that protected areas help to reduce land conversion and the loss of important species habitats (Andam *et al.*, 2008; Beresford *et al.*, 2013; Heino *et al.*, 2015). Yet, protected areas vary in their effectiveness and are not distributed perfectly. Currently, 17% of all threatened birds, mammals and amphibians are not found in a single protected area (Venter *et al.*, 2014). The distribution of protected areas is also biased towards locations that are less likely to face land conversion (Joppa & Pfaff, 2009). As a result, the effectiveness of protection is easily overestimated, and protected areas closer to roads and cities would probably be more effective (Joppa & Pfaff, 2011). In general, protected areas offer no guaranteed prevention of land-use change. Especially in Asia, protected areas have been found to offer only limited protection against forest losses

(Spracklen *et al.*, 2015). Protected areas are also not guaranteed to exist forever, as they are occasionally subject to protected area downgrading, downsizing and degazettement (PADDD) (Mascia *et al.*, 2014; Symes *et al.*, 2016). In some regions, multi-use protected or indigenous areas can be more effective in preventing fire-related deforestation than strict protection (Nelson & Chomitz, 2011).

Within the Convention on Biological Diversity, governments agreed on the Aichi targets, which promote expansion of protection to at least 17% of the terrestrial area (CBD, 2010). A co-ordinated protected area network expansion that reaches this target could triple the average protection of species ranges, but it will soon become unattainable if projected land-use changes occur (Pouzols *et al.*, 2014). One problem is that greater effectiveness of protected areas – i.e. a higher number of species covered – implies more extensive agricultural opportunity costs (Venter *et al.*, 2014). As a co-benefit, protected areas can also reduce land-use emissions that result from deforestation (Scharlemann *et al.*, 2010; Soares-Filho *et al.*, 2010).

4.5. Agricultural intensification

Agricultural intensification has been the main source of higher production in recent decades and could also help to reduce deforestation and combat losses of other natural vegetation. Beginning in the 1960s, successes in crop research were translated to the conditions of the developing word. This so-called 'Green Revolution' led to yield increases of over 200% for wheat and over 150% for maize between 1960 and 2000 in developing countries (Pingali, 2012). A frequent argument is that the Green Revolution has saved millions of hectares of land from being converted into croplands (Borlaug, 2007; Stevenson *et al.*, 2013). This offers a perfect case for further investment in yield-increasing technology.

A potential for further yield increases remains. Yield gaps refer to the difference between average, realised yields and potential yields (yields of adapted crops grown without limitations from water, nutrients or diseases) (Lobell *et al.*, 2009). Realised maize yields in Africa are, for instance, only 20% of the potential (Lobell *et al.*, 2009), and rain-fed wheat has attained less than 50% of its potential in Russia (Schierhorn *et al.*, 2014). It is estimated that closing these yield gaps – mainly by removing nutrient limitations – could increase the production of major crops by 45% to 70% worldwide (Mueller *et al.*, 2012). On the global level, it could be possible to achieve higher yields with the same amount of nitrogen fertiliser if it were spatially redistributed (Mueller *et al.*, 2014). Additionally, for pastures, present-day technology could enhance productivity. Only about 7% of herbage that is grown on managed grasslands is used as forage (Chang *et al.*, 2016). Especially in developing regions, such as Southeast Asia, Latin America and Sub-Saharan Africa, current herbage-use efficiencies are low, which thus represents a huge potential to increase ruminant production (Herrero *et al.*, 2013; Strassburg *et al.*, 2014; Chang *et al.*, 2016; Fetzel *et al.*, 2016). However, in general, and especially for grazing systems, land-use intensity datasets are scarce on the global level (Kuemmerle *et al.*, 2013).

A new Green Revolution in regions with low yields, such as sub-Saharan Africa, will likely require a package approach that involves investments in agricultural R&D, transport infrastructure, irrigation and the development of fertiliser markets (Rashid *et al.*, 2013). However, especially in many African countries, spending on agricultural R&D has decreased in recent years (Beintema *et al.*, 2012). Past developments also suggest that higher levels of governance are necessary in order to achieve production increases through intensification (Mandemaker *et al.*, 2011).

It is worth noting that intensification alone does not ensure land saving. Kaimowitz and Angelsen (2008) assume that farmers would still seek to increase their profits under higher productivity, and they would therefore expand production areas. Many empirical studies have attempted to assess the land-saving effect of intensification. For instance, one recent study by Barretto *et al.* (2013) has evaluated the occurrence of intensification and expansion in Brazil. They found that in consolidated agricultural areas, where land is constrained, yields increase and pasture areas decline. At the agricultural frontier, where land is abundant and inexpensive, intensification also coincides with expansion. So, in order to save land, intensification probably requires combination with other measures, such as strict protection.

On the downside, intensification can have detrimental environmental side effects and can reduce biodiversity (Firbank *et al.*, 2008). Higher intensity production has been found to lower plant and bird diversity in Europe, for instance (Flohre *et al.*, 2011). Grassland intensification can also decrease plant species richness (Allan *et al.*, 2015), alter biological composition by negatively affecting certain species and favouring others (Lush *et al.*, 2014) and can cause homogenisation and reduction of β-diversity (Gossner *et al.*, 2016). Sustainable intensification has been suggested to overcome the adverse negative environmental impacts of yield increases (Pretty & Bharucha, 2014). The term 'sustainable intensification', as such, is vaguely defined and often employed as a buzzword without clarification of its departure from current agricultural practices (Tittonell, 2014; Petersen & Snapp, 2015). Nevertheless, a variety of alternative agriculture approaches could help to diminish the contradiction between agricultural production and the conservation of biodiversity that exists under present-day, high-intensity production systems (Perfecto & Vandermeer, 2010; Chappell & LaValle, 2011), and examples where higher yields did not affect biodiversity offer hope that this could materialise (Clough *et al.*, 2011).

5. Modelling an unknown future

Land-use models have a long history of use for understanding and optimising land-use patterns, assessing consequences of land use and making predictions. Yet, even the history of land use over the last centuries remains relatively unknown, and earlier estimates are more uncertain (Klein Goldewijk & Verburg, 2013). Knowledge for the future is even more limited, and models can likely provide merely educated guesses. Still, there is great interest in modelling future land use – and for good reasons.

In von Thünen's *Der isolierte Staat* (The Isolated State), which was first published in 1842, he reflects on optimal land-use patterns. He concludes that in an idealised landscape, and in order to maximise profits, circular zones of certain land use would evolve around a city according to distances to the market, the prices of goods and land rents (von Thünen, 1910; Grotewold, 1959). State-of-the-art land-use models build upon these fundamental observations when they feature land-use patterns that are determined by cost minimisation or maximisation of producer surpluses (Lotze-Campen *et al.*, 2008; Havlík *et al.*, 2011).

Models in general – and land-use models in particular – are simplified representations of reality that are constructed to fulfil different tasks. They are useful in two ways: (i) to test one's knowledge of key processes that determine land use and (ii) to project alternative future pathways (Veldkamp & Lambin, 2001). Veldkamp and Lambin (2001) have also noted that 'models of land-use change can address two separate questions: where are land-use changes likely to take place (location of change) and at what rates are changes likely to progress (quantity of change)'.

Models of future land use usually do not answer questions such as 'How will the world look like?' but rather address 'How could land use look like if there were strong political efforts for climate change mitigation and given that we have a high growth of population?' Thus, modelling strongly relies on the use of scenarios, which are representations of plausible assumptions on parameters such as population, crop yields or trade liberalisation. In this way, models 'help identify management options for maximizing sustainability goals' (Jones *et al.*, 2016).

5.1. Different approaches for modelling land use and agriculture

Land-use modelling is performed on different temporal and spatial scales, and the number of modelling approaches is probably as large as the range of research questions that they address. In view of this, the objective of this section is not to provide a comprehensive review of all land-use modelling methods but rather to provide insight into the variety of approaches.

Starting in the 1960s, first models of plant photosynthesis and soil water balance were developed (Jones *et al.*, 2016). Around the same period, urban planning employed computer-based land-use models (Harris, 1965; Wegener, 1995). Nowadays, models of land use and agriculture are utilised on almost every geographical level and for a multiplicity of purposes. They are also used on watershed (e.g. Verburg *et al.*, 2002; Kim *et al.*, 2011), national (e.g. Lapola *et al.*, 2010; Moss *et al.*, 2011), supranational (van Delden *et al.*, 2010; Britz & Witzke, 2014) and global levels (Ronneberger *et al.*, 2009; Havlík *et al.*, 2011; Calzadilla *et al.*, 2014; Meiyappan *et al.*, 2014; Hasegawa *et al.*, 2016). As with the spatial level, modelling approaches differ according to their purpose.

Cellular automata are grid-based models in which the state of neighbouring cells in the previous time step determines the state of each cell (Irwin & Geoghegan, 2001). This class of models is applicable for simulating the spread of forest fires, for example (Hernández Encinas *et al.*, 2007), including those intentionally ignited during slash-and-burn agriculture (Fujioka & Abe, 2016).

Agent-based land-use models (ABMs) incorporate human decision making by explicitly modelling 'agents' – mostly individuals or households – interacting with each other and their environment (Matthews *et al.*, 2007; Groeneveld *et al.*, 2017). Magliocca *et al.* (2013), for instance, have used agent-based modelling to study the relationship between population density and agricultural intensification. Arneth *et al.* (2014) have suggested the inclusion of representative ‘agent functional types’ in global land-use models; these are similar to the ‘plant functional types’ in ecological modelling and thus represent processes such as societal learning.

Suitability ranking is a rule-based approach that is common in land-use planning but is also utilised to forecast the spatial development of land use. Its basic rationale is that additional production is allocated to the most suitable location or grid cells in the model in response to a growing need for one commodity. A study by van der Hilst *et al.* (2012) has employed this procedure to examine land availability for bioenergy crop production in Mozambique and determined land suitability according to proximity to the same land-use class, distance to roads and population density. The LandSHIFT model (Schaldach *et al.*, 2011) and the IMAGE model (Stehfest *et al.*, 2014) also use this procedure in allocating land use.

Economic optimisation modelling that also relies on geographical data – which is the approach that this thesis adopts – is somewhat similar to suitability ranking, but it offers several advantages. It implicitly assumes a social planner that maximises welfare or minimises costs of production. Technically, this type of model is solved through mathematical optimisation, whereby a commercial solver such as CLPEX (IBM, 2012) or CONOPT (ARKI, 2012) determines local or global maxima or minima for the given objective function. Via the objective function, these models also imply suitability ranking. In response to an increase in food demand, agricultural land use will expand to the most suitable cells on the basis of the objective function – so, for instance, to the cells where production is possible at the lowest costs. In contrast to a simple suitability model, however, an increase in demand could also initiate higher investments into yield-increasing R&D, as in the case of MAgPIE (Dietrich *et al.*, 2014). Furthermore, the economic objective function also allows for assessment of the price effects of policies or the implications of a CO_2 price policy on land use.

The integration of both an economic core and biophysical constraints is a relatively new development (Michetti & Zampieri, 2014). Until roughly the turn of the millennium, there had been a strong distinction between economic models ‘focus[ing] on drivers of land-use change on the demand side’ and ‘geographic models focus on the development of spatial patterns of land-use types by analyzing land suitability and spatial interaction’ (Heistermann *et al.*, 2006). Today, a number of such integrated models are available which share common features. Due to their nature as global scale models, their analyses usually focus on processes that are important on the global scale, such as the large-scale deployment of bioenergy, or on general trends and drivers on the global or world-region scale, such as population growth. This type of model has been used to assess climate change impacts on agriculture, for instance (Nelson *et al.*, 2014), as well as the effects of increased bioenergy demand on food prices (Lotze-Campen *et al.*, 2014). They are partial equilibrium (PE) models of the agricultural sector, and partly the forestry sector, or are computable general equilibrium (CGE) models (van der Werf & Peterson, 2009). In an Agricultural Model Intercomparison and Improvement Project (AgMIP) exercise, six CGEs and five PEs participated (Delincé *et al.*, 2015). While CGE models are capable of considering effects in different economic sectors, they usually represent the land-use sector in less detail. Global models usually have in common that they aggregate countries to a limited number of regions, but the number differs drastically between the models (5 to 101 in Delincé *et al.*, 2015). Models with a more detailed, spatial representation of land use commonly rely on yield information from biophysical models such as EPIC (Izaurralde *et al.*, 2006), used by GLOBIOM (Havlík *et al.*, 2011), or LPJmL (Bondeau *et al.*, 2007) in the case of MAgPIE (Lotze-Campen *et al.*, 2008).

5.2. MAgPIE

This thesis largely relies on the Model of Agricultural Production and its Impact on the Environment (MAgPIE) (Lotze-Campen *et al.*, 2008). MAgPIE is a partial equilibrium model of land use which derives patterns of global agricultural production through an optimisation approach. The objective function of the model is to reduce global costs of agricultural production. In this way, it iteratively calculates cost-optimal patterns of land use in 5- or 10-year time steps.

The production side of the model is based on yields for crops, pasture and bioenergy plants, and carbon densities from the hydrology and vegetation model LPJmL (Bondeau *et al.*, 2007; Waha *et al.*, 2012; Müller & Robertson, 2014). These yields are calibrated to country-level yields from FAOSTAT (2013) and to match total regional cropland and pasture area in the starting year of the model. Land-use patterns in 1995, the year of model initialisation, are based on Erb *et al.* (2007). This geographical information is fed into the model on a spatial resolution of 0.5°. The demand side of the model is represented by ten world regions. Per-capita caloric demand and livestock shares in consumption are estimated on the basis of a regression on historical gross domestic product (GDP) and consumption and are projected for the future (Bodirsky *et al.*, 2015).

All activities in the model relate to costs that the model seeks to minimise. Factor costs for production are based on values from the Global Trade Analysis Project (GTAP) database (Narayanan & Walmsley, 2008). Yield increases in the model are achievable through a reallocation of production to locations with higher productivity. The MAgPIE model also features an endogenous representation of R&D investments that is founded on the relationship between historical R&D investments and following rates of yield increase (Dietrich *et al.*, 2014). Where water availability limits production, a switch to irrigated production can improve crop yields but requires initial costs for irrigation infrastructure expansion and annual irrigation costs (Bonsch, 2015). Investment decisions are based on a 30-year investment horizon and a 7% discount rate.

This thesis particularly considers the future extent of forests and the analysis of CO_2 emissions that accompany land-use change. Carbon emissions from deforestation are calculated from stock changes of the different land-use types based on carbon densities from LPJmL. Vegetation and soil carbon are assumed to change to the new land-use type over 20 years. Carbon accumulation as a result of afforestation occurs according to logistic forest growth curves, which depend on the climatic region (Humpenöder *et al.*, 2014).

The amount and location of land-use change depends on the demand, costs for production, intra-regional transport and land conversion, and other alternatives besides expansion to increase agricultural production. Intra-regional transport costs are calculated from total regional agricultural transport cost from GTAP (Narayanan & Walmsley, 2008), and travel times to major cities (Nelson, 2008). Since the location of land-use change heavily influences the effects on biodiversity and greenhouse gas emissions, land-conversion costs and protected areas were implemented spatially explicit for the study described in Chapter III. Costs for land conversion were divided into two components: land clearing and land establishment. Land-clearing costs are a function of the vegetation density of the initial land-use class, while costs for establishment depend on the succeeding land-use class. They are higher for croplands (1,000 to 10,000 USD) than they are for pastures (500 to 5,000 USD). Both costs are scaled with the per-capita GDP of the region. Protected areas were implemented based on the World Database on Protected Areas (UNEP-WCMC, 2016).

Global costs of agricultural production are minimised under the fulfilment of regional demand and self-sufficiency constraints (Schmitz *et al.*, 2012). To this end, cells with similar yield levels, carbon density and transport distances are aggregated to, for instance, 1,000 clusters (Dietrich *et al.*, 2013). After the optimisation, the model results regarding cluster level are disaggregated to the full spatial resolution of 0.5°. The model is programmed in GAMS (2012) and solved with the solvers CPLEX (IBM, 2012) and CONOPT (ARKI, 2012). Model inputs and outputs are mainly processed with R (R Core Team, 2016).

MAgPIE has been applied to address a broad range of research questions, for example, it has been used to study future greenhouse gas emissions (Schmitz *et al.*, 2012), both CO_2 emissions from land-use change (Popp *et al.*, 2014a) and agricultural non-CO_2 greenhouse gas emissions (Popp *et al.*, 2010; Weindl *et al.*, 2010). It has also been employed to forecast technological progress in agriculture (Dietrich *et al.*, 2014), nitrogen fertiliser requirements (Bodirsky *et al.*, 2014) and agricultural water usage (Bonsch *et al.*, 2015), and has been utilised to estimate the influence of governance level on agricultural development (Wang *et al.*, 2016), climate change impacts on agriculture and society (Stevanovic *et al.*, 2016), and terrestrial mitigation potentials (Humpenöder *et al.*, 2015). Moreover, the model has been extensively used to estimate potentials and consequences of bioenergy production in a stand-alone version (Popp *et al.*, 2012; Humpenöder *et al.*, 2014; Lotze-Campen *et al.*, 2014; Bonsch *et al.*, 2016) or within the MAgPIE-REMIND integrated assessment model (Popp *et al.*, 2011, 2014b; Klein *et al.*, 2014; Kriegler *et al.*, 2017).

6. Research questions

The previous sections have highlighted several challenges that are associated with current developments in land use, and they have noted certain anticipated challenges. This thesis poses several questions concerning the future of land use and its environmental side effects. Forests, their future extent, and how this influences climate and biodiversity are particularly central to this work, which assesses how forests could mitigate climate change and considers the socio-economic consequences of potential forest expansion. Some of the leading questions of this thesis are as follows: How is it possible to slow down and eventually stop and revert the loss of tropical forests? What are suitable policies to achieve this aim? Which positive and negative environmental and societal side effects would accompany this?

Since the avoidance of further deforestation is a key priority for climate change mitigation, Chapter II focuses on policies that may reduce tropical forest losses and thus decrease greenhouse gas emissions. As further agricultural expansion and loss of forests and other natural vegetation not only induce climate change but also affect biodiversity, Chapter III explores the loss of areas that are important for the conservation of biodiversity. It specifically investigates the effects of pasture intensification and an expansion of protected areas on reducing land-use change in Latin America and Africa. Moreover, it examines positive side effects for climate change mitigation and the influence of agricultural trade liberalisation. Lastly, large-scale reforestation and afforestation have been discussed as a means to remove carbon dioxide from the atmosphere. Chapter IV then takes into account CDR potentials of afforestation and assesses how the increased competition for land may affect food prices.

This thesis is structured in several chapters that have been published as individual papers in scientific journals. The next section details the research questions that the individual papers address.

6.1. Chapter II: How strongly are tropical forests threatened by further agricultural expansion, and what are appropriate policies to limit deforestation?

The first study of this thesis focuses on the future of tropical forests. It considers how much forest area Africa, Latin America and Pacific Asia may lose until the middle of the century. Previous studies have indicated that continuous agricultural trade liberalisation could especially threaten tropical forests. This study therefore concentrates on the ability of various policies to reduce deforestation under opening agricultural markets. It assesses whether a price on CO_2 emissions from land-use change could slow down deforestation and which price level would be necessary. It compares this policy to a gradual increase of the protected areas network. Lastly, it investigates additional investments into yield-increasing technology that could help to reduce deforestation.

6.2. Chapter III: How do protected areas and pasture intensification influence the loss of forests and natural vegetation, especially within conservation priority areas?

The study of Chapter III assesses losses of natural vegetation with a special focus on Latin America and Africa, which are two regions with large areas of relatively undisturbed forests and which contain several important biodiversity hotspots. The study estimates the strength of the impact of future agricultural expansion on these areas and analyses how an expansion of protected areas could reduce natural vegetation loss. A key consideration for this question is whether a protection in one place would lead to a reallocation of agricultural expansion to nearby locations without reducing the overall amount of land-use change. This assessment is combined with another aspect that is essential but has nevertheless received insufficient attention thus far: the influence of pasture intensification. Even though pastures represent a larger share of agricultural area than croplands, existing research has mainly explored effects of cropland intensification. This study, however, assesses the influence of different rates of pasture intensification on land-use change in Africa and Latin America.

6.3. Chapter IV: How much could afforestation contribute to climate change mitigation, and how would it affect food prices?

While deforestation is responsible for significant carbon emissions, reforestation could significantly contribute to climate change mitigation. However, afforestation on croplands and pastures would also interfere with food production. This study is interested in the potential of afforestation for carbon dioxide removal, as well as its consequences for food prices. It is structured by scenarios in which afforestation is limited to specific latitudinal zones; this is motivated by the knowledge that afforestation in higher latitudes reveals only a limited cooling effect. The albedo decrease that accompanies the conversion from agricultural land use to forests reduces the amount of short-wave radiation that is reflected back to space, and it therefore counteracts the carbon cooling effect. This study therefore assesses how a limitation of afforestation to tropical regions could alleviate its possible effect on food prices. Lastly, it evaluates how increased trade could further hamper price hikes.

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Agricultural trade and tropical deforestation: interactions and related policy options

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Abstract

The extensive clearing of tropical forests throughout past decades has been partly assigned to increased trade in agricultural goods. Since further trade liberalisation can be expected, remaining rainforests are likely to face additional threats with negative implications for climate mitigation and the local environment. We apply a spatially explicit economic land-use model coupled to a biophysical vegetation model to examine linkages and associated policies between trade and tropical deforestation in the future. Results indicate that further trade liberalisation leads to an expansion of deforestation in Amazonia due to comparative advantages of agriculture in South America. Globally, between 30 and 60 million ha (5–10 %) of tropical rainforests would be cleared additionally, leading to 20–40 Gt additional CO_2 emissions by 2050. By applying different forest protection policies, those values could be reduced substantially. Most effective would be the inclusion of avoided deforestation into a global emissions trading scheme. Carbon prices corresponding to the concentration target of 550 ppm would prevent deforestation after 2020. Investing in agricultural productivity reduces pressure on tropical forests without the necessity of direct protection. In general, additional trade-induced demand from developed and emerging countries should be compensated by international efforts to protect natural resources in tropical regions.

Keywords

Land-use change; Trade liberalisation; Tropical deforestation; Forest protection; Agricultural productivity growth

Introduction

Throughout the past three decades, tropical deforestation has contributed between 12 and 25 % to worldwide greenhouse gas emissions (Houghton 2003; Fearnside and Laurance 2003; van der Werf *et al.* 2009). Total net release of carbon from forest change in the 1990s varied according to different methodology and data sources between 0.5 and 2.2 PgC per year, having increased considerably since the 1950s (Ramankutty *et al.* 2006). A more recent study estimates average net emissions from tropical land-use change at 1.5 PgC per year in the 1990s and 1.1 PgC per year between 2000 and 2007 (Pan *et al.* 2011). Besides generating carbon emissions, deforestation leads to socio-economic damages for the local population (Barraclough and Ghimire 2000), reduced water cycling (Fearnside 2005), increased flood risk (Bradshaw *et al.* 2007), disruptions to the local climate (Costa and Foley 2000) and severe loss of biodiversity (Gorenflo and Brandon 2005). From FAO country studies, it is assessed that since the 1980s on average around 13 million ha of forest area has been lost every year (Ramankutty *et al.* 2006; FAO 2010).

Cropland expansion is considered to be one of the key drivers behind tropical deforestation. Commercial and subsistence agriculture are related to about three-quarters of deforestation (Hosonuma *et al.* 2012). Another study about deforestation in Brazil based on satellite data indicated that up to 23 % is triggered by cropland expansion and 66 % by pasture expansion (Morton *et al.* 2006). By using the Landsat database from FAO, Gibbs *et al.* (2010) revealed that between 1980 and 2000, about 55 % of new agricultural land in the Pan-Tropics came from intact forests and about 30 % from disturbed forests. Particularly, in South America, large-scale and enterprise-driven agriculture fuelled by rising consumer demand is a major cause (Parker *et al.* 2009). In contrast, in Central Africa, extraction of natural resources (e.g. timber) and in Pacific Asia pressure from commercial agricultural plantations are seen as the main driving forces behind the forest loss (Lambin *et al.* 2010). Although some recent sources have referred to a decreasing deforestation rate (Kauppi *et al.* 2006; FAO 2010), the remaining rainforest worldwide is in severe danger due to increasing demand for food and other agricultural products (Gibbs *et al.* 2010).

Besides the general rise in agricultural demand, several studies point out that further trade liberalisation is and will be an important factor for deforestation activities. Barbier (2000) demonstrated this relationship with case studies from Ghana and Mexico. In Brazil, improved access to international markets has pushed soy and beef production causing a surge in deforestation (Fearnside 2005; Nepstad *et al.* 2006). Based on satellite data, DeFries *et al.* (2010) concluded that forest loss is largely driven by urban population growth and international exports of agricultural products. Other studies have used a global modelling approach to analyse future effects of trade liberalisation. Verburg *et al.* (2009) and Schmitz *et al.* (2012) have shown that the rates of tropical deforestation and global greenhouse gas emissions are likely to rise with increased trade liberalisation in the future. Similar studies have emphasised that liberalising trade leads only to small land-use shifts in Europe but dramatic shifts in developing regions with negative implications for the environment (van Meijl *et al.* 2006; Eickhout *et al.* 2010).

To induce climate change mitigation and reduce further deforestation, different policies are available (Forner *et al.* 2005; Kolstad *et al.* 2014). These include direct regulatory approaches, economic incentives, or government provision of technology to tackle the problem. Direct regulation is mainly applied to protected areas (PAs); it has been shown to be effective (Nelson and Chomitz 2011; Beresford *et al.* 2013) and is linked to the recent slowdown of deforestation in the Amazonian rainforest (Soares-Filho *et al.* 2010). Economic incentives include classic measures like taxes or subsidies but also tradable emissions allowances. Pricing greenhouse gas emissions from the land-use sector has been proposed as one promising approach and has been analysed extensively through the application of large-scale integrated land-use models (Kindermann *et al.* 2008; Wise *et al.* 2009; Thomson *et al.* 2010). The sequestration and storage of carbon in vegetation can also be rewarded by Payments for Ecosystem Services (PES). Rewarding measurable, below-baseline emissions is also the idea behind the REDD scheme (Reducing Emissions from Deforestation and Degradation) currently discussed under the United Nations Framework Convention on Climate Change (UNFCCC). Governments could also invest in yield increasing Research & Development that lowers the pressure on expansion into forests. Whether this is a promising strategy has been under discussion for several years.

The Borlaug hypothesis, named after the father of the Green Revolution Norman Borlaug, suggests that yield increases lead to a lower spatial need for production and thus have and will save natural ecosystems such as forests. In contrast, according to Jevon's paradox, at the local forest frontiers, new technologies can be labour saving, thus freeing workforce for expanding agriculture (Angelsen and Kaimowitz 2001; Hertel 2012). Byerlee *et al.* (2014), however, conclude in their recent literature review that at a global level, investment in R&D to improve productivity remains one of the best ways to reduce pressure on increasingly scarce land resources and conserve natural ecosystems.

Previous studies have either focused on trade liberalisation or on forest protection measures, but none have looked at the important interplay between these. We here integrate both effects and consider explicitly the interaction between trade liberalisation and deforestation. We apply the economic land-use model MAgPIE ("Model of Agricultural Production and its Impact on the Environment"), which takes global and regional interactions into account and simulates spatially explicit land-use patterns. MAgPIE uses endogenously derived technological change and land expansion rates, which make it unique in the field of land-use modelling. Biophysical processes and inputs are considered through the link with the global vegetation-hydrology model LPJmL. The main goal of our study is to investigate consequences of different trade volume scenarios and forest protection policies on land-use change, carbon emissions, net exports, and technological change rates over the coming five decades. As forest protection scenarios, we assume an expansion of protected areas, different carbon price scenarios and one case in which agricultural productivity in forest regions is increased through higher investments in Research & Development and infrastructure. The latter is used to highlight the important interplay between land expansion and technological change (Lotze-Campen *et al.* 2010; Dietrich *et al.* 2013; Popp *et al.* 2012). We start by explaining the model framework with the implementation of trade and forest and by describing the applied scenarios. Following this, we present results of the analysis which are, finally, compared and discussed.

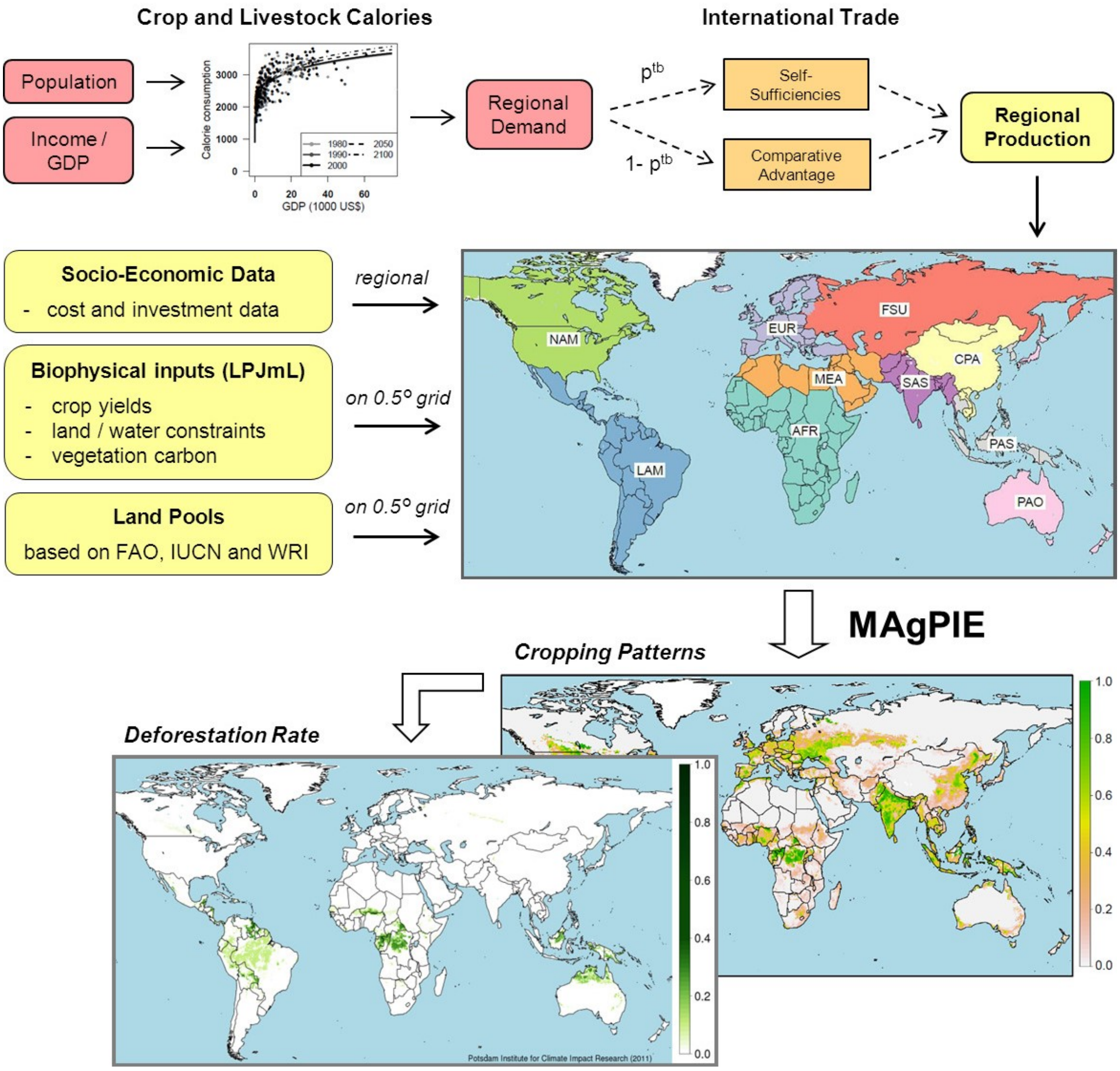

Fig. 1 Simplified MAgPIE flow chart of key processes highlighted in this study (demand and trade implementation, land pools and spatially explicit land-use change). With exogenous data about population and GDP development, we calculate regional demand and the livestock share. The former is then translated to regional production depending on the international trade scenario. Further inputs for MAgPIE are socio-economic data like production costs, biophysical inputs from LPJmL and land-type data based on various sources (FAO, IUCN and hydrology WRI). After optimisation of MAgPIE, possible outputs are cropping patterns of different crops or maps with deforestation rates. MAgPIE divides the world into ten regions: AFR = Sub-Sahara Africa, CPA = Centrally Planned Asia (incl. China), EUR = Europe (incl. Turkey), FSU = Former Soviet Union, LAM = Latin America, MEA = Middle East and North Africa, NAM = North America, PAO = Pacific OECD (Australia, Japan and New Zealand), PAS = Pacific Asia, SAS = South Asia (incl. India)

Methods

General model description

For the analysis, we use the recursive dynamic optimisation model MAgPIE ("Model of Agricultural Production and its Impact on the Environment"). In the following, we briefly present the main model features for this study. For further details, we refer to the extensive model documentation (Lotze-Campen *et al.* 2008, 2010; Popp *et al.* 2010, 2011;

Schmitz *et al.* 2012) and the mathematical description, which is attached as supplementary material.

Figure 1 presents a simplified flow chart of the inputs for MAgPIE. The model reflects three layers: global, regional (reflected by ten world regions, see Fig. 1) and cellular layers (based on 0.5 degree resolution). MAgPIE simulates time steps of 10 years (starting in 1995) and uses in each period the optimal land-use pattern from the previous period as a starting point. Required calories in the demand categories are derived through a cross-country regression based on a medium population scenario (UN 2011) and a medium income-growth scenario (projections based on Heston *et al.* (2011)). With the implementation of international trade, it is determined how many calories are produced domestically and how many are imported. In MAgPIE, trade can be either fixed, if it is allocated according to historic self-sufficiency rates (1995 values from FAO (2011a)), or liberalised, which means that regions with comparative advantages produce more at the expense of less competitive regions. The share of the two options is determined by the trade balance reduction factor p_tb (see Fig. 1). More details on the trade implementation are described in Schmitz *et al.* (2012). The resulting calories are produced by 16 crop groups (temperate cereals, maize, tropical cereals, rice, soybean, rapeseed, groundnut, sunflower, oil palm, pulses, potato, cassava, sugar beet, sugar cane, cotton, others) and 5 livestock types (ruminant meat, pig meat, poultry meat, egg, milk) in the particular regions.

Further inputs to MAgPIE are socio-economic data, mainly costs, which define the cost minimisation objective function. In the baseline version of the model, four categories of costs arise: (1) Production costs are taken from GTAP (Narayanan and Walmsley 2008) and contain factor costs for labour, capital, and intermediate inputs. (2) Technological change is endogenously implemented in MAgPIE. That means the model decides (based on an investment regression) how much additional technological change is required and cost-effective. Costs are based on investments in agricultural Research & Development as well as infrastructure investments (Dietrich *et al.* 2013). They rise exponentially with the state of agricultural development of a region (Dietrich *et al.* 2012). The endogenous implementation allows MAgPIE to project future yield increases and the costs involved. (3) Land expansion involves costs for preparation of new land and basic infrastructure investments (Krause *et al.* 2012). Land conversion costs are based on country-level marginal access costs generated by the Global Timber Model (GTM) (Sohngen *et al.* 2009). Regarding the conversion of intact and frontier forests (IFF), we base our cost parameterisation on reference values from case studies. Merry *et al.* (2002) analysed forest transition in Latin America with a case study of Bolivia and calculated conversion costs of 600 to 700 US$/ha. Similar costs accrue in Indonesia where the value for converting rainforest to cropland is 550 US$/ ha (Simorangkir 2007). Another case study from Latvia, however, reveals considerably higher costs of 1,500 US$/ ha (Lazdins *et al.* 2008). In developed countries, this value (based on marginal access costs) increases even further up to 7,500 US$/ha (Sohngen *et al.* 2009). The large variation in costs is due to topography, forest type, soil conditions, applied technology and the governmental system. As a base value we assume 1,000 US$/ha for tropical land conversion. We applied a sensitivity analysis of this parameter by varying it in 200 US$ steps from 200 US$ to 1,800 US$ (see

Fig. 9). (4) Intraregional transport costs for every commodity unit reflect the distance to intraregional markets and the quality of the infrastructure. Data for transport costs are derived from GTAP (Narayanan and Walmsley 2008) and travel time to the nearest city is reflected by a 30 arc-second resolution data set (Nelson 2008). For long-term investments, like land conversion or R&D, we assume an annual discount rate of 7 %, which reflects the opportunity costs of capital at the global level (IPCC (2007), chapter 2.4.2.1).

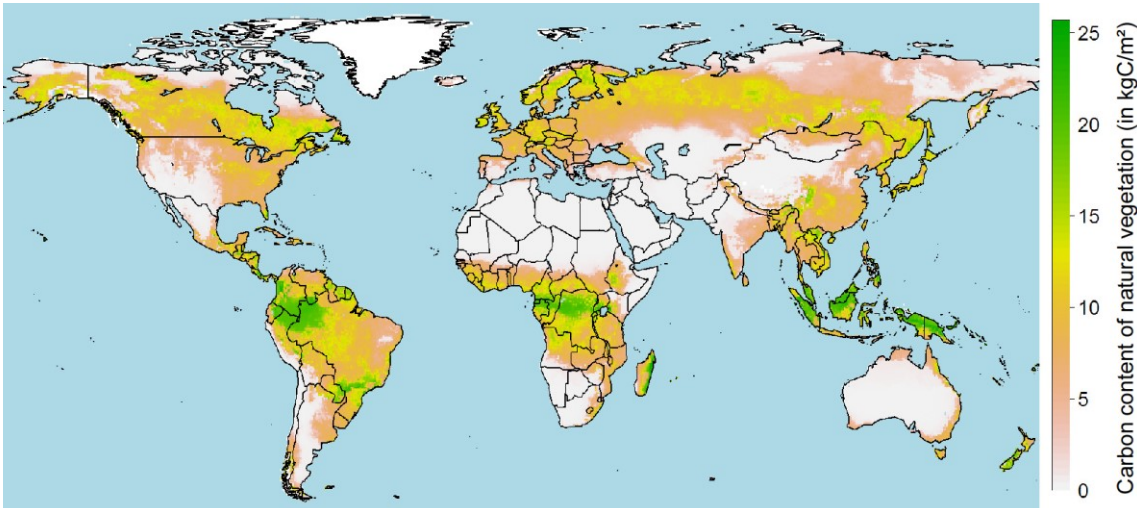

Fig. 2 Grid-cell-specific carbon content (0.5 degree) of natural vegetation (in kgC/m²) from LPJmL (average from 1990 to 1999) used in MAgPIE

For the representation of biophysical processes, MAgPIE is linked to the global biophysical vegetation-hydrology model LPJmL (Bondeau *et al.* 2007). LPJmL endogenously models the dynamic processes linking climate and soil conditions, water availability and plant growth, and takes the impacts of CO_2, temperature and radiation on yield directly into account. The link to MAgPIE is generated via rainfed and irrigated yields for different crops, rainfed and irrigated land-use fractions (Fader *et al.* 2010), water inputs, like irrigation requirements and water availability (Rost *et al.* 2008), and the carbon content of the various vegetation types. These outputs from LPJmL are used in a 0.5 degree resolution in MAgPIE. The same resolution is used for the determination of land types per grid cell. The different land pools are taken from a consistent land-use database developed by Krause *et al.* (2009) which is based on Erb *et al.* (2007) and integrates crop suitability indicators (van Velthuizen *et al.* 2007), intact and frontier forest types (Bryant *et al.* 1997; Potapov *et al.* 2008) and protected areas (UNEP-WCMC 2006). Intact and frontier forests can also be denoted as undisturbed natural forests. Together with other natural vegetation not defined as grazing land or forest (around 122 million ha), it constitutes the land pool that is made available for cropland expansion (around 734 million ha). The remaining land pools, like pasture and managed forests, are not regarded for cropland expansion. When land-use change occurs and land is converted to a different type (e.g. forest to cropland), MAgPIE accounts for carbon emissions by taking the differences in LPJmL-derived carbon stocks between the two land pools. The used LPJmL model version is able to capture changes in above- and belowground vegetation carbon (see Fig. 2) but not in soil carbon. Related carbon emissions are reported as CO_2-equivalent emissions after each time step.

Scenario design

The aim of this study is to investigate interactions between international trade policy and forest protection measures (Table 1) and their consequences on tropical deforestation patterns.

Table 1 Scenario definition

Policies	**Trade policy**	**Forest policy**		
Scale	*global*	**AFR**	**LAM**	**PAS**
Reference scenarios [*reference*]	Constant	Basic forest protection		
Trade scenarios				
(a) No forest policy [*nopol*]	Liberalisation	Basic forest protection		
(b) Increasing forest protection over time [*time*]	Liberalisation	Unitil 2040	Until 2030	Until 2030
(c) Low CO_2 price [*lowprice*]	Liberalisation	Low CO_2-price		
(d) CO_2-price to achieve 550 ppm [*550 ppm*]	Liberalisation	High CO_2price		
(e) Additional investment in TC [*TC*]	Liberalisation	1 % TC p.a.		

Concerning trade policy, our analysis largely follows the policy scenario of the predecessor study (Schmitz *et al.* 2012), except that trade liberalisation starts in 2015 (instead of 2005). Hence, our reference case keeps the trade patterns fixed over time, whereas the trade scenarios assume further progress in the Doha Development Round,[1] leading to liberalisation efforts comparable to situations in the 1980s and 1990s, when large global liberalisation efforts were undertaken. Based on Dollar and Kraay (2004) and Conforti and Salvatici (2004), we assume that trade barriers are continuously reduced by 10 % each decade. The trade policy is the same in all five trade scenarios, but the scenarios differ according to their forest policy (Table 1). Whereas the scenario *nopol* assumes no forest protection measures in order to highlight the differences of the trade effect compared to the reference case, the other four scenarios assume different global and regional policy measures to reduce deforestation.

As a first scenario, we introduce policies to restrict deforestation and to implement protected areas (PAs). Based on Soares-Filho *et al.* (2006), we consider a defined share of intact and frontier forest as protected and increase this share over time (*time* scenario). For the three main tropical IFF regions, we assume a different time span (2040 in AFR and 2030 in LAM and PAS) until full forest protection is achieved depending on awareness level and governmental structures (Table 2). For comprehensibility reasons and to depict the whole range of possible outcomes, we allow for no deforestation in these protected areas. As a further scenario set-up, we introduce a CO_2 price as climate mitigation policy, which has to be paid in cases of deforestation and increases the costs of land conversion. In contrast to other approaches, which use constant carbon prices over time (e.g. Kindermann

[1] The Doha Development Round is the latest round of trade negotiations of the World Trade Organisation (WTO). It was launched in 2011 with the aim of improving the access to global markets. For more information on the stage and agenda of the Doha Round, see Martin and Mattoo (2011)

et al. 2008), our price assumption rises over time. We differentiate two cases. First, we reflect a low-price scenario (*lowprice*), in which the price per tonne of CO_2 starts at 5 US$ and rises continuously to 12.5 US$ (Fig. 3). In 2013, an average of 4.9 US$ was paid per tonne of CO_2 on the voluntary offset market (Peters-Stanley and Gonzalez 2014). In a second CO_2 price scenario, called *550ppm*, we consider the other case, in which CO_2 emissions from deforestation are included in a potential global carbon market. The CO_2 price is in this case based on modelling results from the ReMIND model for the Energy Modeling Forum (EMF-24) (Luderer *et al.* 2012), which assumes a maximum concentration of greenhouse gas emissions of 550 ppm (Fig. 3).

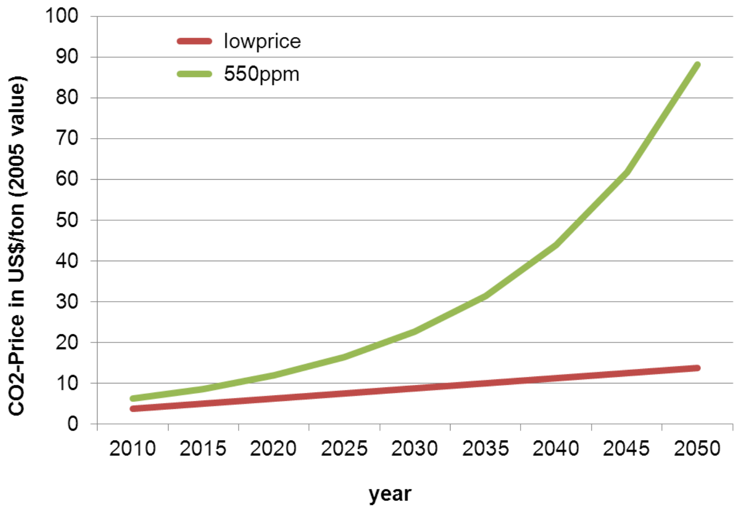

Fig. 3 Modelled CO2-Price (in US$/tonne) for the lowprice and 550ppm scenario until 2050

Finally, the last scenario assumes that the three forest regions, Latin America (LAM), Sub-Sahara Africa (AFR) and Pacific Asia (PAS), receive financial means to increase their yields by 1 % per year. This kind of exogenous technological change (TC) is a special case since no direct intervention of forest protection is assumed and only indirect effects on the forest area will be obtained. At the same time, the countries are allowed to invest in TC on top of that external investment. The hypothesis behind this scenario is that higher investments in TC can reduce the rate of forest destruction without any forest protection.

Table 2 Forest protection rate in the past (2000–2010) and assumed rates for the future (2010–2050) in the trade scenario time

Scenario	Region	2000-10 (%)	2010-20 (%)	2020-30 (%)	2030-40 (%)	2040-50 (%)
Basic protection	AFR	8	8	8	8	8
(observed)	LAM	25	25	25	25	25
	PAS	12	12	12	12	12
Protection over time	AFR	8	31	54	77	100
(assumed)	LAM	25	50	75	100	100
	PAS	12	41	70	100	100

Results

Tropical deforestation and carbon emissions

Table 3 provides an overview showing the potential area of tropical intact and frontier forest (IFF) in the three forest regions in 2050 as well as the change between 2010 and 2050 under the different scenarios. The concentration of IFF in Latin America (~80%) is also reflected in the deforestation patterns, as the region sees the highest forest loss in all scenarios. Since a much smaller share of tropical IFF is located in Central Africa (~10%) and Southeast Asia (~9%), deforestation is quite small in absolute terms, but percentual changes in IFF are much higher than in LAM (in Central Africa up to 99 % depending on the scenario).

Table 3 Intact and Frontier Forest (IFF) in 2050, deforestation area (2010-2050), associated CO_2 emissions and the net average carbon emissions of deforested area in the different scenarios

Region	Result	Unit	Reference	Nopol	Time	Low price	550 ppm	TC
Latin America (LAM)	IFF in 2050	10^6 ha	339.5	299.7	388.5	411.1	549.6	343.3
	Deforestation (2010-50)	10^6 ha	140.5	180.3	91.5	68.9	20.4	136.7
	CO_2 emissions (2010-50)	Gt CO_2	60.0	84.5	42.9	27.3	5.5	58.3
	Average carbon emissions	kgC/m²	11.7	12.8	12.8	10.8	7.4	11.6
Sub-Saharan Africa (AFR)	IFF in 2050	10^6 ha	0.7	0.9	9.2	34.5	63.6	1.1
	Deforestation (2010-50)	10^6 ha	63.7	63.5	55.2	29.9	0.8	63.3
	CO_2 emissions (2010-50)	Gt CO_2	40.8	40.5	36.2	17.0	0.5	38.8
	Average carbon emissions	kgC/m²	17.5	17.4	17.9	15.5	15.4	16.7
Pacific Asia (PAS)	IFF in 2050	10^6 ha	31.2	35.0	50.3	47.1	45.2	49.4
	Deforestation (2010-50)	10^6 ha	24.3	20.5	5.2	8.4	10.3	6.1
	CO_2 emissions (2010-50)	Gt CO_2	10.9	9.9	2.6	0.9	2.1	2.8
	Average carbon emissions	kgC/m²	12.2	13.2	13.6	2.9	5.6	12.5

In Latin America, around 140 million ha of IFF is deforested between 2010 and 2050 in the reference case, leading to 60 Gt CO_2 emissions. With additional trade liberalisation, this value grows to 180 million ha and about 85 Gt CO_2 emissions. The forest protection scenario (*time*) and the two price scenarios (*lowprice* and *550ppm*) lead to lower deforestation rates than in the reference case and to almost no emissions after 2040 (Fig. 4). With exogenous TC, additional CO_2 emissions can be reduced to a similar level to that of the reference case (60 Gt CO_2). Most effective is the integration of deforestation in a potential carbon market (*550ppm* scenario), leading to a total IFF loss of only 20.4 million ha and corresponding emissions of 5.5 Gt CO_2. In the *lowprice* scenario, deforestation is reduced to 69 million ha and with full forest protection until 2030 around 92 million ha will still be cleared prior to 2030.

For the Central African rainforest, the picture looks different. Almost all IFF will be gone under the *reference*, the *nopol* and the *TC* scenarios (around 63 million ha). This leads to relatively more CO_2 emissions (40 Gt), since the average carbon content in AFR is higher than in the deforested area in LAM. Full forest protection until 2040 saves 9.2 million ha of IFF, the *lowprice* scenario saves around 35 million ha and the *550ppm* scenario saves almost the whole IFF (64 million ha). In Pacific Asia, deforested area decreases under trade liberalisation. Additionally, in contrast to the other regions, the *time* and *TC* scenarios are most effective by conserving around 50 million ha of the original 55.5 million. Additionally, the lower CO_2 price saves 2 million ha more than the higher price scenario (*550ppm*).

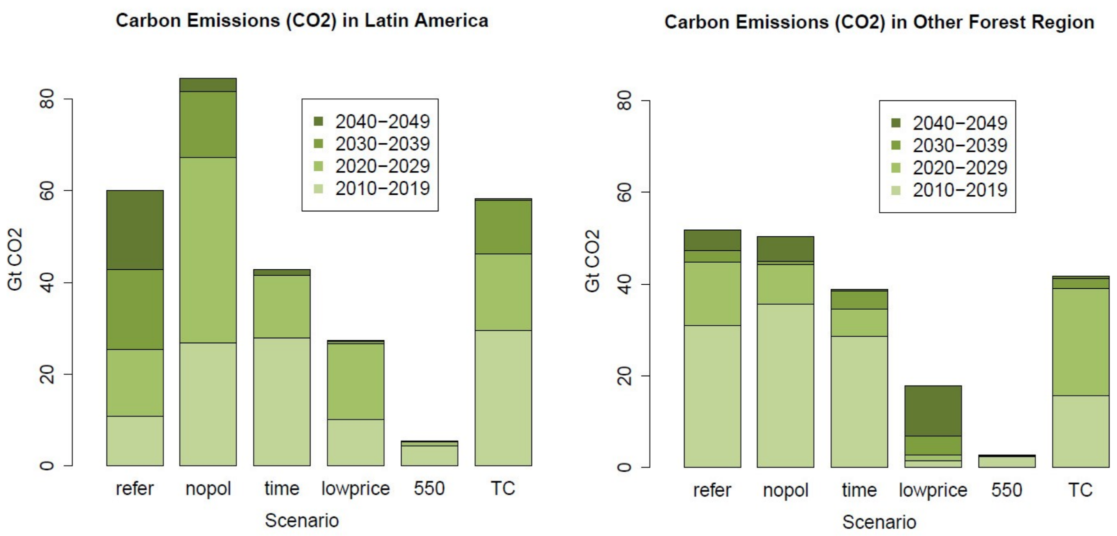

Fig. 4 CO_2 Emissions (in Gt) from tropical deforestation over time and for the two forest regions (LAM and OFR)

The net average carbon emissions per deforested hectare in all scenarios is highest in Central Africa (Table 3), where the northern part of the rainforest has the highest carbon densities (see Fig. 2). In South America, average carbon intensity is lower, since mostly cells at the border with a lower carbon content are affected by deforestation (see Fig. 5). As the model minimises costs, considering a CO_2 price for released carbon (as in the *lowprice* and *550ppm* scenarios) includes an additional decision criteria to the objective function. In these scenarios, we observe a substantial reduction in the average per hectare carbon emissions since the model has an explicit incentive to minimise carbon release by choosing low-carbon cells for land conversion.

For presentation purposes, we have aggregated the model results into four regions. Latin America is treated separately due to its importance for IFF and the agricultural sector. Sub-Sahara Africa and Pacific Asia are grouped in the category “Other-tropical-Forest Regions” (OFR). For net export and technological change rates, the remaining regions are grouped together as Non-tropical-Forest Developing Countries (NFDC) (mainly China, India, Russia, and the Middle East) and OECD countries. The pace of deforestation varies substantially between scenarios (Fig. 4). Forest clearance in LAM is much faster under the *nopol* scenario than under the reference scenario (drawing level with the 2030 baseline values in 2020 and exceeding the 2050 baseline in 2030). Including a low CO_2 price reduces emissions in LAM until 2050 to a level compared to the *nopol* scenario in 2020. In OFR, we obtain that in some scenarios (*reference*, *nopol*, *lowprice*) deforestation is higher in the last time step than in the penultimate time step. In the other scenarios, almost no deforestation takes place after 2040 due to full protection (*time*), high CO_2 prices (*550ppm*) or high agricultural productivity (*TC*).

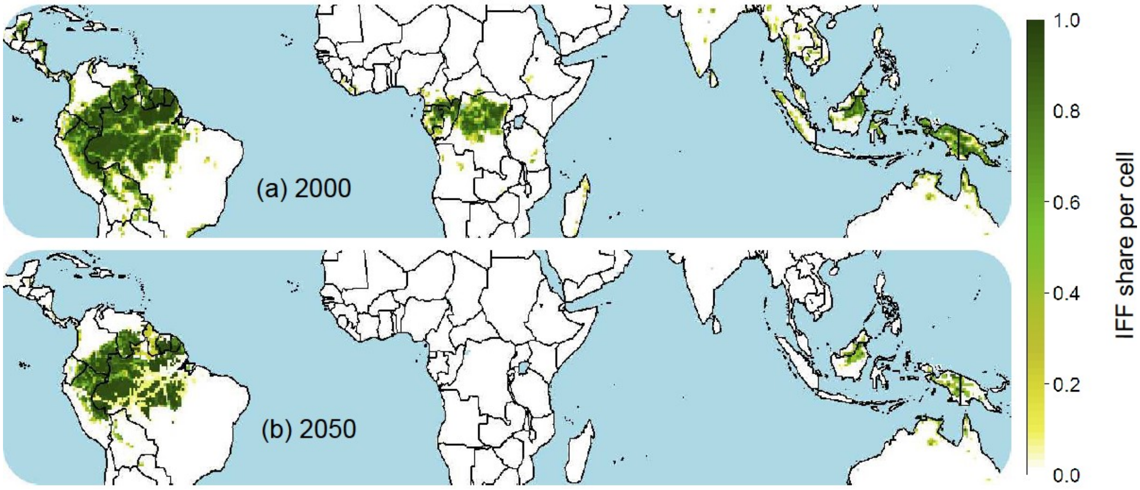

Fig. 5 Share of tropical intact and frontier forest per grid cell in the reference case in the years 2000 and 2050

In the following, we present grid-specific maps, which support the understanding of local dynamics. Figure 5a presents the tropical intact and frontier forest (IFF) in the year 2000. The tropical IFF forest is mainly located in Amazonia, Central Africa (mainly DR Congo, Cameroon, Gabon and Congo) and Southeast Asia (mainly Malaysia, Indonesia, the Philippines and Papua New Guinea). Compared to the state in 2000, Fig. 5b highlights the potential area of IFF in 2050 for the reference case. The Amazonia rainforest is considerably reduced especially at the borders in the south and west, but also within the forest, where infrastructure exists. The situation in Central Africa is even more intense, since in the reference case almost all IFF area would be cleared. In Pacific Asia, forest area is reduced significantly in some locations, up to a complete loss of IFF.

To analyse the importance of trade liberalisation and forest protection measures in a spatially explicit way, we investigate the scenarios' differences to the baseline setting in 2050 with difference maps (Fig. 6). Positive values indicate a higher share of IFF in the scenarios and a negative value indicates further deforestation. The effects of trade liberalisation on deforestation rates are shown in Fig. 6a (*reference* in 2050 minus *nopol* in 2050). In Latin America, the northern part of Amazonia and some border areas in the west are most negatively affected by trade liberalisation. Additionally, the interior close to existing infrastructure faces slight increases in deforestation. In Africa, nothing changes as the whole forest would be gone in both scenarios, whereas Pacific Asia has lower and North Australia higher deforestation rates. Analysing the effects of forest protection measures, we show that deforestation in LAM is very sensitive to forest protection. If parts of the rainforest are protected with an increasing rate (Fig. 6b), it mostly helps interior areas of the forest. Only some cells in the north of the forest are still deforested but to a lower extent than without protection policy. Both CO_2 price scenarios lead to much lower deforestation rates in the interior of the forest. In the *550ppm* scenario, this is most effective in the south (Fig. 6d). Finally, in the *TC* scenario, almost no differences can be detected compared to the reference case with respect to South America, except for some border cells in the north and west. In Africa, the CO_2 price scenarios have the biggest effect on deforestation, protecting the northern part and in the *550ppm* scenario also the southern and western part. The expansion of protected areas (*time* scenario) has only small effects on deforestation patterns, and investments in agricultural productivity (*TC* scenario) have no effects on deforestation as the whole forest is still cleared for agriculture. In Pacific Asia, all forest protection measures have positive effects with highest forest savings in Papua New Guinea.

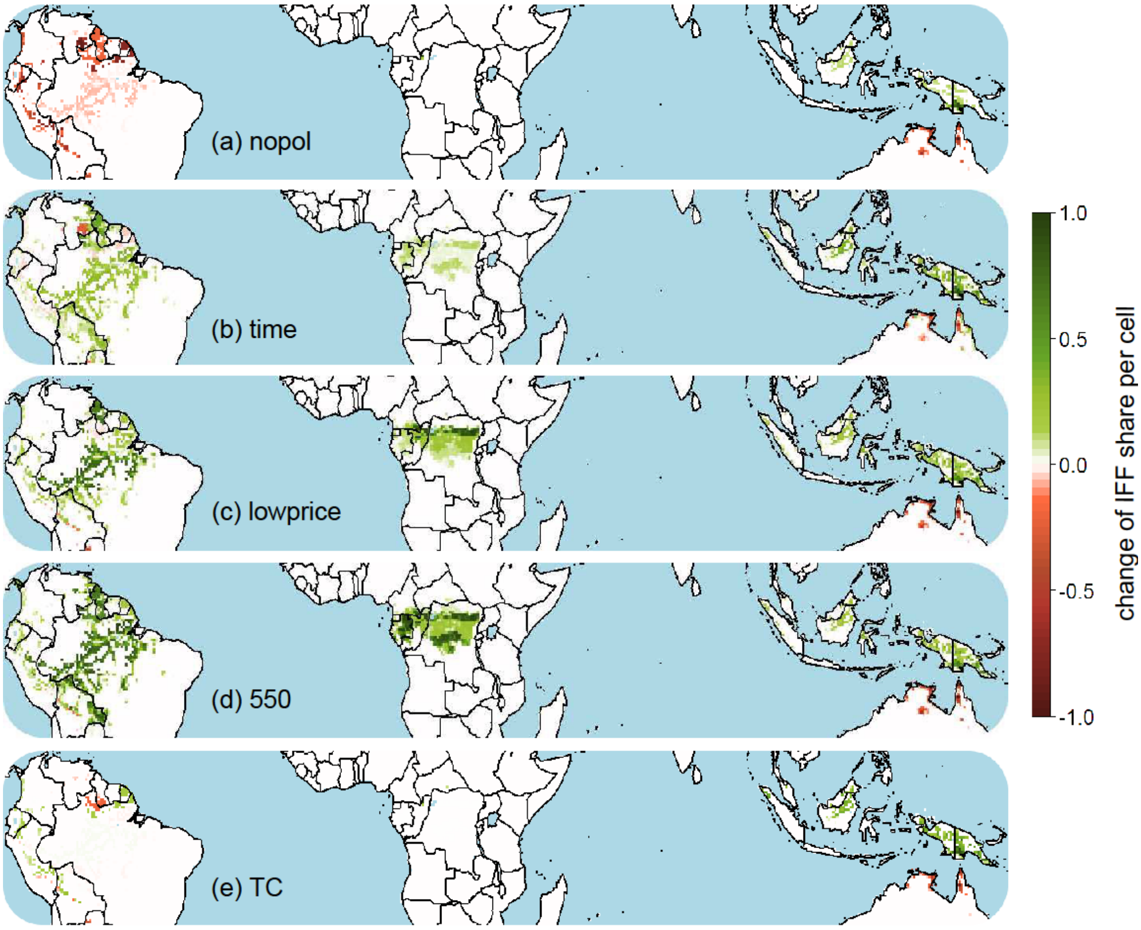

Fig. 6 Change of intact and frontier forest share per grid cell in the five trade scenarios compared to the reference case in 2050 (red cells display additional deforestation, green cells display less deforestation)

Net export and technological change rates

The analysis of net export rates indicates regions with comparative advantages in agricultural production. Figure 7 illustrates net export rates for cereals, oilcrops, sugar and meat in the reference case and the trade scenarios.

In general, under trade liberalisation, Latin America exports more of every commodity compared to the reference scenario. In case of cereals, LAM turns from a net importer to a net exporter. Under forest protection, LAM becomes a net importer again whereas the TC scenario generates the highest cereal net exports. Other commodities are less (oilcrops) or not at all (sugar, meat) affected by various forest protection policies and remain on a high export level. Trade liberalisation allows Non-tropical-Forest Developing Countries (NFDC) to reduce their imports in oilcrops at the expense of OECD countries, which face a drop in export levels. The rise in sugar exports in LAM leads to additional imports in NFDC and OECD countries. Concerning meat, the overall extent of trade is rather low in 2050. Regions with tropical IFF increase their exports in livestock, whereas NFDC increase imports and OECD countries turn from exporters to importers.

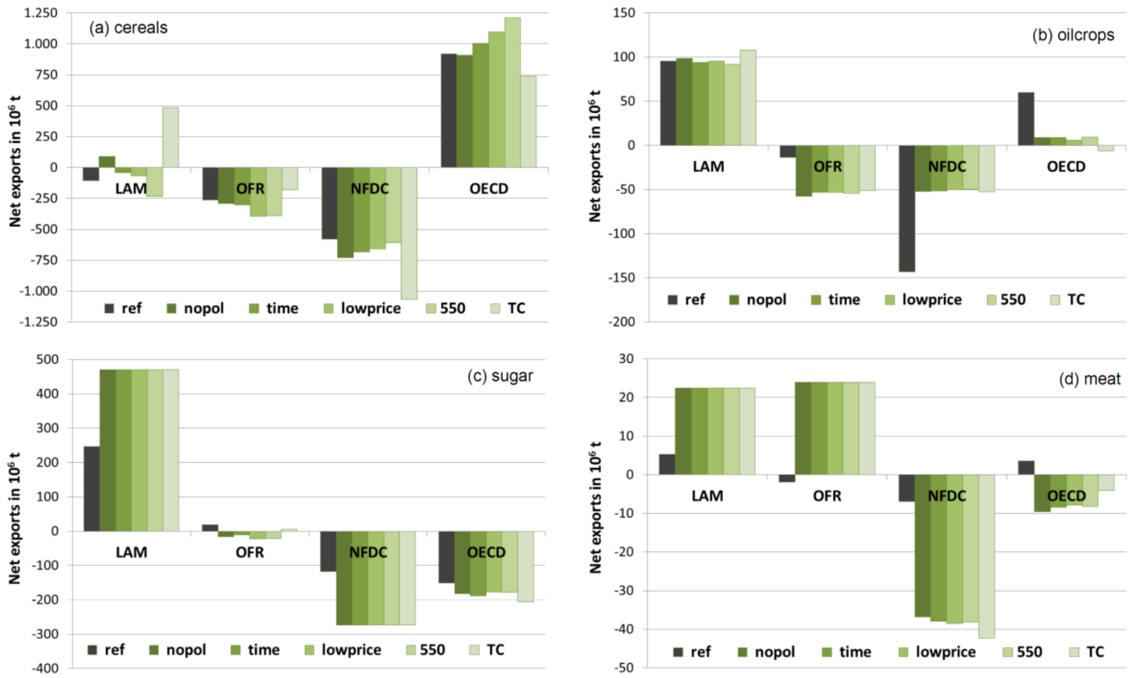

Fig. 7 Aggregated net exports (2010–2050) for the traded commodities (cereals, oilcrops, sugar and meat) in Latin America (LAM), Other Forest Regions (OFR), Non-tropical-Forest Developing Countries (NFDC) and OECD countries

Technological change (TC) rates are endogenously derived by MAgPIE (Dietrich *et al.* 2012, 2013), indicating the need for investments in technological development of the agricultural sector per region. In LAM, no investment into TC is observed in the reference case, production increase is mainly the result of an increase in cropland at the expense of tropical rainforests (Fig. 8). In turn, in all other regions, TC rates decrease with trade liberalisation compared to the reference case. Among the trade liberalisation scenarios, TC rates are lowest in the *nopol* scenario in LAM (0.1 %) and OFR (0.54 %) and highest where 1 % annual Technological Change is provided at no costs (*TC* scenario). These high TC rates, however, do not change cropland area in these two regions a lot, but slightly reduce the expansion of cropland in OECD countries. In general, forest protection increases the need for TC in regions rich in intact and frontier forests, if growing food demand is to be fulfilled.

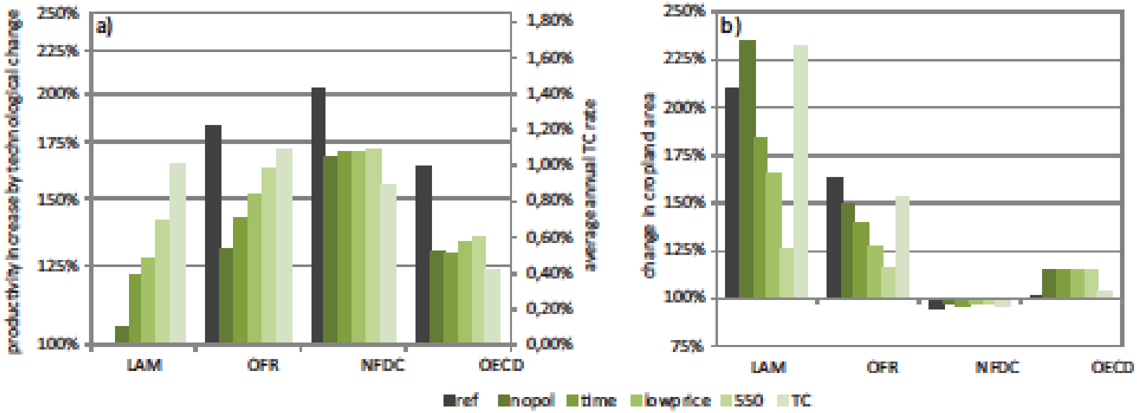

Fig. 8 a Productivity increase by technological change and respective annual TC rates and b increase in cropland of food and fodder crops in Latin America (LAM), Other Forest Regions (OFR), Non-tropical-Forest Developing Countries (NFDC) and OECD countries

Sensitivity analysis

Our model results depend largely on exogenous parameters. In order to verify the results, we regularly perform sensitivity tests with the crucial parameters. For this study, we have chosen land conversion costs (lcc) and the trade balance reduction factor, which triggers the amount of trade liberalisation. In the first case, we vary lcc from 200 US$/ha to 1,800 $/ha in 200 US$/ha steps in each scenario, which amounts to 54 model runs. The same amount of model runs is required for the second sensitivity test, in which we vary the trade balance reduction factor by 2.5, 5, 7.5 and 10 % below and above current values in each time step.

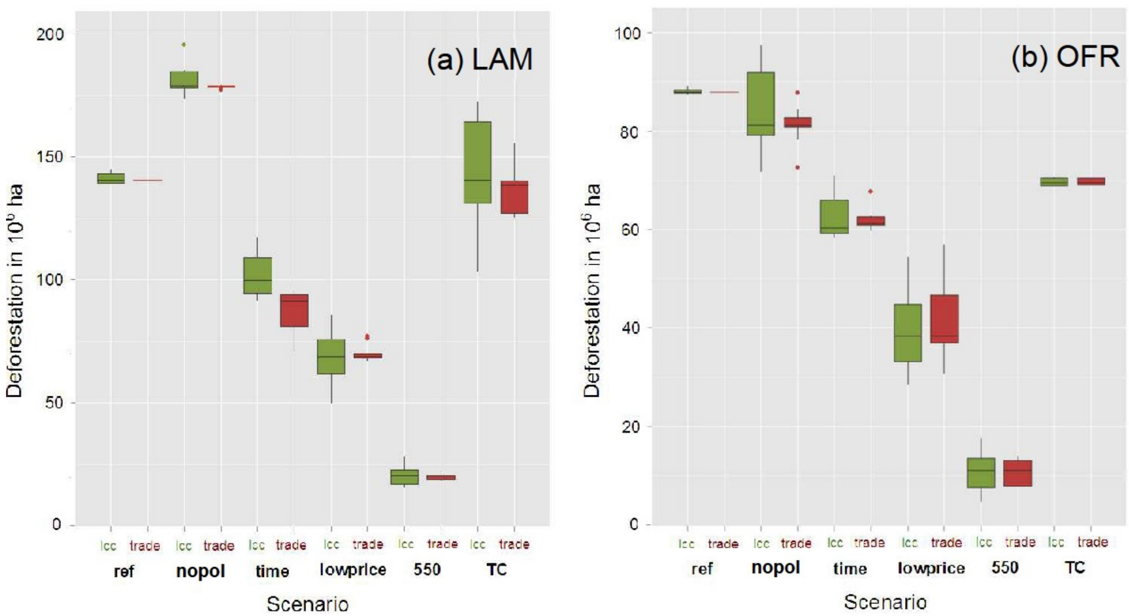

Fig. 9 Sensitivity of intact and frontier forest (IFF) area in LAM (Latin America) and OFR (Other Forest Regions) in 2050. For the analysis, land conversion costs (lcc) are varied in 200 US$/ha—steps from 200 US$/ha to 1,800 US$/ha (green boxplots) and the trade balance reduction (trade) is varied in 2.5 % steps (up to 10 %) around the current setting (red boxplots). The boxplots display minimum, lower quartile, median, upper quartile and maximum values

Resulting boxplots display the variation (minimum, lower quartile, median, upper quartile and maximum) in deforestation area of land conversion costs in green and the trade balance reduction in red for each scenario and the forest regions (LAM and OFR) (Fig. 9).

We obtain a quite heterogeneous picture with the general trend that the model outcome appears to be much more sensitive towards variations in land conversion costs than in trade liberalisation. However, in most cases, the rank order between scenarios is not affected, except two cases: The *TC* scenario in LAM and the *nopol* scenario in ORF appear to be either higher or lower in deforestation than the reference case depending on the chosen land conversion costs.

Discussion

In the preindustrial period, demand for agricultural land, fuelled by population growth, was the main driver for deforestation in temperate zones (Simmons 1987). After the industrial revolution, the situation started to change and the rising wealth of industrialised countries initiated a domestic forest transition (Meyfroidt and Lambin 2011). However, globalisation and increasing demand for goods in developed countries has shifted parts of the production to land-rich developing countries, leading to tropical deforestation (Lambin *et al.* 2001). This relation, also referred to as the virtual trade in land (Würtenberger *et al.* 2007), is triggered by the costs of trade (like tariffs, transport and information costs), which have been substantially reduced during the past century (Feenstra 1998; Jacks *et al.* 2008). Since it is likely that this trend will continue (Josling 2010), further deforestation is likely to lead to considerable damage to local environments and populations, as well as to the climate system. It is therefore relevant to examine how future growth in trade will affect deforestation rates and how different forest protection policies might influence the interplay between land expansion and trade competitiveness.

With the spatially explicit land-use model MAgPIE, we analyse effects of trade liberalisation and different forest protection policies. Compared to other global land-use models, it has the advantage that technological change and land expansion are implemented in an endogenous and competitive way. Associated investment costs are optimised together with production and transport costs on a global level. Biophysical inputs are derived from the process-driven vegetation-hydrology model LPJmL. In this study, we do not explicitly consider future scenarios of bioenergy demand, since that has been done in separate studies with the ReMIND-MAgPIE model system (Popp *et al.* 2011, 2012). As these and other studies (e.g. Gibbs *et al.* (2008)) have shown, bioenergy production only saves carbon, if the associated additional agricultural production does not come at the expense of forest land or alternatively, is achieved by agricultural productivity gains.

Nor do we here explicitly consider different governmental systems and political situations in the regions, and how this would influence investments in agriculture. A further drawback of the model is the current lack of a link between pasture and cropland expansion. As the interaction between these elements is crucial, future model development will concentrate on this link to improve the accuracy of model outcomes.

Our simulation results from 2000 to 2010 are in good agreement with observation data (FAO 2011b). For instance, in the case of Latin America, we simulate an average annual deforestation rate of 3 million ha of intact and frontier forest (IFF) compared to 4.25 million ha observed by FAO in this period. However, since FAO considers the whole unmanaged forest, the deforested IFF area in FAO statistics should be lower and much closer to our value. Nepstad *et al.* (2009) report an annual value of around 2 million ha (1996–2005), only for the Brazilian Amazon. In contrast, in Central Africa (4.5 vs. 3.4) our values are moderately higher and in Pacific Asia (2.7 vs 0.9) significantly higher than FAO observations. The large gap in Pacific Asia can be partly explained by recent reforestation efforts in this region (Lamb 2011), which are considered in FAO statistics but are not relevant for our definition of IFF.

Overall, our results show that in the main forest regions, Latin America, Sub-Sahara Africa and Pacific Asia, cropland area would significantly increase over time under constant trade and forest protection. With growing trade liberalisation, the most prominent region in terms of IFF area, Latin America, would clear an additional 40 million ha of forest area, leading to 25 Gt additional CO_2 emissions by 2050. At the same time, due to its comparative advantage, Latin America is the only region which requires higher technological change (TC) rates than in the reference case and expands its exports in each of the four major traded commodities (cereals, oilcrops, sugar, and meat). In contrast, Sub-Sahara Africa reduces its production level due to trade liberalisation. However, this decrease has no influence on the level of deforestation and is purely triggered by lower investments in technological change. In the *reference* and *nopol* scenarios, the low forest protection of the past in Africa is assumed to continue. This leads to dramatic forest loss in Central Africa in these scenarios. Although the disappearance of the whole tropical rainforest seems unrealistic, it gives an indication that especially the forests of Central Africa are likely to come under huge pressure in the future if no policy intervention is undertaken. Countries in Pacific Asia decrease their deforestation rate under liberalisation compared to the reference case.

The main reason for this is that these countries have low comparative advantages in most agricultural commodities, which leads to further imports under liberalisation. However, the pace of deforestation there still increases with liberalisation, leading to higher rates until 2020. Land-scarce regions like the Middle East, North Africa and south Asia are projected to see the highest growth in imports. With increasing liberalisation, there is less pressure to increase productivity in these regions, resulting in significantly lower investment in technological advances.

Reducing emissions from land-use change requires intervention to protect forests. We combined trade scenarios with different forest protection measures, divided into direct regulations, market instruments and compensation payments. Only in Latin America, forest protection leads to higher investment in TC. Except for some slight reductions, net export rates stay constant due to higher agricultural productivity. Hence, forest regions do not lose their competitive advantage as a consequence of forest protection.

As a direct regulation, we increased protected areas (PAs) over time. We chose the rather extreme scenario of full forest protection in order to depict the whole range of possible outcomes. Soares-Filho *et al.* (2010) tried to quantify the impact of PAs in the Amazonian rainforest and concluded that 37 % of the recent decline in deforestation was due to new PAs and 44 % due to lower agricultural activity. In another study, they estimated a reduction in deforestation of around 100 million ha by comparing a business-as-usual case with a strict governance scenario (involving an expansion of PAs and other legal protection enforcement) (Soares-Filho *et al.* 2006). Our continuously increasing rate of PAs in Latin America (*time* scenario) follows their governance scenario as far as possible and achieves savings of almost 90 million ha compared to a scenario without any further forest protection (*nopol*). Nepstad *et al.* (2009) even discussed the possibility of ending deforestation by 2020 (which is confirmed by our *550ppm* scenario), based on the assumed continuation and extension of recent efforts, like expansion of PAs, externally financed funds and regulation efforts by the agribusiness sector. However, if not monitored or applied globally, protecting forests in one place can lead to displacement of land use to other regions (Soares-Filho *et al.* 2010; Meyfroidt *et al.* 2010) and resulting carbon leakage (Wunder 2008). Although we have not directly analysed this mechanism, we observe some non-continuous effects between different time steps and scenarios. For instance, in ORF, between 2030 and 2040, deforestation is higher in the scenarios *time* and *lowprice* compared to the *nopol* scenario, whereas it is the other way round for LAM. Since agricultural area is not allowed to expand into IFF in LAM and PAS in this time step, agricultural area in Central Africa expands at the expense of IFF area. The establishment of protected areas should, therefore, be an international effort in order to avoid leakage effects and to support the political will in target countries (Soares-Filho *et al.* 2006).

As a representative policy for market instruments, we included a CO_2 price as a climate mitigation policy for avoided deforestation. With a price sufficiently high to reach the 550ppm concentration target, total emissions related to deforestation are below 10 Gt CO_2 by 2050. This rather sensitive behaviour is in line with other studies. The MiniCAM model is even more sensitive towards a CO_2 price by generating no land use-related carbon emissions in a 550ppm scenario (Wise *et al.* 2009). Its successor, the GCAM model, calculates deforestation levels under a 526ppm scenario amounting to around 30 Gt emissions between 2020 and 2050 (Thomson *et al.* 2010). Finally, the study by Kindermann *et al.* (2008) provides a comparison of three different models, GTM, DIMA and GCOMAP, by calculating marginal abatement cost curves. They show that with assumed constant carbon prices, deforestation in Latin America is fully avoided in 2020 with a CO_2 price of between 30 and 40 US$/tonne. In our study, this is already achieved with prices of 12 to 20 US$/tonne. With regard to climate mitigation, the inclusion of CO_2 prices has the advantage over other measures that the carbon intensity per unit of land is explicitly considered. As a consequence, carbon-rich vegetation is valued higher and land expansion moves to places where forests and other natural vegetation contain relatively less carbon.

Lastly, we applied a scenario of indirect forest protection in order to investigate the effect of additional growth in agricultural productivity on deforestation (*TC* scenario). Results suggest that investment in technological change could potentially reduce the pressure on tropical rainforests. However, it has to be noted that an additional yield growth of 1 % per year requires huge investment in the agricultural sector (Dietrich *et al.* 2013) and that this yield increase would not be sufficient to prevent deforestation completely. As shown by others as well, additional and complementary measures are needed (Wise *et al.* 2009; Thomson *et al.* 2010). In this context, Angelsen (2010) points out that local yield increases may encourage local deforestation and that, therefore, agriculture in low-forest areas should be supported instead of agriculture close to the forest frontier.

Conclusion

From our analysis, we draw several conclusions. First, more trade liberalisation leads to a substantial increase in tropical deforestation in Latin America, driven by the strong growth in agricultural exports. Therefore, global liberalisation efforts, for instance by the World Trade Organization (WTO), should not be undertaken without considering global forest protection measures.

Second, policies to protect forest area do not necessarily lead to losses in trade competitiveness, since the reduced land availability is compensated in most cases by higher technological change rates. This contradicts often expressed concerns that policies to protect forests reduce economic growth or international competitiveness (Banerjee *et al.* 2009).

Third, pricing CO_2 emissions from deforestation could effectively conserve large parts of the tropical rainforests, avoid over 100 Gt of carbon emissions and also preserve some of the most biodiverse ecosystems. Voluntary payments for avoided deforestation, as discussed under REDD+, can provide the same incentive.

Fourth, developed countries accelerate tropical deforestation due to their agricultural demand and should be aware of their responsibility regarding the virtual trade in land. Awareness of this problem has risen in recent years, and first attempts to tackle it have been made. The European Union, for instance, has set sustainability standards for biofuels. But also a decreasing demand for agricultural products by the global North would reduce the pressure on tropical forests.

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Unbridled international trade eats up the land-savings from pasture intensification and diminishes the potential for conservation

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Abstract

Agricultural expansion is a leading driver of habitat loss in the world's most biodiverse regions, but little is known on how future land use change may encroach on remaining natural habitat, especially when taking the level of agricultural intensification and international trade into account. Using an economic land-use model, we assessed potential losses of natural vegetation with a focus on how these may threaten biodiversity hotspots and frontier forests. We analysed agricultural expansion under proactive and reactive biodiversity protection scenarios and for different rates of pasture intensification. We found that the growing demand for food led to a significant expansion of croplands at the expense of pastures and natural vegetation. In the reference case of our study, global cropland area increased by more than 400 Mha hectares between 2015 and 2050, mostly in Africa and Latin America. Grazing intensification was a main determinant of this development. In Africa, higher rates of pasture intensification resulted in smaller losses of natural vegetation and a reduced decline in biodiversity hotspots and frontier forests. In this region, investments into raising pasture productivity in conjunction with proactive land-use planning are essential to reduce further losses of areas with high conservation value. In Latin America, in contrast, higher productivity was compensated for by more livestock exports, highlighting that uncontrolled trade can reduce the land savings of pasture intensification. Reactive protection of biodiversity hotspots significantly reduced the decline in natural ecosystems in Latin America. We conclude that protection strategies need to adapt to region-specific trade positions. In regions with a high involvement in international trade, strict protection should be preferred over strategies aimed at increasing pasture productivity. Considering region specific differences is important in maximising the effectiveness of biodiversity conservation policies.

Introduction

Globally, biodiversity is declining at alarming rates with no major change of this trend in sight (Butchart *et al.* 2010; CBD 2014; Visconti *et al.* 2016; WWF 2016). Rather, one of the main underlying drivers of biodiversity loss, namely, agricultural land use, is likely to increase with a higher demand for agricultural products in the future (Sala 2000; Tilman *et al.* 2001; Newbold *et al.* 2015; Maxwell *et al.* 2016). Global population is expected to increase from about 7.3 billion people today to between 8.5 and 10.8 billion in 2050 (UN 2015; KC and Lutz 2017). Together with an increase in wealth, and an associated increased per capita consumption, this will eventually heighten the pressure on land-based resources and on natural ecosystems and their biodiversity (Tilman *et al.* 2011; Alexandratos and Bruinsma 2012).

Despite this rising pressure, biodiversity research in the last 25 years has strongly focused on the effects of climate change on biodiversity and neglected effects of land-use change (Titeux *et al.* 2016). The global extent and impact of agriculture is so vast, it now rivals biomes in predicting species richness (Kehoe *et al.* 2016). Indeed, agriculture currently threatens over three times more species than climate change (Maxwell *et al.* 2016) and this threat is likely to remain dominant during the 21st century even when compared to other threats such as climate change and invasive species (Sala 2000; Betts *et al.* 2015). Based on a global dataset of observations, Newbold *et al.* (2015) estimate that land-use change and associated pressures have already reduced local species richness by about 14% during the last decades, and are projected to lead to a decline of another 3.4% until the end of the century. Agriculture impacts biodiversity primarily through an expansion into intact forests in the tropics (Gibbs *et al.* 2010), which are those richest in biodiversity (Mace *et al.* 2005). Especially in Latin America and Africa, agriculture is responsible for about two thirds of total deforested area (Kissinger *et al.* 2012; Hosonuma *et al.* 2012), and it is these two regions where ongoing agricultural expansion is expected to threaten biodiversity most in the future (Laurance *et al.* 2014). In arable regions of the Amazon and in tropical Africa, about one third of all species could potentially be lost due to continued agricultural expansion and intensification (Kehoe *et al.* 2017).

Protected areas are one of the most important tools for biodiversity conservation, and can effectively reduce land-use change and deforestation within their boundaries (Andam *et al.* 2008; Geldmann *et al.* 2013; Beresford *et al.* 2013). However, better planning is needed in designing and choosing the location of such protected areas in order to optimise conservation benefits (Margules and Pressey 2000). Currently the location of protected areas is biased towards places that are less likely to be affected by land conversion (Joppa and Pfaff 2009), and while this can save conservation funds, it can also miss areas that may be most at risk. A number of approaches help to inform scientists and policymakers in prioritizing conservation efforts and indicate where future protection is needed most. Brooks *et al.* (2006) summarized and classified several of these global schemes according to the two dimensions of vulnerability and irreplaceability, and accordingly into prioritization schemes that are proactive or reactive. An example of a reactive conservation prioritization template are Conservation International (CI) biodiversity hotspots (Mittermeier *et al.* 2004). They are characterized by a high number of endemic species but have also seen strong habitat loss in the past, which is likely to continue in the future. Frontier forests (Bryant *et al.* 1997), in contrast, are an example of a proactive conservation template, as the majority of these largely pristine areas are considered to be under relatively low current threat. Protected areas, however, do not come without drawbacks. They can incur leakage effects, where protection shifts the location of land-use change, so that more land-use change happens just outside the areas under protection (Lui and Coomes 2015). Leakage can also occur even in very distant places, through the connectivity of global agricultural trade and the increasing outsourcing of countries' land-use footprints (Meyfroidt *et al.* 2010; Lenzen *et al.* 2012; Meyfroidt *et al.* 2013).

Agricultural intensity determines how much area is needed to produce the same output. In the past decades, increasing yields were responsible for the bulk of the overall production increase. For example, while food production has grown by 145% since the 1960s, agricultural area has increased by only 11% (Pretty 2008). Today, more than 90% of all areas suitable for crop production outside protected areas and dense forests are already occupied by agriculture (cropland or pasture) (Zabel *et al.* 2014). Therefore, further agricultural intensification alongside strict protection schemes seems to be key for safeguarding natural ecosystems. Continuous yield increases are especially needed to conserve intact forest ecosystems under increasing demand for bioenergy (Popp *et al.* 2011). Erb *et al.* (2016) show that a global adoption of Western, meat-heavy diets and an avoidance of further deforestation would require higher pasture intensities. Grasslands are of special importance, as pastures and meadows amount to about 70% of today's agricultural land (FAOSTAT 2016), but research on the environmental consequences of agricultural land use has focused mostly on arable crops (Alexander *et al.* 2015). Pasture intensification is a key requirement for the conversion of grassland to cropland, which is an important process in many model-based projections of the future (Popp *et al.* 2017). At the same time, grazing systems have a substantial potential for productivity increases. The herbage-use efficiency, i.e. the forage removed by ruminants as a proportion of herbage growth, is below 7% globally (Chang *et al.* 2016), and the productivity is especially low in many developing countries (Herrero *et al.* 2013). In Sub-Saharan Africa and Latin America average seasonal herbage-use efficiency is only ~10 and 16% (Fetzel *et al.* 2016). Strassburg *et al.* (2014) showed that pasture intensification and the conversion to cropland had a land-saving effect in the past. In those Brazilian states where total cultivated land remained constant or shrank, an expansion of cropland was compensated for by a contraction of pasture land. However, to be land saving, intensification must be accompanied by other measures and policies that directly address the conversion of natural land, such as protected areas (Barretto *et al.* 2013; Phalan *et al.* 2016).

Land-use change and agricultural production, do not only lead to a decline in biodiversity, but also contribute to climate change as they are responsible for more than 20% of all anthropogenic greenhouse gas emissions (Tubiello *et al.* 2015), with deforestation being the main source of land-use based CO_2 emissions (Smith *et al.* 2014). The resulting change in climatic conditions represents an additional threat to biodiversity, and could further amplify the effect of habitat loss (Mantyka-Pringle *et al.* 2012). Protected areas, when managed properly, could reduce some of the land-use change emissions (Scharlemann *et al.* 2010; Soares-Filho *et al.* 2010).

In this study, we use an economic land-use model to assess the interplay of different protection policies and pasture intensification scenarios, with a special focus on areas with high conservation value. We ask whether pasture intensification could help to reduce losses in natural vegetation, and where and why it might need to be supplemented by strict protection. To this end, we construct a reactive protection scenario focusing on areas currently under high threat and a proactive prioritization scenario focusing on still largely untouched ecosystems that may be threatened in the future. We then analyse potential trade-offs between reactive and proactive conservation prioritization scenarios. A strong focus is set on Latin America and Sub-Saharan Africa as these regions have large pristine forests of high conservation value, but strongly differ in their involvement in the international trade of agricultural commodities.

Methods

The Model of Agricultural Production and its Impact on the Environment (MAgPIE 3.0) was used for this study and amended in several aspects. MAgPIE is a partial-equilibrium model of the agricultural sector and produces scenarios of future land use by minimizing agricultural production costs for a given demand for food.

Demand for agricultural products was derived from a regression of per-capita consumption on time and GDP per capita, at the level of ten world regions (Bodirsky *et al.* 2015). Projections for GDP (Dellink *et al.* 2017) and population (KC and Lutz 2017) were taken from SSP2, the middle-of-the-road scenario of the Shared Socio-economic Pathways (O'Neill *et al.* 2017). Population reaches 9.1 billion in 2050 in this scenario and per-capita GDP increases by about 50% in the most developed regions and almost 300% in the developing regions between 2015 and 2050. Production was based on potential yields for rain-fed and irrigated production from the dynamic vegetation, crop and hydrology model LPJmL (Bondeau *et al.* 2007; Müller and Robertson 2014). These yields were calibrated in two steps to represent 17 different crop types. In a first step, management intensity was chosen as to match yields with FAO data at the country level. In a second step, yields were calibrated in a way that regional cropland and pasture areas agreed with FAO data at the regional level. Livestock production was modelled for five categories with specific feed baskets, with grazing demand being estimated on the basis of feed energy requirements and available concentrate feed (Weindl *et al.* in press). The demand for forest products was not explicitly considered in the model. Instead, forestry areas in the model, based on national forestry data from FAO (2010), were kept constant over time.

In response to the increased demand for agricultural produce the model either reallocated and expanded cropland and pasture, or intensified cropland and pasture production. The starting year for the model projections was 1995, where land use was initialized to match land-use patterns by Erb *et al.* (2007) which are consistent with FAOSTAT data. Agricultural expansion was subject to land conversion costs, and cropland expansion was limited to areas suitable according to Fischer *et al.* (2002). For this study, land conversion costs were split into costs for land clearing and land establishment. Land clearing for agriculture on densely vegetated areas was assumed to be more expensive. This takes into account that the previous land type, and its vegetation density, play an important role in determining whether conversion to agriculture takes place (Jasinski *et al.* 2005). Land-clearing costs in the model ranged between 500 US$ per hectare for the region with the lowest GDP and 10000 US$ per hectare for the highest GDP in 1995 for very dense vegetation, which reflects the range previously reported (CostHelper; Pattie and Merry 1999; Simorangkir 2006). For the extreme case of no aboveground vegetation, land-clearing costs were as low as zero. Land-establishing cost, in addition, ranged between 1,000 and 10,000 US$ per hectare for cropland and 500 to 5,000 US$ for pasture. These costs represented the need for infrastructure and field preparation and were estimated based on manual model output calibration.

Agricultural intensification in the model happened through endogenous investments into yield-increasing technological change or irrigation. Costs for agricultural research and development were derived from historically observed investment-yield ratios (Dietrich *et al.* 2014). For this study, the development of agricultural yields was initially modelled endogenously, and then modified according to specific scenarios (see below). In addition, the model endogenously decided on the use of irrigation based on cost-effectiveness related to infrastructure and application (Bonsch *et al.* 2015).

Agricultural trade between the model regions was partly restricted by historical self-sufficiency ratios, while the remaining trade flows were allocated based on comparative advantage (Schmitz *et al.* 2012). Consistent with the SSP2 storyline, we assumed trade to further liberalize in the future, meaning that the amount of trade according to comparative advantages in the model increased. Transport costs within each region were derived from a map of travel times to major cities (Nelson 2008) and total regional transport costs (Narayanan and Walmsley 2008).

Land-use change emissions were calculated based on carbon stock changes, and carbon densities for the different vegetatio6n types from the LPJmL model. For the regrowth of natural vegetation we assumed logistic carbon accumulation curves (Humpenöder *et al.* 2014).

All these aforementioned factors (food demand, yields, water availability, trade, transport costs, land conversion costs) in combination determined the spatial pattern of land use in the model. Cellular data were used on a spatial resolution of 0.5 degree, while the demand side was represented by the 10 world regions (figure S1). Due to computational constraints, spatial data were aggregated to 2000 simulation units comprising cells with similar yield levels, carbon density and distance to markets (Dietrich *et al.* 2013). The model was then solved within the GAMS (2012) modelling system with the solvers CONOPT (ARKI 2012) and CPLEX (IBM 2012) and with the objective to minimize global costs of agricultural production.

Agricultural land-use change in forests and biodiversity-rich areas

We assessed the change in natural vegetation (forests and other natural vegetation) within Latin America and Africa with a special focus on conservation priority areas. To this end, we selected two well-established concepts. On the one hand, following the classification by Brooks *et al.* (2006), we focused on areas which are assumed to be highly vulnerable and therefore in strong need for protection (CI biodiversity hotspots). On the other hand, we looked at areas with a perceived lower current threat (frontier forests). The CI hotspots, represent a reactive approach that targets areas with a high number of endemic species which have already lost 70% of their native habitat (Mittermeier *et al.* 2004). In Latin America, they mainly cover the Atlantic Forest, the Cerrado, large parts of Mesoamerica, the Andes and Chilean Forests. In Africa the focus is mainly on the Horn of Africa, Madagascar and Guinean Forests of West Africa (Mittermeier *et al.* 2004). Frontier forests are defined as ecologically intact, relatively undisturbed natural forests. Protecting such areas may represent a more proactive approach to conservation. In Latin America and Africa, frontier forests include the Amazon and the Congo basin (Bryant *et al.* 1997). Geographic information of these areas was overlaid with the land-cover classes forests and other natural vegetation (figure 1).

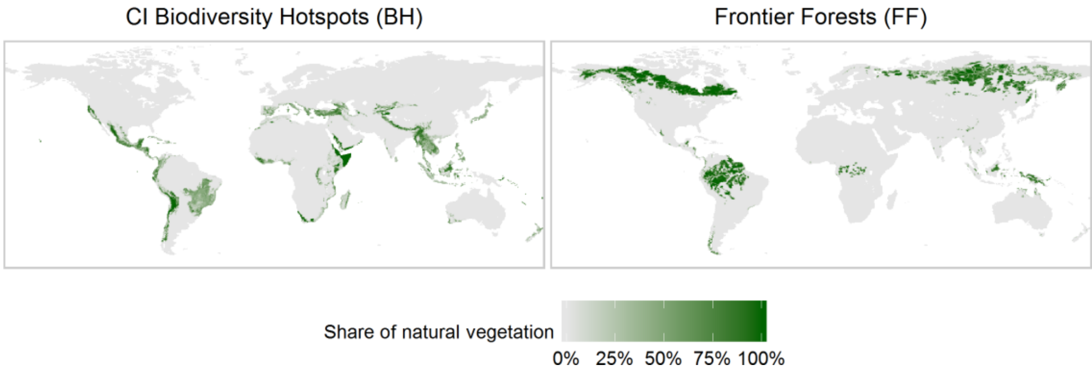

Figure 1 Natural vegetation (forests and other natural vegetation) within conservation priority areas. Biodiversity hotspot areas were assumed protected in the "BH protect" scenario, Frontier forests in the "FF protect" scenario

Scenarios

Within the modelling framework we analysed several scenarios with different protection of conservation priority areas and different levels of pasture intensification (table 1), and assessed consequences for the extent of natural vegetation. Our reference case for this study was a scenario parameterized according to the SSP2 storyline. For this we assumed a base level protection of terrestrial natural vegetation (forests and other natural vegetation) according to the protected areas from the World Database on Protected Areas (WDPA) with IUCN categories Ia, Ib and II (UNEP-WCMC 2016). Two scenarios were constructed where all natural vegetation within either biodiversity hotspots (BH) or frontier forests (FF) was assumed to be protected. We acknowledge that fully protecting either biodiversity hotspots or frontier forests in order to prevent land-use change are strong assumptions, but we were interested in exploring the effect of two very different conservation approaches. Intensification was first determined endogenously by a model run based on cost effectiveness and represents our 100% intensification case. For the other scenarios the endogenously derived pasture intensification rates were modified systematically from the year 2015 onwards. In Latin America or Africa, they were reduced to 0%, lowered by 50% or increased to 150% compared to the endogenous modelled 100% case. For all other regions they remained as in the reference case (100%) (see also figure S2).

Table 1 Description of scenarios. Scenarios were constructed as a matrix of different protection and intensification combinations. Pasture intensification sensitivity was studied separately for Latin America and Africa. The reference case is a combination of current protection and 100% intensification.

Protection scenarios	
Current protect	Forests and other natural vegetation protected according to the World Database on Protected Areas (WDPA)
BH protect	All forests and other natural vegetation within CI biodiversity hotspots (see figure 1) protected from land-use conversion
FF protect	All forests and other natural vegetation within frontier forests (see figure 1) protected
Pasture intensification sensitivity	
0%	No pasture intensification beyond the 2015 levels in either Latin America or Africa. According to reference (100%) in all other model regions
50%	Pasture intensification of only 50% of the reference case in Latin America or Africa
100%	Pasture biomass usage according to the endogenously modelled development in the reference case
150%	Pasture intensification of 150% compared to the reference case in Latin America or Africa

Results

Agricultural land-use change

An increasing demand for food led to an ongoing expansion of croplands between 2015 and 2050 at the global level, and more than 400 Mha hectares of cropland were newly established in the reference case of our study. Two processes dominate the picture (see figure 2): intensification and conversion of pastures to cropland (-289 Mha); and a significant loss of forests (-87 Mha).

Two regions stick out as they feature large areas of both pasture and unprotected forests, which are suitable for conversion into cropland, making them especially prone to future land-use change: Latin America and Africa. This is in strong contrast to other regions with similarly high shares of forests, like the Former Soviet Union (FSU), where only a minor share of the forest area (about 7%) is considered suitable for the conversion to cropland. Latin America and Sub-Saharan Africa also account for the bulk of the projected land-use change and more than 80% of global deforestation until 2050, Africa with a decline in pasture land by 127 Mha and deforestation of 49 Mha, and Latin America with a conversion of 46 Mha of pasture land and 27 Mha of forest loss. Considering this, we focus our subsequent analysis on these two regions.

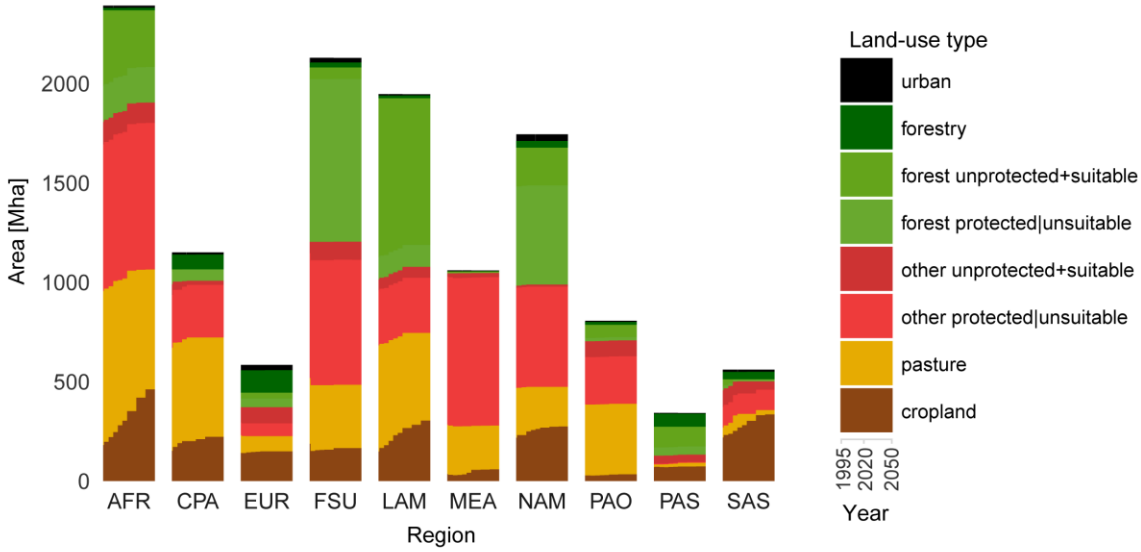

Figure 2 Modelled land-use change over time in the reference case (current protection and 100% pasture intensification). Sum of one bar represents the total land area of one model region. AFR: Sub-Saharan Africa, CPA: Centrally planned Asia, EUR: Europe, FSU: States of the former Soviet Union, LAM: Latin America, MEA: Middle East/North Africa, NAM: North America, PAO: Pacific OECD (incl. Japan, Australia, New Zealand), PAS: Pacific Asia, SAS: South Asia.

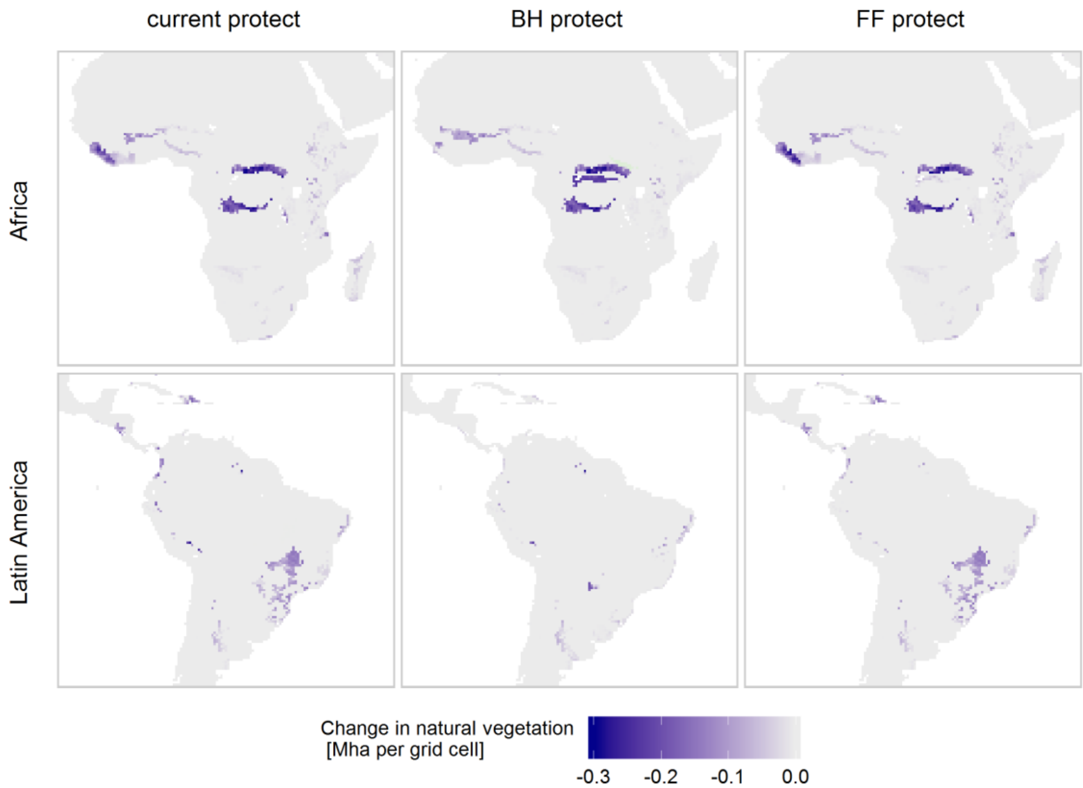

Figure 3 Maps of projected losses of natural vegetation (forests + other natural vegetation) between 2015 and 2050 in Africa and Latin America. For the endogenously determined, 100% intensification assumption.

In the reference case, where the model determined pasture intensification internally based on cost effectiveness (100% intensification) and under current protection, land conversion focused mainly on southern Brazil in Latin America and the Congo Basin in Africa (figure 3). A protection of biodiversity hotspots substantially reduced the loss of natural vegetation in Latin America, while in Africa it rather led to a reallocation of agricultural expansion from Western Africa towards Central Africa. Under this level of intensification, the loss of frontier forests was rather small, and therefore their protection had only limited influence.

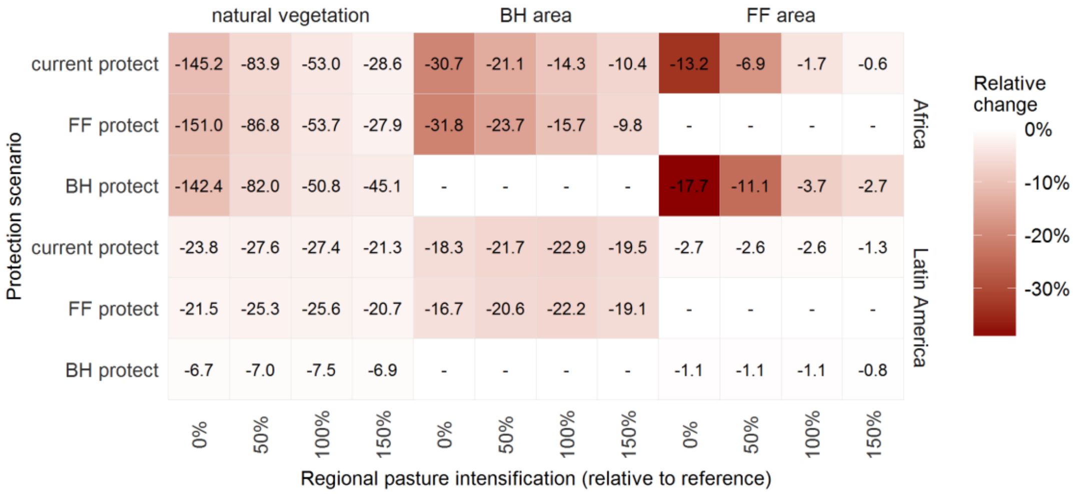

Figure 4 Loss of natural vegetation as a function of protection and intensification. Model-based changes in forest area and other natural vegetation between 2015 and 2050. Numbers show the absolute area change in Mha, colours represent the relative change compared to 2015. Left columns (natural vegetation) refer to the sum of all forests and other natural vegetation within the region. BH area and FF area refer to the natural vegetation within biodiversity hotspots (BH) and frontier forests (FF). "FF protect" and "BH protect" are scenarios where all natural vegetation within frontier forest or biodiversity hotspots were assumed protected from land conversion.

The modelled area of natural vegetation in Latin America and Africa was affected in different ways by protection and pasture intensification. In Africa, intensification had a strong influence on the loss of natural vegetation (figure 4). Lower rates of intensification led to higher losses of forests and other natural vegetation in this region, regardless of the policy scenario. For the current protection case, the loss of natural vegetation in Africa ranged between 145 Mha for no additional pasture intensification after 2015 to 28 Mha for 150% yield levels compared to the reference case (see also figure S2). In other words, strong intensification in Africa could result in land savings in 2050 larger than the total area of Ethiopia. Under zero pasture intensification more than 17 Mha of African frontier forests were lost. The protection of frontier forests (FF) or biodiversity hotspots (BH) had relatively low influence on the overall loss of natural vegetation, independent of the intensification level.

In Latin America, in contrast, the area of natural vegetation was influenced much less by the level of pasture intensification, but depended more strongly on the protection of biodiversity hotspots. The protection of BH reduced the loss of natural vegetation in the 100% intensification scenario by more than 70%, from 27.4 Mha to 7.5 Mha and resulted in hardly any reallocation of land conversion to other locations within Latin America.

Biodiversity hotspots and frontier forests in Africa and Latin America were similarly influenced by intensification as the overall area of natural vegetation in these regions.

Higher rates of intensification lowered the decline in African CI biodiversity hotspots and frontier forests, while no such trend was observed for Latin America. A leakage effect that occurred in Africa deserves special attention. When BH areas were protected this increased the loss of frontier forests, especially in combination with low rates of pasture intensification (see figure 4).

Emissions

The land-saving effect of pasture intensification in Africa and its absence in Latin America were also reflected by the land-use change emissions (figure 5). Especially in Africa pasture intensification provided the co-benefit of reducing land-use change emissions. Independently of the protection scenario, higher rates of pasture intensification reduced land-use change emissions in Africa. In the current protection case, emissions amounted to 109 Gt CO_2 without further pasture intensification, but dropped to 44 Gt CO_2 in the 150% intensification case. In Latin America, the effect of intensification on emissions was much weaker and they ranged only between 25 and 33 Gt CO_2. The protection of biodiversity hotspots and frontier forests did not help to mitigate emissions in Africa. In Latin America, protection, especially of biodiversity hotspots, showed a small co-benefit for emissions.

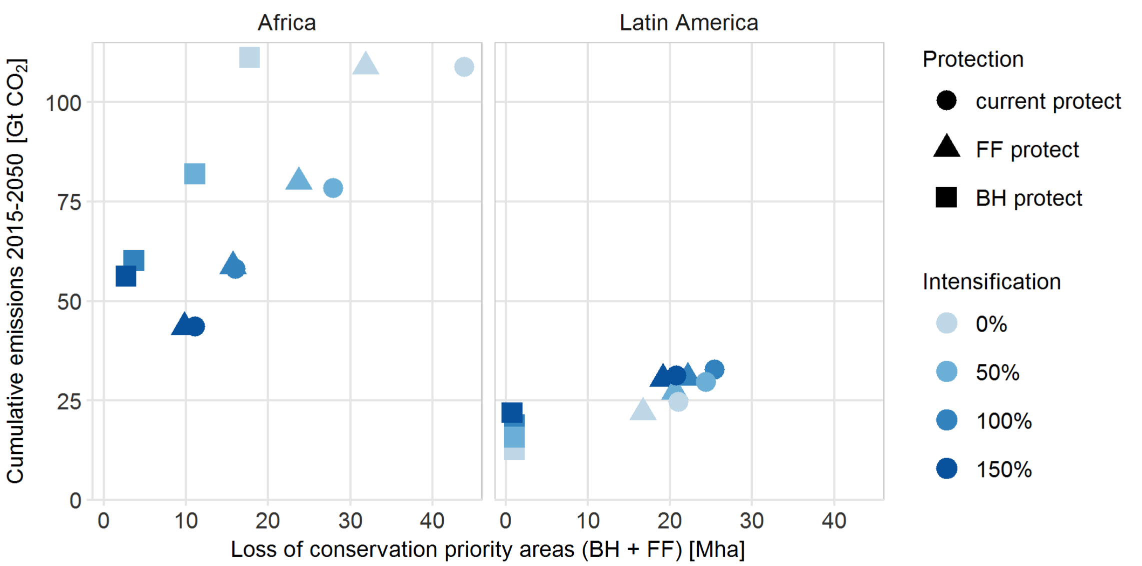

Figure 5 Co-benefits of protection and grazing intensification on cumulative land-use change emissions and the loss of conservation priority areas

The influence of trade

Foreign agricultural trade estimates revealed major differences between the two focus regions. While in 2050 Latin America became a major exporting region for crop and livestock products in the model, crop exports from Africa were comparably low and it relied stronger on livestock imports than today. Consequently, varying pasture intensities had a different impact on trade depending on the region in question. In Latin America, lower exports of livestock products compensated for lower pasture intensity, while crop exports were rather inelastic (figure 6). In Africa, in contrast, livestock imports remained almost unchanged in light of varying levels of pasture intensity, and crop exports were only slightly influenced by the level of pasture intensity.

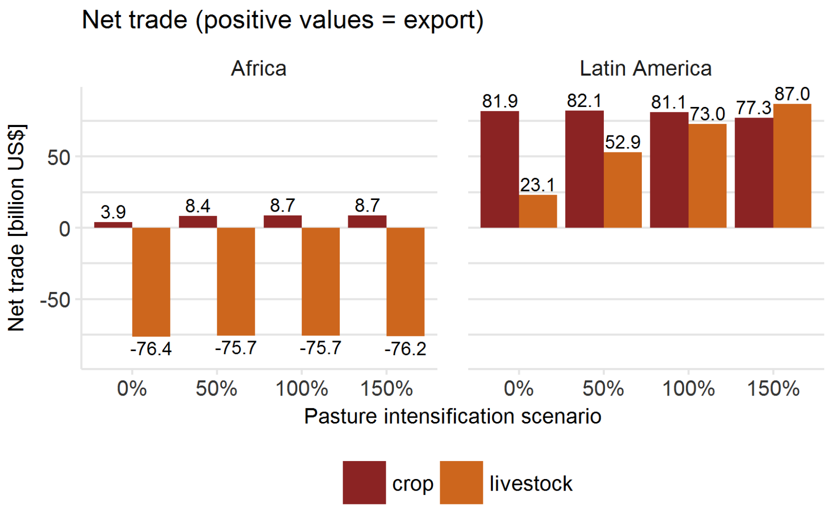

Figure 6 Model-based agricultural trade balance for Latin America and Africa in 2050

Discussion

Without suitable policies, the growing demand for food is likely to lead to continuous losses of natural vegetation in Africa and Latin America until the middle of the century. More than 80 Mha may be lost between 2015 and 2050, according to our model-based analysis. This land-use change also happens in areas rich in biodiversity. Our study confirms that especially biodiversity hotspots that have seen significant declines in primary vegetation in the past (Myers *et al.* 2000), will undergo further losses of natural vegetation in Africa and Latin America if no adequate measures are taken.

The two focus regions in this study, Latin America and Africa, showed very different responses to the protection and intensification scenarios. The total area of natural vegetation converted to agriculture in Latin America reacted relatively insensitively to the scenarios of pasture intensification assessed here. In Africa, we observed a stark contrast where pasture intensification was a key determinant of future land use. Higher rates of pasture intensification consistently led to lower rates of land-use conversion. Both regions faced strong increases in domestic demand (see also figure S4), and differences were mainly explained by their different role in international trade. While Latin America was a strong livestock exporter by 2050, Africa remained a strong importer. Lower rates of pasture intensification resulted in lower exports of meat from Latin America, while the same thing directly translated into an expansion of agricultural area in Africa. International trade is therefore a key determinant of the domestic land-saving effect of intensification. This is in line with Hertel *et al.* (2014), who argue that an African Green Revolution would only be land-saving if crop markets remain relatively segmented. Our study suggests that trade can eat up the land savings from pasture intensification. In our model, productivity increases in Latin America were almost entirely turned into higher exports. We conclude that in order to conserve key biodiversity areas this aspect has to be taken into consideration. Pasture intensification is especially effective in regions where a growing demand for food would otherwise directly translate into a further expansion of agricultural areas. It is, however, not land saving when it simply increases export production.

This study also revealed an important leakage effect, namely an increased loss of frontier forests under a protection of biodiversity hotspots. In Africa, this shift of agricultural expansion from biodiversity hotspots to pristine forests was observed in our study, especially in combination with low rates of pasture intensification. Policies aimed at reducing the loss of natural ecosystems need to consider this potential leakage effect. Other leakage effects were not analysed in depth by this study, as we were mostly interested in the domestic effect of the assessed policies. Policies that affect production in one region will, however, also influence the production in other regions. This is because, agricultural production has strong interlinkages between different production systems (e.g. between soybean and cattle ranching) and is coupled across regions through trade and demand (Gasparri and le Polain de Waroux 2015).

The land-saving effect of intensification

Our study confirms that intensification does not always result in a land-saving effect, and the consequences of pasture intensification may strongly differ across regions. The land saving effect of intensification has often been questioned (Angelsen and Kaimowitz 2001; Rudel *et al.* 2009). Kaimowitz and Angelsen (2008) argue that a higher productivity in cattle production will not automatically lead to a reduced pressure on forests. It may enable farmers to increase their herds, reduce prices and increase demand, or lead to additional exports that diminish the land saving effect. This phenomenon is also referred to as the "Jevons' paradox" or rebound effect. We found that higher pasture intensity in Latin America increased regional production, but it did not lead to considerable land saving, as livestock exports increased simultaneously. Changes in domestic demand in response to changing prices were not assessed in this study. We are aware that this is a limitation of our modelling approach, but it also more clearly depicted the importance of trade for the effectiveness of biodiversity protection policies. We would also argue that the price effect on demand is likely to decrease in the near future as tropical regions are projected to become wealthier. The GDP figures from Dellink *et al.* (2017), which are underlying our demand scenario, assume to more than double between 2015 and 2050 in Latin America and more than triple in Africa. This increases the meat consumption in these regions in the model, but also lowers the share of income spend on food and likely also reduces the price elasticity of food demand significantly (Muhammad *et al.* 2011). The decrease in food expenditure is largely independent of the level of pasture intensification (figure S5).

A number of studies have empirically assessed the land saving effect of intensification and have sometimes found only limited evidence for its existence. We argue that this is partly because many empirical studies only consider a net decline in absolute or per-capita agricultural land consumption as land-saving, since observed data provides no information on what would have happened without intensification. Rudel *et al.* (2009) analysed when crop yield increases coincided with declines in cultivated area and concluded that this combination was occurring infrequently and could often be explained by increased imports and stricter conservation policy. Ewers *et al.* (2009) found a slight decline in per-capita cropland area with increasing staple yields in developing countries, but no such effect in developed countries. Barretto *et al.* (2013) showed that in consolidated agricultural areas in Brazil, where land is constrained, yields increased and pastures were replaced by cropland. At the agricultural frontier, where land is abundant and inexpensive, intensification often coincided with expansion. In our modelling study, we were able to assess different levels of pasture intensification, and could therefore define land-savings as any reduction of land-use conversion below the baseline observed in the scenario without intensification. Especially in tropical countries, where incomes and per-capita food consumption increase strongly, constant or only slowly increasing agricultural areas may already represent an important land-saving effect. This is also confirmed by Hertel *et al.* (2014) who found that yield increases in the past were accompanied by increased cropland, but lower yield increases would have resulted in even higher land conversion. A study by Cohn *et al.* (2014), also based on an economic land-use model, shows that the way cattle ranching intensification is promoted matters for its land saving effect. A subsidy to farmers for adopting intensification practices reduced prices and increased exports from Brazil, while a tax for not doing so decreased exports and reduced deforestation more effectively.

Effects on biodiversity

Whether protection combined with high intensity production or wildlife-friendly farming is better for biodiversity is also a hotly debated topic in conservation research (Green *et al.* 2005; Kremen 2015). Agricultural expansion is not the only driver of biodiversity losses, and agricultural intensification, depending on the means used to intensify, can negatively affect biodiversity (Tscharntke *et al.* 2005; Firbank *et al.* 2008; Allan *et al.* 2015; Gossner *et al.* 2016). This is especially so on pastures, where conventional intensification is typically connected to lower species abundance and richness (Newbold *et al.* 2015).

Our study shows that in some cases higher rates of intensification could reduce the loss of natural vegetation, also within areas of high value for biodiversity conservation. However, we have not assessed the direct effect this grassland intensification would have on farmland biodiversity, and can therefore not estimate the total effect on biodiversity in our scenarios. Any reduction in habitat loss would have to be weighed against the decline in biodiversity potentially caused by intensification. To stop or at least slow down the decline of biodiversity it is likely that a combination of areas without land use, under low-intensity farming, and highly-intensified agriculture are needed (Kremen 2015; Butsic and Kuemmerle 2015). The share of these systems would depend on the regional context, including the biodiversity value and the agricultural suitability and production potential.

Model boundaries

Protected areas are seen as a cornerstone of biodiversity conservation (Gaston *et al.* 2008). Here we created simplified scenarios where either CI biodiversity hotspots or frontier forests were assumed protected, which avoided any additional loss of forest or other natural vegetation within their boundaries. We are aware that these conservation schemes were not meant as direct blueprints for protection, and a full protection of these areas is likely not feasible given the enormous pressures on land, the high costs for adequate management, and the fact that may hotspots have already lost substantial parts of their natural vegetation (Sloan *et al.* 2014). We also acknowledge that while protected areas have been found to significantly reduce land-use change, they do not completely prevent it (Andam *et al.* 2008; Beresford *et al.* 2013; Heino *et al.* 2015; Potapov *et al.* 2017). It was, however, not our aim to create realistic scenarios of protection, which was beyond the possibilities of this global modelling approach; instead we aimed to contrast the potential consequences of two very different conservation approaches. This way we could analyse whether protection of some areas could potentially lead to leakage to other areas, and allowed us to identify the main determinants of the land saving effect, such as the influence of trade.

For this study, we used scenarios where pasture intensities were prescribed based on an a priori endogenously modelled trajectory. In general, data on land-use intensity for today is scarce (Kuemmerle *et al.* 2013), and reasonable estimates for the future are difficult. However, we do know that grazing systems in Latin America and Africa are currently far below their ecological potential, so that substantial intensification could be achieved (Latawiec *et al.* 2014; Fetzel *et al.* 2016). Our approach should be seen a sensitivity analysis of pasture intensity, and we did not consider in detail which techniques would be needed to reach the respective intensification. Here we also neglected that the protection of natural ecosystems could by itself result in higher rates of intensification (Schmitz *et al.* 2015; Eitelberg *et al.* 2016).

Future land-use change was modelled with an economic model, and two of our main assumptions are that by modelling agriculture we account for the most important factor of land-use change, and secondly that agricultural land-use patterns follow market principles and can be modelled through cost minimization. Agriculture remains to be the most important driver of deforestation in Latin America and Africa, while other processes such as mining, infrastructure, urban expansion and timber production are of less importance for deforestation (Boucher *et al.* 2011; Kissinger *et al.* 2012; Hosonuma *et al.* 2012). Whether agricultural land-use change can be modelled with an economic model depends on the level to which farmers act according to market principles. Small-scale subsistence farmers may decide on the location of their farms to a lesser extent based on cost optimality. Nevertheless, even though about 50% of all agricultural land is operated by family farms (Graeub *et al.* 2016), their importance for deforestation has decreased in recent decades (Boucher *et al.* 2011). Also is subsistence farming without any purchasing or selling of food is already relatively uncommon (van Vliet *et al.* 2015), which means that almost all farms are to some degree influenced by market conditions, which are reflected by our model. We are therefore convinced that we have considered the most important drivers of agricultural expansion.

Outlook

Our study underpins that there is no universal conservation strategy suitable for different world regions, but protection strategies need to consider thoroughly the specific drivers of agricultural land-use change. We assume that African food demand, especially for livestock products, will increase strongly until the middle of the century. The degree to which this demand is satisfied through area expansion also depends on the amount of pasture intensification, where intensification could, in combination with protection, help to reduce the loss of important biodiversity hotspots. In Latin America, livestock production is driven by a mix of domestic and foreign demand. In this case, strict protection of areas most prone to land use conversion is most promising, reduces the potential for agricultural expansion, and thus helps to prevent the loss of natural vegetation.

We have here concentrated on the absolute loss of natural vegetation within frontier forests and biodiversity hotspots under different levels of pasture intensification. Future research will need to look more closely at the effect these different strategies have for biodiversity. When considering the overarching issue of producing enough food without causing harm to biodiversity, two other avenues of research merit more attention: first, detangling the conditions under which potential land saving through conventional intensification outweighs the negative effects on biodiversity, and second, investigating the various promising methods for sustainable intensification where yields are increased at minimal harm to biodiversity (Perfecto and Vandermeer 2010; Chappell and LaValle 2011; Murgueitio *et al.* 2011; Tscharntke *et al.* 2012).

Of course, other measures besides intensification and protection could also help to reduce agricultural expansion. Especially a reduction in agricultural demand through lower food waste and reduced livestock consumption could decrease land conversion substantially (Foley *et al.* 2011; Bajželj *et al.* 2014; Erb *et al.* 2016). Such demand side mitigation options may also reduce food prices (Stevanović *et al.* 2017), and could simultaneously reduce greenhouse gas emissions and negative health effects (Tilman and Clark 2014).

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Supporting information

Model parameterization

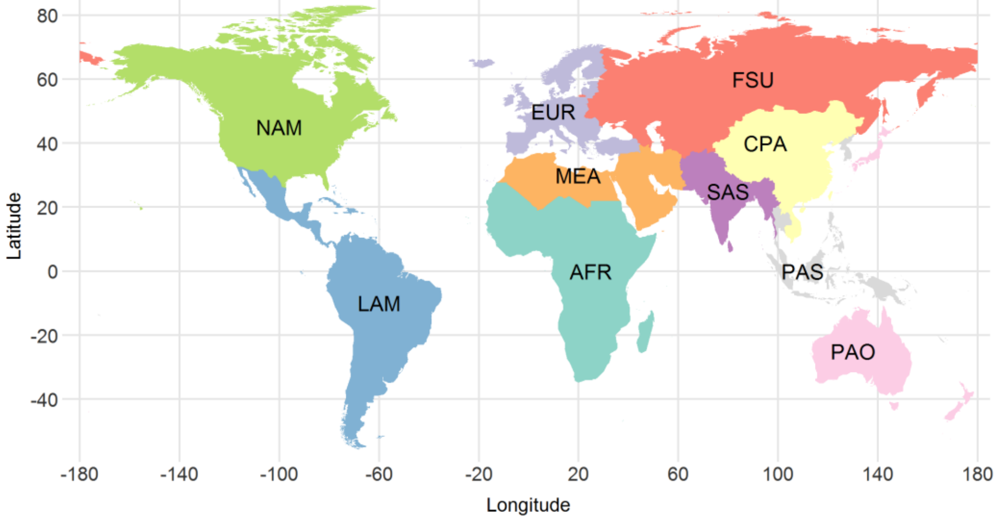

Figure S1 MAgPIE model regions
AFR: Sub-Saharan Africa, CPA: Centrally planned Asia, EUR: Europe, FSU: States of the former Soviet Union, LAM: Latin America, MEA: Middle East/North Africa, NAM: North America, PAO: Pacific OECD (incl. Japan, Australia, New Zealand), PAS: Pacific Asia, SAS: South Asia

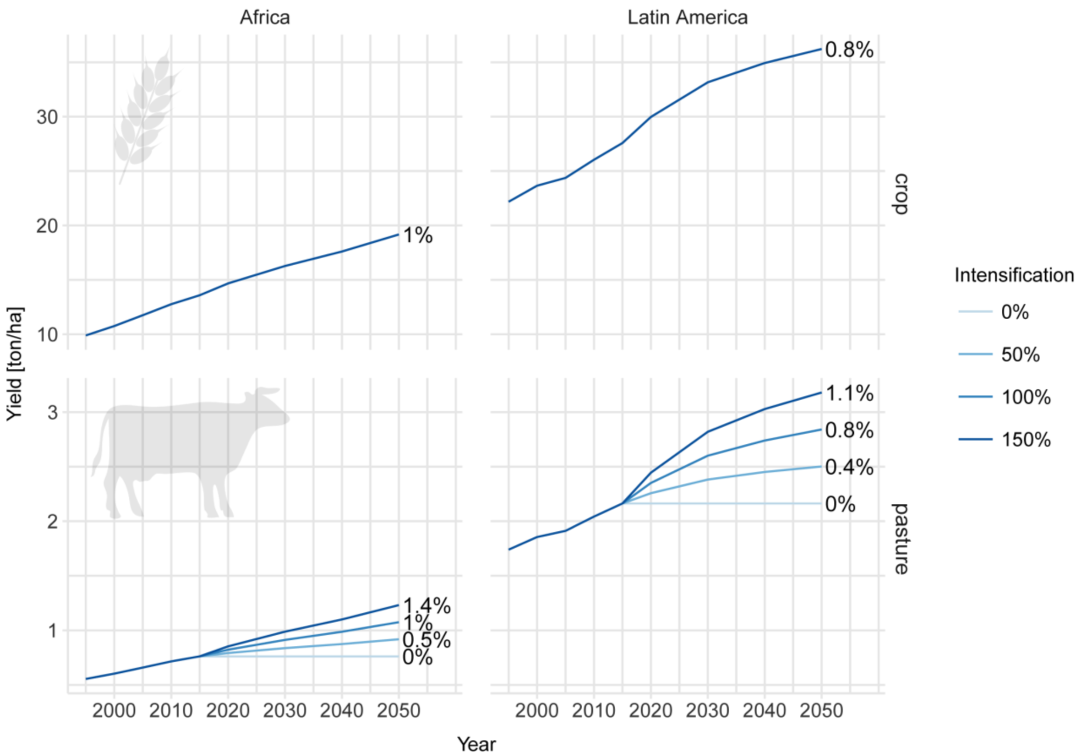

Figure S2 Rain-fed cropland and pasture yields
100% intensification refers to an endogenous modelling of R&D investments. Pasture yields are varied systematically in comparison to this, while for all other regions they remain at the 100% level. Figures in the graph represent the average annual yield increases between 2015 and 2050.

Results

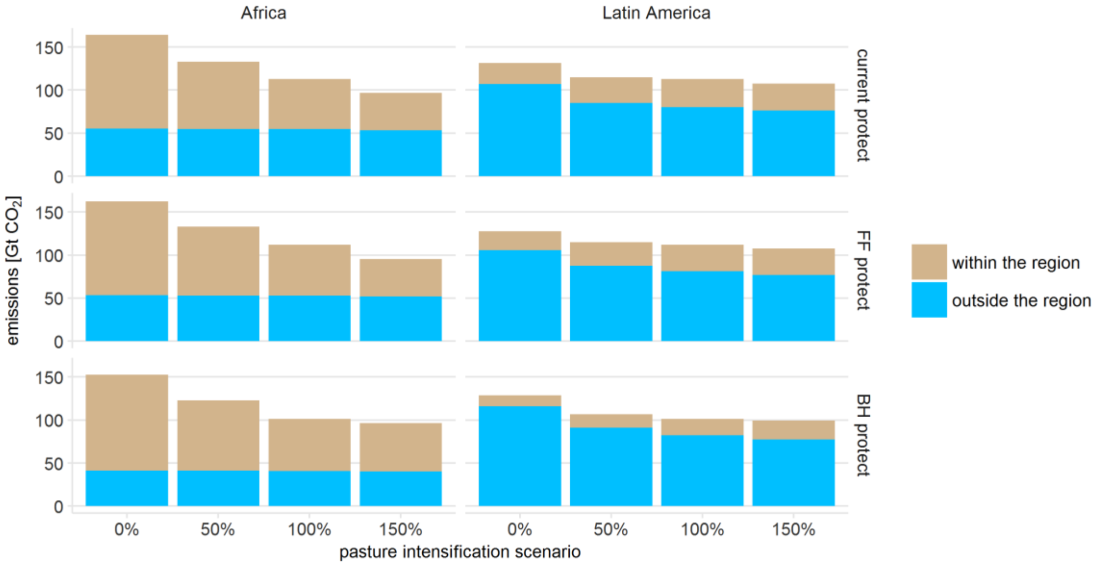

Figure S3 Global cumulative land-use change CO_2 emissions between 2015 and 2050
Shown for all simulated protection and pasture intensification scenarios. Blue parts of the bars represent the fraction of emissions not caused within the respective region, e.g. in the case of Africa (left side) these are the summed up emissions from all nine other model regions including Latin America

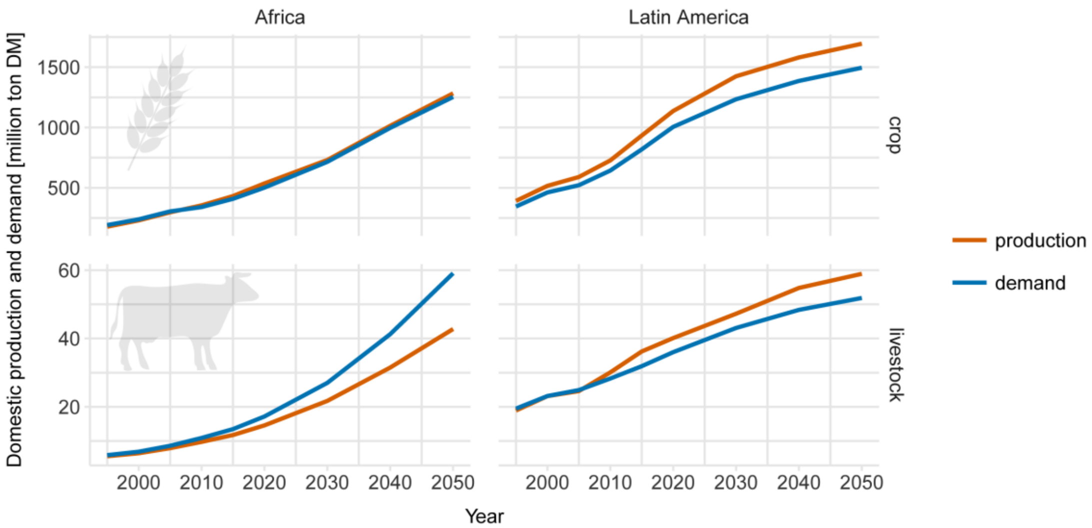

Figure S4 Domestic production and demand
Production and demand of livestock and crop products in Latin America and Africa in the reference case

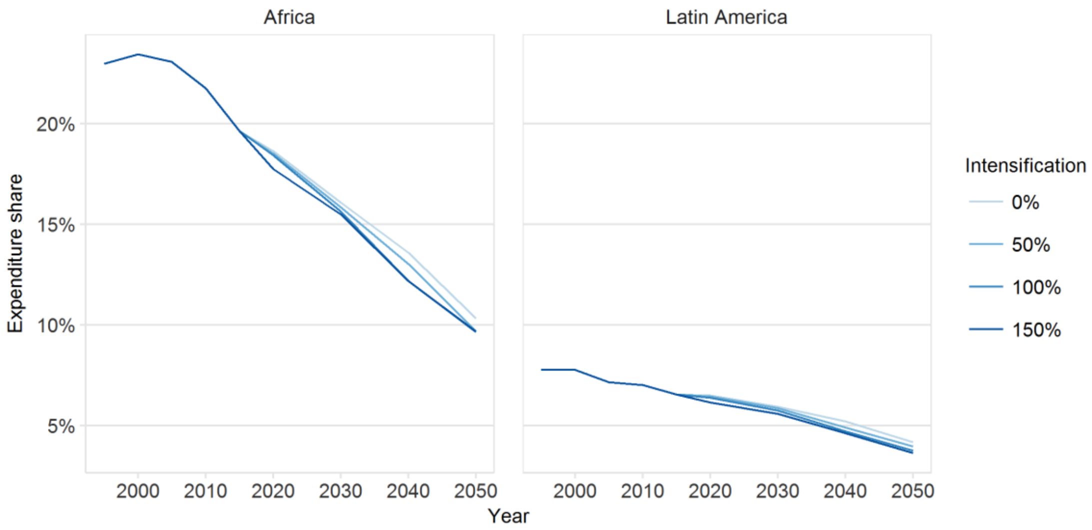

Figure S5 Modelled expenditure shares for the current protection scenario and different levels of pasture intensification
Calculated as the product of per-capita demand and modelled consumer prices for all products and divided by the assumed per-capita GDP. Prices in the model represent raw agricultural products without the costs of further processing.

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Afforestation to mitigate climate change: impacts on food prices under consideration of albedo effects

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Abstract

Ambitious climate targets, such as the 2°C target, are likely to require the removal of carbon dioxide from the atmosphere. Afforestation is one such mitigation option but could, through the competition for land, also lead to food prices hikes. In addition, afforestation often decreases land-surface albedo and the amount of short-wave radiation reflected back to space, which results in a warming effect. In particular in the boreal zone, such biophysical warming effects following from afforestation are estimated to offset the cooling effect from carbon sequestration. We assessed the food price response of afforestation, and considered the albedo effect with scenarios in which afforestation was restricted to certain latitudinal zones. In our study, afforestation was incentivized by a globally uniform reward for carbon uptake in the terrestrial biosphere. This resulted in large-scale afforestation (2580 Mha globally) and substantial carbon sequestration (860 $GtCO_2$) up to the end of the century. However, it was also associated with an increase in food prices of about 80% by 2050 and a more than fourfold increase by 2100. When afforestation was restricted to the tropics the food price response was substantially reduced, while still almost 60% cumulative carbon sequestration was achieved. In the medium term, the increase in prices was then lower than the increase in income underlying our scenario projections. Moreover, our results indicate that more liberalized trade in agricultural commodities could buffer the food price increases following from afforestation in tropical regions.

Introduction

To achieve ambitious climate targets, such as limiting global mean temperature increase to below 2°C compared to preindustrial levels, a strong decline in global greenhouse gas (GHG) emissions is urgently needed (Clarke *et al* 2014). Yet simply reducing GHG emissions might not be sufficient, or might only be achievable at high cost, so that carbon dioxide removal from the atmosphere (CDR) could become necessary in the second half of the century. Accordingly, most scenarios of the fifth assessment report of the IPCC (AR5) that are consistent with the 2°C target include negative net CO_2 emissions (Clarke *et al*

2014, Fuss *et al* 2014). This is also acknowledged in the recent Paris Agreement of the UNFCCC, in which parties agreed to aim for a balance between anthropogenic emissions and sinks of greenhouse gases in the second half of the century (UNFCCC 2015). Land-based mitigation strategies such as afforestation and avoided deforestation could make important contributions to achieving this target (Smith *et al* 2014).

Afforestation offers a high carbon sequestration potential at moderate cost, and could therefore become an alternative to or could complement other mitigation options. Cost estimates for afforestation are lower than for other carbon removal technologies such as bioenergy with carbon capture and storage (BECCS) and by an order of magnitude lower than for direct air capture (DAC) (Smith *et al* 2015). Strengers *et al* (2008) calculated supply curves of afforestation on abandoned agricultural land and found that in 2075 more than 50% of the overall potential could be supplied at costs of less than 200 $/tC, which is relatively cheap compared to other mitigation options. Edmonds *et al* (2013) showed that a 2°C warming at the end of the century would be possible without BECCS, but would require substantial carbon sequestration through afforestation, especially if mitigation action is delayed in some countries. Calvin *et al* (2014) illustrated that afforestation is an economically attractive option. When in their study a carbon tax consistent with limiting radiative forcing to 3.7 W/m^2 was applied to the energy and land-use system, global forest area increased by about 20%. Humpenöder *et al* (2014) found that a reward for terrestrial carbon uptake could provide an incentive for large-scale afforestation, resulting in cumulative removal of more than 700 Gt CO_2 by 2095. With such a huge potential, afforestation could play a considerable part in climate change mitigation efforts.

On the downside, large-scale afforestation might lead to a considerable increase in food prices through increasing competition for land between forest and agricultural production. Similar concerns have been raised in the past with regard to first-generation biofuel production, but the demand for biofuel was only one factor of many that contributed to food price hikes in recent years and its contribution was estimated to be rather modest (Mueller *et al* 2011, Persson 2015). Similarly, a model intercomparison study showed that second-generation bioenergy production consistent with the 2°C target could result in rather moderate food price increases up to 2050 if the land available for the expansion of agriculture were not restricted and if necessary investments into technology and development (R&D) were anticipated (Lotze-Campen *et al* 2014). Afforestation, however, may need substantially more area to achieve a similar level of carbon dioxide removal to BECCS (Humpenöder *et al* 2014), and could therefore have a much stronger influence on land-use competition. Bioenergy crops are harvested regularly, while once established, forests need to be maintained also under declining carbon accumulation rates if the carbon is to remain stored. Wise *et al* (2009) found that a carbon tax on terrestrial and industrial emissions could lead to an expansion of managed forests but also to a more than doubling of corn prices. In a study by Reilly *et al* (2012) a price on land carbon emissions created an incentive to reforest but also increased food prices. Calvin *et al* (2014) assessed the effect of afforestation with the integrated assessment model GCAM and found that wheat prices increased to 320% in 2095 compared to 2005 values.

The effectiveness of afforestation for climate mitigation differs depending on the location, making its application unfavourable in some regions. This is because establishing forests leads to two effects that often have an opposing influence on the average global temperature. On the one hand, while growing, trees take up carbon from the atmosphere and store it in their biomass (biogeochemical effect). On the other hand, changing land-cover to trees also affects the amount of short-wave radiation reflected back to space (biogeophysical effect), directly by surface albedo and indirectly by the contribution to cloud formation. This biogeophysical effect varies as a function of latitude (Bonan 2008). Several studies with earth system models have shown that an expansion of forest in the tropics results in cooling, while afforestation in the boreal zone might have only a limited effect or might even result in global warming (Bala *et al* 2007, Bathiany *et al* 2010, Arora and Montenegro 2011). Bright (2015) and Bright *et al* (2015) provide a good overview over the biogeochemical and biophysical processes that affect global and local temperatures as a consequence of land-cover and management change.

In the study presented here, we assessed global and regional food price impacts of large-scale afforestation with the Model of Agricultural Production and its Impacts on the Environment (MAgPIE). Earlier studies, using similar methods, have assessed bioenergy potentials (van Vuuren *et al* 2009, Erb *et al* 2012), requirements for and consequences of forest and biodiversity protection (Kraxner *et al* 2013, Overmars *et al* 2014, Erb *et al* 2016) or estimated climate change impacts on food prices (Delincé *et al* 2015). Five scenarios were analysed, one in which a CO_2 price on land-use-change emissions avoids deforestation and three where the CO_2 price created an additional incentive for afforestation. In these cases afforestation was either unrestricted, prevented in the boreal zone, or limited to the tropical zone. These scenarios were compared to a business-as-usual case without emission pricing. As afforestation was expected to increase food prices, we furthermore assessed whether more liberalized trade conditions could have an alleviating effect on food prices.

Methods

The land-use model MAgPIE

Future land-use, carbon sequestration and food price development as affected by afforestation were modelled with the partial equilibrium model MAgPIE (Lotze-Campen *et al* 2008, Popp *et al* 2014, Humpenöder *et al* 2014, 2015). MAgPIE is an agro-economic land-use model that minimizes the global costs of agricultural production for a given agricultural demand under a set of economic and biophysical constraints. By this it computes optimal, spatially explicit future land-use patterns in five-year time steps.

Agricultural demand in the model is based on projections of future population and gross domestic product (GDP) of the SSP2 scenario (Dellink *et al* 2015, KC and Lutz 2014, O'Neill *et al* 2015). This scenario assumes that global population peaks in 2070 at 9.4 billion people, while per capita GDP continues to increase until 2100. Future demand for calories and livestock share in consumption are derived through a regression model that has been estimated with historical data for calories consumed and GDP development (Bodirsky *et al* 2015) (see also figure S2 and S3). Feed demand for livestock production results from animal-specific feed baskets (Weindl *et al* 2010, 2015). Socio-economic parameters, such as the demand, are exogenously fed into the model at the level of ten geo-economic world regions.

The model considers the production of 17 different crop groups and 5 livestock commodities. Bioenergy production was not included in this study. Potential crop yields, carbon densities and water availabilities are derived by the Dynamic Global Vegetation Model LPJmL (Bondeau *et al* 2007, Fader *et al* 2010, Waha *et al* 2012, Müller and Robertson 2014) on a spatial resolution of 0.5°. For the starting year of the model (1995) crop yields were calibrated to match attained country yield levels and regional production areas reported by FAOSTAT. For an efficient, non-linear modelling under computational constraints, spatial input data were aggregated to 600 clusters with similar crop yields, hydrological conditions and market access (Dietrich *et al* 2013a).

In the model there are several options to respond to future changes in demand or other pressures on the land-use system, such as afforestation. The land-use pattern can react flexibly so that one land-use class can be extended at the expense of others, e.g. cropland can be expanded onto former pasture areas, or afforestation might take place on present-day croplands. The model can also reallocate production to locations that are more productive, domestically within a region or via international trade. Another option implemented is the use of irrigation. Finally, agricultural production can be intensified by endogenous investment decisions in yield-increasing technological change.

Agricultural production and all options to increase production are associated with costs. Factor costs account for costs related to capital, labour and fertilizer use and were derived from the GTAP database (Narayanan and Walmsley 2008). The change from one land-use class to another is subject to regionally differing land conversion costs (Schmitz 2012). Yield increases induced by technological change are endogenous in MAgPIE and are connected to additional investment costs for Research & Development (R&D). These costs were derived through a regression between historical investments and observed yield increases (Dietrich *et al* 2013b). An investment horizon of 30 years and a discount rate of 7% are assumed for all investment decisions. Starting from the present distribution of areas equipped for irrigation (Siebert *et al* 2007), the model can increase irrigated areas at investment costs for the creation of the infrastructure and costs for operation and maintenance (Bonsch *et al* 2014). The cost effectiveness of production is also influenced by intraregional transport costs which make production at locations far from markets more expensive.

Food commodities can be traded between the world regions. Two trade pools are implemented in the model. Within the first trade pool, trade flows are fixed to fulfil regional, historically observed self-sufficiency rates calculated from FAOSTAT (2010). For the following time steps, the influence of this first trade pool is reduced depending on the scenario, and food commodities are to a larger share traded according to regional comparative advantages (Schmitz *et al* 2012) (figure S4).

Afforestation and avoided deforestation are incentivized by a price on CO_2 emissions from the land system. While the CO_2 price renders deforestation and the conversion of pasture to cropland more costly, carbon dioxide removal through afforestation is rewarded and lowers the costs in the objective function of the model. Afforestation is implemented as induced regrowth of natural vegetation. Carbon accumulation in living biomass follows sigmoidal tree growth curves where the upper limit is defined by carbon densities from the LPJmL model. Soil and litter carbon densities are assumed to increase linearly over 20 years, starting from the weighted average carbon density of cropland and pasture (Humpenöder *et al* 2014, 2015). For this study we assumed a CO_2 price that starts at 30 US$ per tonne of CO_2 in 2020 and increases by 5% each year (similar to Calvin *et al* (2012) and Kriegler *et al* (2013)).

Scenarios

Afforestation is considered to be most effective in the tropical zone because the combined effect of carbon sequestration and albedo change are assumed to lead to a net cooling, while for the boreal and temperate zones the effect is presumably much lower. To assess the food price effects of afforestation under differing levels of ability to decrease global temperatures, we considered three scenarios where afforestation was limited to certain latitudinal zones. Within these areas the decision to afforest was based on its cost-effectiveness under a CO_2 price on land-use emissions. The effect of albedo was not included directly in the model, but scenarios with different influence on albedo-induced radiative forcing were assessed. In the first scenario afforestation was not restricted at all (*unrestricted aff*), in the second not allowed in the boreal zone north of 50°N (*no boreal aff*), and finally it was limited to the tropical zone between 20°S and 20°N (*only tropical aff*). The definition of tropical and boreal zones thereby follows Bala *et al* (2007). These afforestation scenarios were compared to a scenario of *avoided deforestation*, where terrestrial CO_2 emissions were also priced but no afforestation was considered, and to a *business as usual* (*BAU*) case without any emissions pricing (see also table 1).

While limiting large-scale afforestation to the tropics seems plausible from a climate mitigation perspective, it could still result in severe food price hikes in tropical regions. Enhanced international trade of agricultural commodities could be one option to buffer these price increases in tropical regions. For the *only tropical aff* case we therefore assessed how more liberalized trade influenced food prices (*only tropical aff tradelib*). In this scenario, trade departed more quickly from historical agricultural trade patterns towards more international trade based on comparative advantages. While in our default setting the influence of historical trade patterns decreased by 0.5% per year, in this scenario it was reduced by 1% per year (see also figure S4).

For these scenarios we calculated Laspeyres food price indices that comprise vegetable and livestock products. The Laspeyres formula weights prices according to base year quantities and is also the common approach used, for instance, by The World Bank (2015) to calculate its consumer price index. Food prices derived from MAgPIE reflect the marginal costs of food production (shadow prices), i.e. the costs that would arise for the production of one additional commodity unit. They are formed as a consequence of altered demand and production costs and therefore show the relative long-term commodity price development. Food prices in the BAU scenario are driven by the increasing demand for food from a growing and wealthier population. In the *avoided deforestation* scenario food prices additionally reflect the pricing of land-use-change emissions, the thus reduced attractiveness to reduce the area of forest or convert pastures to cropland, and the increased need to invest into yield-increasing technology. Food prices in the afforestation scenarios are the result of all these factors and an additional reward on carbon uptake through afforestation which leads to decreasing agricultural areas.

Results

Land demand and required technological change

The growing demand for food (figure S2) leads to an expansion of croplands in the *BAU* scenario. Globally, cropland area increases by 360 million hectares (Mha) until 2100, leading to a reduction of the area of pasture by 275 Mha and of forests and other natural vegetation by about 85 Mha. The introduction of a price on CO_2 emissions from land-use change stops the net conversion of forest to agricultural areas on a global level. In the *avoided deforestation* scenario, cropland expands by 77 Mha, with most of the change happening in Africa (40 Mha) at the expense of pasture (17 Mha) and forest (13 Mha), while in Europe there is some regrowth of forests (14 Mha).

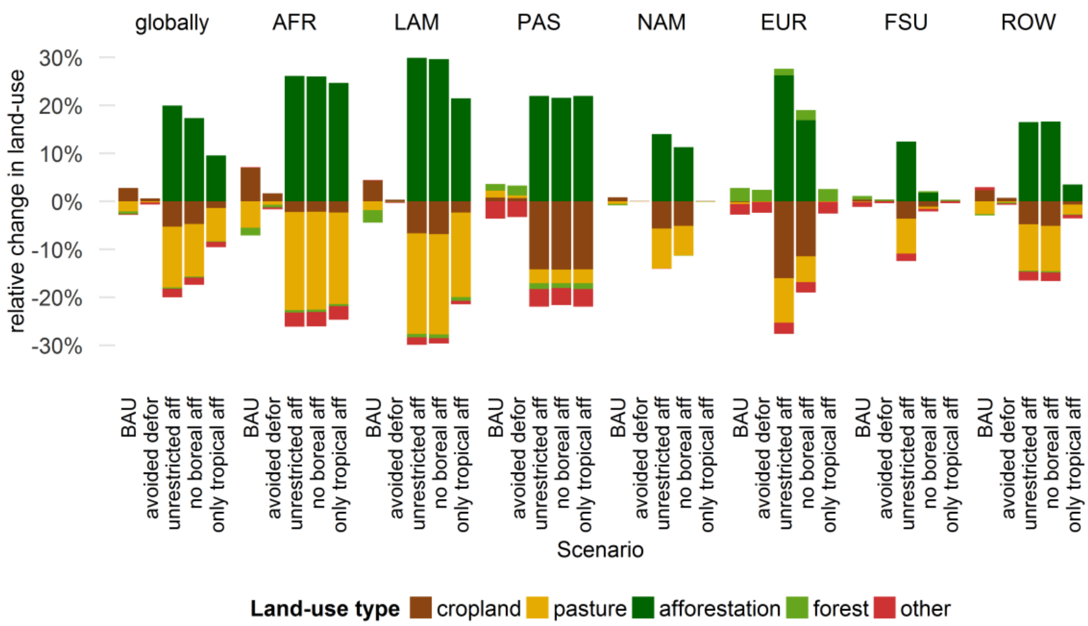

Figure 1 Change in land-use between 2010 and 2100 relative to total areas of the world or the regions (%). Positive values represent a net expansion of the land-use class, negative values a reduction. The land-use type "other" refers to other natural vegetation not classified as forest.

AFR: Sub-Saharan Africa, LAM: Latin America, PAS: Pacific Asia, NAM: North America, EUR: Europe, FSU: Former Soviet Union, ROW: Rest of the World (four remaining model regions aggregated)

In the afforestation scenarios the CO_2 price provides an incentive for afforestation so that forest area increases substantially in all regions where this option was given considering the latitudinal restrictions. Under *unrestricted afforestation*, more than 2500 million ha are newly afforested globally between 2010 and 2100, which is equivalent to an increase of global forest area by more than 60%. The largest areas of afforestation in absolute terms are in Africa (630 Mha) and Latin America (600 Mha), but afforestation is also substantial in all other regions when compared to their total land areas (figure 1). While in most regions afforestation leads mainly to a reduction in pasture, in Europe and Pacific Asia more croplands are converted to grow forests.

The restriction to *no boreal* afforestation reduces the afforested area by about 13% globally, but hardly changes the amount of land conversion in tropical regions. In the *only tropical* afforestation scenario, in contrast, the area of forest establishment is cut by half (table 1). While it remains at comparable levels in the tropical regions Africa and Pacific Asia it is lower in Latin America (435 Mha), because areas in the south (<20°S) were not considered for afforestation (figure S9).

Table 1 Scenario description and resulting afforested area, cumulative land-use emissions, food prices indices and technological change rates. Reference year for the figures is 2010.

Scenario	Afforestation	CO_2 price	Afforested area [Mha]		Cumulative emissions [Gt CO_2]		Food price index (2010 = 100)		Average annual yield-increasing technological change rate	
			2050	2100	2050	2100	2050	2100	2050	2100
BAU	No	No	0	0	88	91	103	92	0.76%	0.44%
avoided defor	No	Yes	0	0	8	2	128	95	1.09%	0.61%
unrestricted aff	allowed globally	Yes	1614	2577	-356	-860	186	442	1.66%	1.34%
no boreal aff	allowed <50°N	Yes	1351	2240	-330	-791	180	402	1.60%	1.29%
only tropical aff	allowed 20°S-20°N	Yes	921	1235	-266	-525	152	138	1.38%	0.81%

While in the *BAU* scenario investments into yield-increasing R&D are rather modest, the introduction of a price on CO_2 emissions prevents further agricultural expansion and necessitates higher yields in the *avoided deforestation* scenario. In the afforestation scenarios, pasture and cropland area decrease globally, which results in even more substantial yield increases needed to fulfil food demands (table 1). Throughout the afforestation scenarios, the highest rates of yield-increasing technological change are seen in 2020, when the pricing policy on land-use emissions is implemented. These rates are, especially in the tropical regions, substantially higher than those observed in the recent past (Fischer *et al* 2014). Until the end of the century average annual technological change rates range between 0.44% in *BAU* and 1.34% in the *unrestricted* afforestation scenario. Large regional differences are observed, with yields being about 5.5 times as high in Africa at the end of the century in the *unrestricted* case compared to 2010, but less than double within Europe in the same scenario (see figure S8 for regional yield development).

Carbon sequestration

Afforestation leads to considerable carbon sequestration. While in the *BAU* case more than 90 Gt of CO_2 are released as a result of land-use change, up to 860 Gt CO_2 are sequestered in the case of *unrestricted* afforestation between 2010 and 2100. The pricing of CO_2

emissions from land-use change in the *avoided deforestation* scenario results in no net release of carbon from the land-use system.

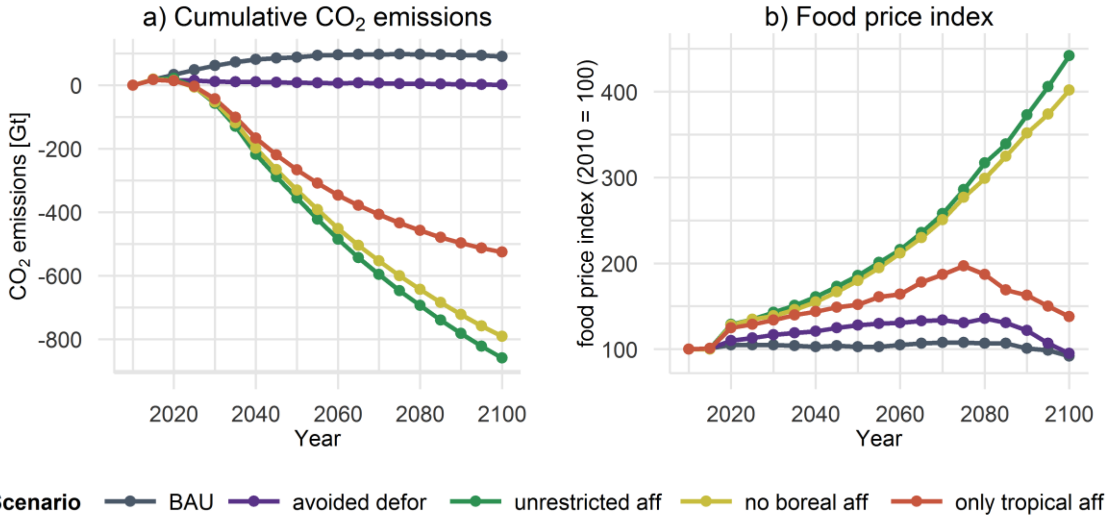

Figure 2 Cumulative emissions and food prices. a) Cumulative CO_2 emissions from land-use change and afforestation from 2010 until 2100. b) Laspeyres food price index for crop and livestock commodities (2010 = 100).

Restricting afforestation to non-boreal and tropical regions reduces the area and therefore the amount of carbon sequestered (figure 2a). For the *no boreal* scenario carbon removal is 8%, and the afforested area about 13% lower globally compared to the *unrestricted* scenario. In the *only tropical* afforestation scenario, terrestrial carbon uptake is about 40% lower than in the *unrestricted* scenario, while afforestation area is reduced by about 50%. The stronger reduction of afforestation area relative to CDR is as a result of higher carbon accumulation rates in temperate and tropical forests compared to boreal regions.

Food price effects

The increasing food demand from a growing population with an increased per capita demand for meat products does not lead to very significant changes in food prices. In the *BAU* scenario, without any pricing of emissions from the land-use system, food prices are projected to stay rather constant, or to decrease slightly to about 10% lower than in 2010 (figure 2b), caused by a decline in demand towards the end of the century (figure S2). The exponentially increasing CO_2 price on land-use-changes emission in the *avoided deforestation* scenario prevents the conversion of pasture and forest to cropland. Increasing land scarcity and the necessary investment costs for research and development increase prices at maximum by about 40% on global average in this case.

Afforestation leads to competition for land between carbon sequestration and agricultural production and results in substantial food price increases. Under *unrestricted* afforestation food prices increase by about 80% up to 2050 and are on average more than four times higher in 2100 than in 2010. Excluding boreal regions from afforestation reduces this effect only by about 9% in 2100. However, when afforestation is limited to the zone of highest cooling effectiveness – the tropics – the food price impact is significantly reduced. In the *only tropical* afforestation scenario, food prices peak in 2075 having increased by about

100%, followed by a decline in prices due to decreasing demand for food at times of high agricultural yields and a slowdown of forest expansion. Especially in the *unrestricted* and *no boreal* scenarios, the additional land-use competition through afforestation influences prices much more strongly than the mere effect of emission pricing in the *avoided deforestation* scenario. Food prices are also sensitive to the CO_2 price. Lower CO_2 prices lead to lower carbon sequestration, but also reduce food prices (figure S9).

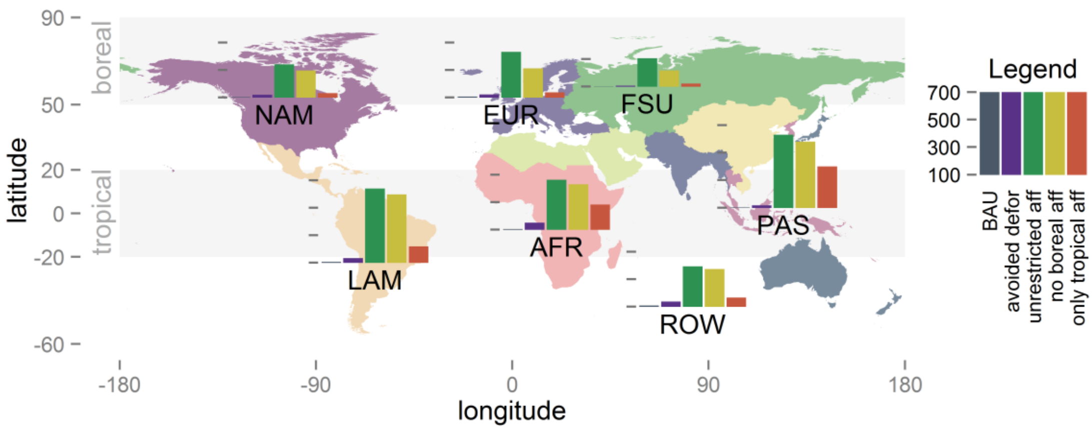

Figure 3 Regional food price indices at maximum over the course of the century (2010 = 100) for 6 out 10 modelled regions

AFR: Sub-Saharan Africa, LAM: Latin America, PAS: Pacific Asia, NAM: North America, EUR: Europe, FSU: Former Soviet Union, ROW: Rest of the World (four remaining model regions aggregated)

Food prices in different regions are affected differently by the modelled afforestation scenarios (figure 3). *Unrestricted* afforestation leads to the highest prices of all scenarios over the century within all regions, with the highest values occurring in Pacific Asia (PAS: 630) and Latin America (LAM: 640). In the Former Soviet Union (FSU) the increase is lowest, with prices three times higher in 2100.

Excluding the boreal zone from afforestation leads to lower food commodity prices than *unrestricted* afforestation, especially in regions that are partly in the boreal zone. In Europe (EUR) and FSU estimated food prices in 2100 are then about 30% lower. FSU turns into a net exporter of crops, EUR into a net exporter of livestock products towards the end of the century (figures S6 and S7), which also influences food prices in other regions. In Africa (AFR) and LAM, prices are 7% lower in the *no boreal* than in the *unrestricted* scenario in 2100, even though afforested area differs by less than 1% (see also figure 1).

Limiting afforestation to the tropical zone results in a food price index much closer to the *BAU* scenario, and much lower than for *unrestricted* and *no boreal* afforestation, but in tropical regions the price increases are still substantial. In Pacific Asia the food price index is highest in 2100 with a value of 400, while in Latin America the maximum index level of 219 is reached in 2070. The influence on temperate and boreal regions is much lower. In EUR, NAM and FSU the price indices are at maximum increased by 35% to 40% compared to the BAU case. In this scenario, food price increases are in all regions lower than the assumed increase in GDP (figure S5).

The effect of global trade under tropical afforestation

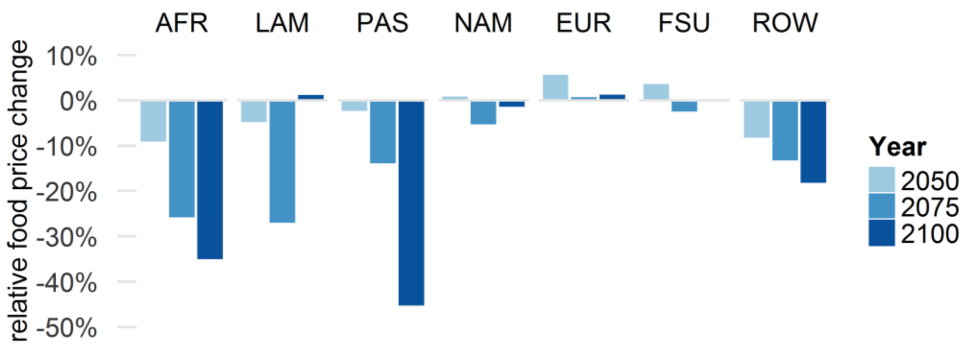

Figure 4 Influence of liberalized trade on regional food prices in 2050, 2075 and 2100. Comparison between the only tropical aff and the only tropical aff tradelib scenario. 2075 is the year in which food prices were highest globally.

More liberalized trade helps to buffer food price increases driven by tropical afforestation. We compared the food prices of the *only tropical* scenario to a scenario where the deviation from historical trade patterns was twice as fast (figure 4). In this *only tropical aff tradelib* scenario the overall, interregional trade volume increases faster (see also figures S6 and S7). Latin America turns from an exporter of food commodities into a net importer towards the end of the century. Africa further increases its imports of livestock products, which are mostly supplied by North America. In 2075, the year in which prices are highest globally, food prices are reduced by more than 25% in Latin America and Africa (figure 4). In Pacific Asia, where food prices are highest in 2100, the price index changes from 400 to 219. Subsequent price increases in Europe are negligible. While trade liberalization has a strong influence on prices, it does not decrease afforested area (1275Mha) or the sequestered amount of carbon (552 Gt).

Discussion

Afforestation impacts food prices

Our results show that large-scale afforestation can lead to significant carbon sequestration in the land-use sector, but can also lead to strongly rising food prices. In our study, these food price increases were the consequence of a large-scale transformation of the land-use sector, where food has to be produced on a much smaller overall agricultural area. In the scenario of *unrestricted* afforestation, cropland area is reduced by almost half to a global value of about 800 Mha in 2100, and pasture shrinks by more than 50% to about 1465 Mha, values that were last observed at around the year 1900 (Klein Goldewijk *et al* 2011). This decline in agricultural areas is enabled by significant investments into yield-increasing technological change and comes along with a pronounced increase in food prices. Avoided deforestation alone does not drastically spike food prices, which is in line with an earlier study by Schneider *et al* (2011). The finding that afforestation drives up food prices is also the result of a previous study by Calvin *et al* (2014), in which afforestation was also incentivized by a price on emissions from land-use, and resulted in increasing wheat prices. In contrast to this study, we report a combined food price index for meat and food-crop products for different afforestation scenarios. We also compare food prices under afforestation to a scenario where emissions from land use are priced, which leads to *avoided deforestation*. This comparison shows that most of the price increase can in fact be attributed to afforestation, while the emissions pricing alone is of lesser importance.

Limiting afforestation to the tropics – where it is most effective in decreasing global temperatures –substantially reduces the impact on food prices. Earlier studies with earth system models showed that afforestation in the tropics, through the combined effect of carbon sequestration and albedo change, leads to a net cooling, while planting trees in the boreal zone might even increase global temperatures (Bala *et al* 2007, Bathiany *et al* 2010, Arora and Montenegro 2011). While this simplified, latitudinal dependence seems to hold true in general, exceptions are possible under specific site conditions. Since historical boreal and tropical deforestation took place on the most productive lands with above-average carbon stocks and below-average snow cover, a reforestation of some boreal areas might also decrease global temperatures (Pongratz *et al* 2011). And an afforestation of tropical and subtropical desert areas could result in net warming because of the prevalence of the albedo effect (Keller *et al* 2014). Desert areas with high albedo, however, were not considered for afforestation in our study. Rather, afforestation was restricted to agricultural areas in certain latitudinal zones, excluding boreal and temperate zones where afforestation might not show a global cooling effect. Integrating the albedo-induced radiative forcing effect of afforestation directly in the model, as has been done by Jones *et al* (2015), should be considered for future model applications.

While limiting afforestation to the tropics reduced food prices globally, food price indices remained higher in tropical regions. These increased price levels in the tropics could be buffered by a more liberalized trade policy, with an ensuing shift of agricultural production to non-tropical regions. However, this interregional reallocation would also increase the import dependency of some tropical regions and might hamper the development of the agricultural sector within these regions.

Afforestation requires the reversal of deforestation and R&D spending trends

Before afforestation can be considered as a serious means to mitigate climate change, deforestation has to come to an end. In our study this happened as soon as there was a price on CO_2 emissions from deforestation. At the moment, however, no such policy is in place on a global level and much of the carbon stored in tropical forests is released into the atmosphere. Gross carbon emissions from tropical regions were estimated to be around 0.81 GtC yr^{-1} between 2000 and 2005 (Harris *et al* 2012), with yearly emissions of deforestation from the Amazon basin alone accounting for 0.18 GtC between 2000 and 2010 (Song *et al* 2015). The current trend is opposite to what we described in our *only tropical* afforestation scenario. Between 1993 and 2012 tropical forests lost aboveground biomass carbon (−0.21 GtC yr^{-1}), while boreal and temperate forests gained it by about the same amount (+0.18 GtC yr^{-1}) (Liu *et al* 2015). However, Brazil – the country with the greatest absolute forest area reduction – has recently reduced its deforestation curve through conservation policies and stricter law enforcement on the ground (FAO 2015b, Assunção *et al* 2015, Tollefson 2015). China has initiated a large afforestation programme, with plans to increase afforested area by 40 Mha by 2020, a measure which was found not only to sequester carbon but also to decrease local land-surface temperatures (Peng *et al* 2014). And in December 2015, ten African countries launched AFR100, and initiative to restore 100 Mha of degraded and deforested land by 2030 – partly as a climate change mitigation measure (WRI 2015). These developments are just few of many that indicate that global afforestation efforts now have better prospects for success.

Continuous yield increases and substantial investment into yield-increasing R&D would be needed to fulfil the food demands of a growing population, especially when agriculture competes with afforestation. The high price on CO_2 emissions, and hence the strong incentive to free up agricultural land for afforestation, initiates continuous yield-increasing technological change in our study, with values well above those observed historically. In contrast to other partial equilibrium land-use models (e.g. GLOBIOM: Kraxner *et al* (2013), GCAM: Calvin *et al* (2014)), technological change is endogenously derived within MAgPIE (Dietrich *et al* 2013b, von Lampe *et al* 2014), and yields tend to increase stronger in response to additional pressures on the land-use system (Nelson *et al* 2014, Lotze-Campen *et al* 2014, Delincé *et al* 2015). During recent decades, yields of main staple crops increased linearly at average rates of 1% (wheat, rice, soybean) and 1.5% (maize), while the relative annual rate of increase constantly dropped (Fischer *et al* 2014). Increased investment into R&D would be needed to make afforestation a realistic option, but when research spending increased in recent years this was largely driven by the development in single countries like China and India. Almost every third OECD country actually had a negative trend in public agricultural R&D spending. And in the developing world, especially in Sub-Saharan Africa, where in our afforestation scenarios yields more than tripled between 2010 and 2100, public spending on agricultural R&D amounted to only about 1.6 billion US$ or 5% of global agricultural R&D spending in 2008, and almost half the African countries had a negative trend in their budgets (Beintema *et al* 2012). This trend of low R&D spending would certainly have to turn around in order to achieve the yields projected in our model.

The yield increases triggered by afforestation could also alter agricultural N_2O and CH_4 emissions, a dynamic that was not in the focus of this study. Intensification could both increase or decrease N_2O emissions from soils, depending on whether intensification is reached through higher inputs (e.g. fertilizer) or better agronomic practices (Lassaletta *et al* 2014, Bodirsky and Müller 2014). CH_4 emissions from the livestock sector would likely be decreased by intensification due to a more efficient feed conversion (Herrero *et al* 2013).

Results set in context

The food-price increases presented in this study have to be seen in the context of a general increase in wealth. For this study we assumed the GDP development of the SSP2 scenario (Dellink *et al* 2015), which is steadily increasing for all model regions, and is also the basis for the increased per capita demand for food products. In most regions the rates of GDP increase are higher or in the same range as the price increases due to afforestation, so that share of expenditure for food would stay constant or decrease for a representative agent (see figure S5). Still, increases in wealth would not necessarily be distributed evenly among the population, so that the change in prices reported here could still have drastic impacts on the poorer parts of society. This is especially true for people whose share of expenditure on food is currently quite high, such as the poorest people in some African and Asian countries who currently expend above 70% of their available income on food (FAO 2015a).

A number of factors influence the formation of food prices, and our study focuses on the more long-term drivers. In the coming decades, a growing global population is expected to increase the demand for food, in particular for livestock products (Alexandratos and Bruinsma 2012, Bodirsky *et al* 2015). This, together with a likely elevated demand for bioenergy, will increase the total demand for agricultural products. These long-term trends are overlain by a number of more short-term factors affecting prices, such as weather variability, financial speculation or restrictive export policies in response to increasing prices (Mueller *et al* 2011). Lagi *et al* (2015), for instance, were able to replicate the FAO food price index between 2004 and 2012 with a dynamic model, where the underlying upward trend was due to an increasing demand for ethanol production, while the short-term peaks were caused by speculation. Our model is designed to capture the medium-term to long-term drivers of food price formation, and reveals the relative difference between afforestation scenarios and a world without forest-based climate mitigation. It does not consider specific policies and drivers on local or short time scale.

Food demand was provided exogenously to the model as a function of per capita income and population. Since price hikes in the afforestation scenarios were quite high with respect to the BAU case, it could be expected that the consumption of agricultural products declines, in spite of relatively low demand-to-price elasticises of food products, especially in high income countries (Muhammad *et al* 2011, Hertel 2011). Also for this reason, MAgPIE represented the upper range of food price estimates when climate change effects were assessed in a model intercomparison (Nelson *et al* 2014). However, we also assumed that currently developing regions become relatively wealthy towards the end of the century when food prices are projected to be at the highest level, which would result in lower shares of income expenditure on food and low demand elasticities.

Afforestation at the scale as described in this study would imply macro-economic effects that should be subject to further research, for instance within a general equilibrium framework. The MAgPIE model is a partial equilibrium model of the agricultural sector, impacts of afforestation on other sectors of the economy such as labour, capital and carbon markets were therefore not part of this study. We would expect that increasing food prices also increase the income of net food sellers, and reduce the incomes of net buyers as non-food expenditures are reduced, which could in consequence change the demand for food (Dorward 2012). Afforestation might also create new jobs in the short term for the planting of trees, but these jobs would vanish once the forests are established. Rent-seeking behaviour and opportunities to invest in land under a policy rewarding carbon removal could substantially shift production input factors from other sectors. Furthermore, our analysis of trade was focused on the agricultural sector. For the only tropical afforestation scenario we assessed how trade liberalization would influence regional food prices. We have, for instance, not considered how the consequential change in trade flows (e.g. increased imports of livestock products to Africa) would have to be compensated by trade flows in other sectors to avoid trade deficits, or how trade liberalization would affect economies in general. Finally, the creation of an international market for carbon credits could create a substantial flow of money from CO_2-emitting countries to those actively sequestering carbon through afforestation. These revenues could be used to finance, among other things, the import of food.

Conclusions

In order to mitigate climate change, land-based carbon dioxide removal will likely have to play an important role. Afforestation has been identified as a comparatively low-cost option to sequester carbon, but side-effects of afforestation at large-scale were so far not much in the focus. Afforestation will, if it competes with food production for the same areas, lead to an increase in food prices. Moreover, as previous research has shown, afforestation in high latitudes will likely only have a small cooling effect on the global average temperature, or could even increase it, because of the counteracting albedo warming effect.

Our study confirms that afforestation offers a high potential for carbon dioxide removal, and more than 860 Gt of CO_2 are sequestered in our unrestricted afforestation scenario up to the end of the century. However, we also find that this afforestation leads to a more than fourfold increase in food prices by 2100. When afforestation is restricted to the tropics – and thus the albedo warming effect avoided – still substantial carbon sequestration can be achieved. This, at the same time, lowers global food prices substantially which nevertheless remain increased in tropical regions compared to a world without large-scale forest expansion. Our study suggests that a liberalization of agricultural trade could further dampen the remaining price increases in tropical regions.

By sequestering carbon though afforestation, tropical regions would offer a valuable service for the benefit of the whole world. An international carbon market for carbon credits could be the source of monetary flows to those tropical countries undertaking afforestation and could compensate for some of the disadvantages coming along with it. Thoughtfully designed policies would have to avoid that established forests are cut down again and release the carbon stored. The raised money should also be used for investments into agricultural R&D, to achieve necessary rates of yield increase. And lastly, policies should be designed in a way which assures that not only land-owner profit, but revenues are also distributed to those people affected most by the food price increases.

We conclude from our study that afforestation should not be seen as the silver bullet of climate change mitigation, but set in the right context and done at the right location it can well be a complement to other mitigation options.

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Supporting Information

Model description

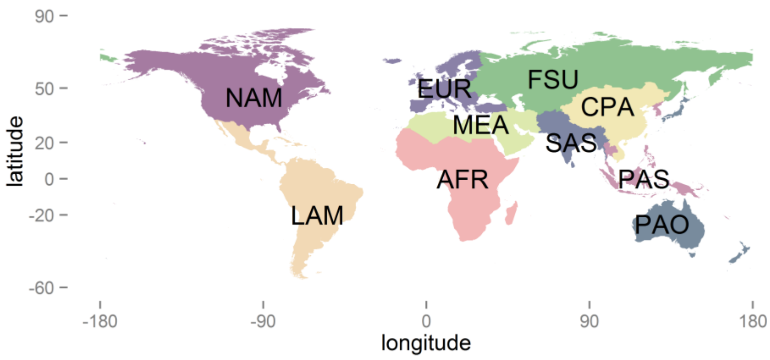

Figure S1 MAgPIE model regions AFR: Sub-Saharan Africa, CPA: Centrally planned Asia including China, EUR: Europe including Turkey, FSU: States of the former Soviet Union, LAM: Latin America, MEA: Middle East and North Africa, NAM: North America, PAO: Pacific OECD including Japan, Australia, New Zealand, PAS: Pacific Asia, SAS: South Asia including India.

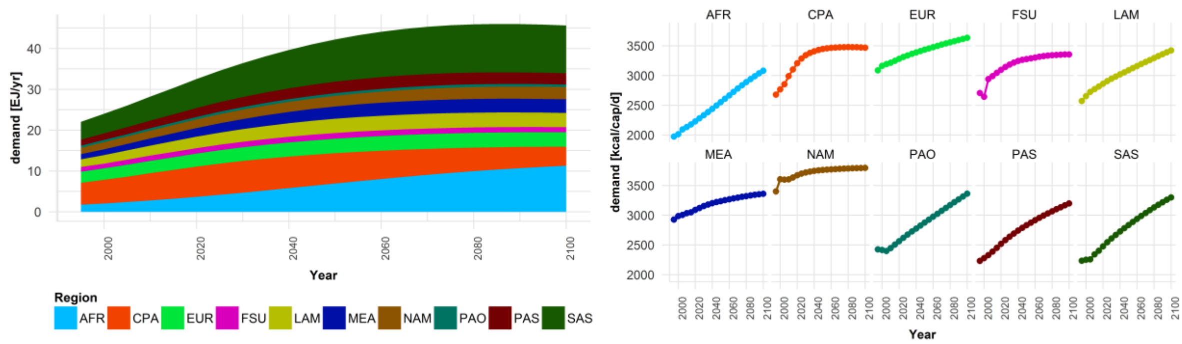

Figure S2 Regional food demand in EJ and in kcal/cap/d. Identical for all scenarios. Per capita demand is driven by the assumed increase in wealth, demand in EJ by per capita demand and population.

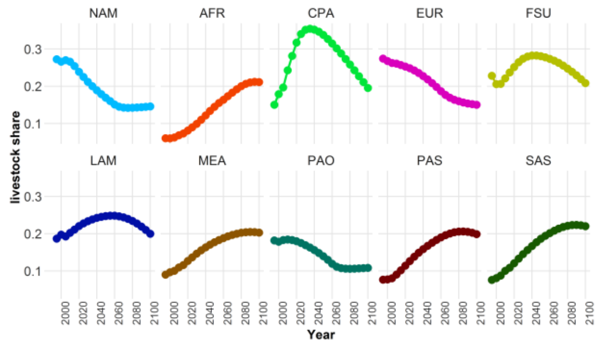

Figure S3 Regional shares of livestock in total food consumption. Identical for all scenarios.

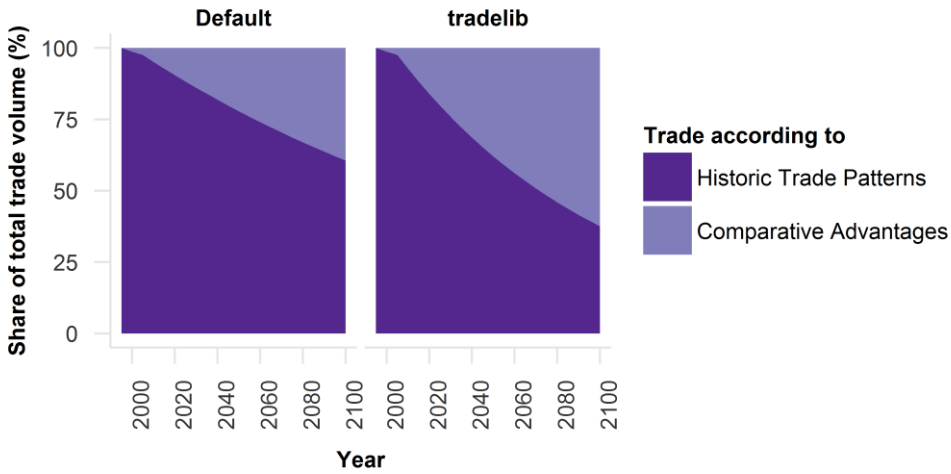

Figure S4 Share in trade pools. Two trade pools are considered in the model. Trade according to the first pool follows historic trade patterns, where regions have to fulfil historically observed self-sufficiencies in food provision. In the second trade pool trade is completely according to regional comparative advantages. The influence of historic trade patterns is reduced over time (Schmitz et al., 2012). The default trade liberalization was assumed for all scenarios besides the *only tropical tradelib* afforestation where we assumed faster trade liberalization.

Results

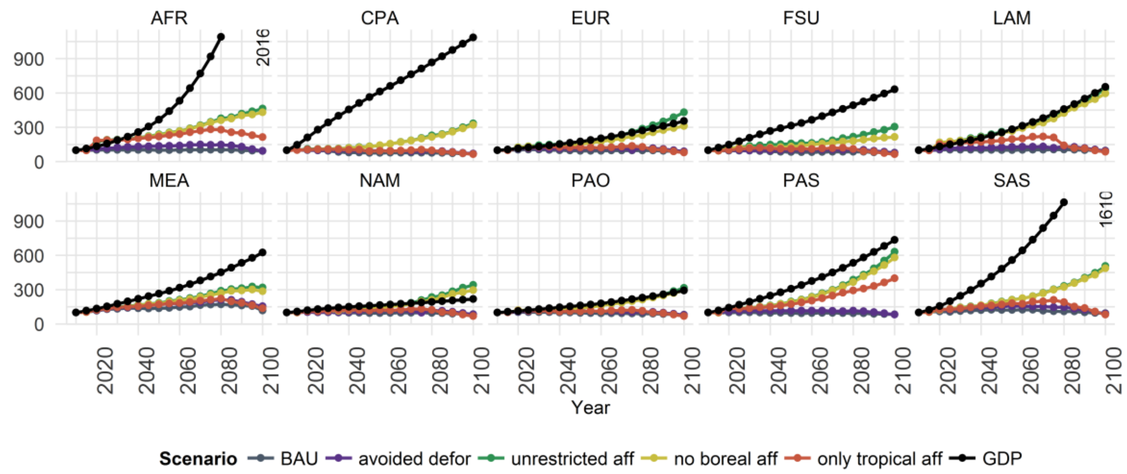

Figure S5 Regional food price index (coloured lines) and normalized GDP (MER) (black line). Both indexed to a value of 100 in 2010. Cut off at an index value of 1100 (in AFR and SAS); GDP index in 2100 then given in numbers.

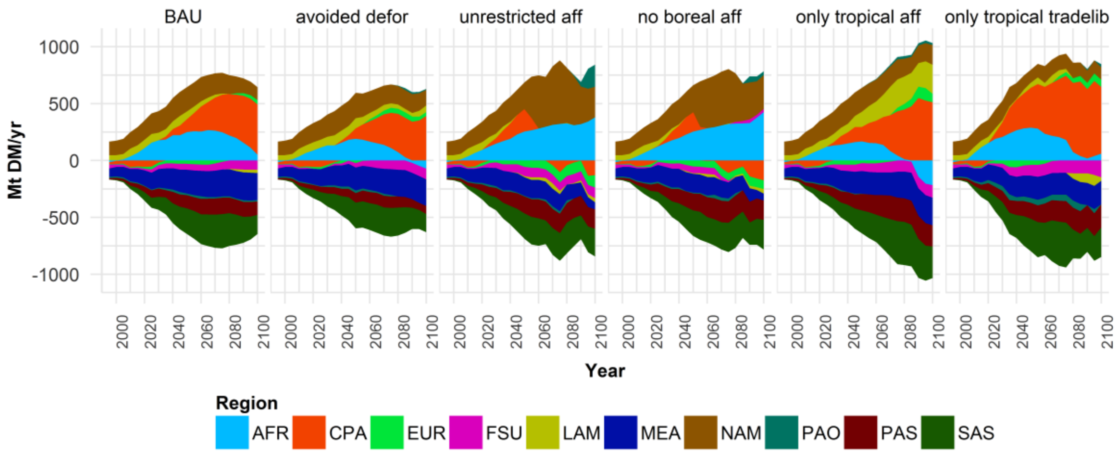

Figure S6 Net trade in crops. Values > 0: Exports, Values < 0: Imports.

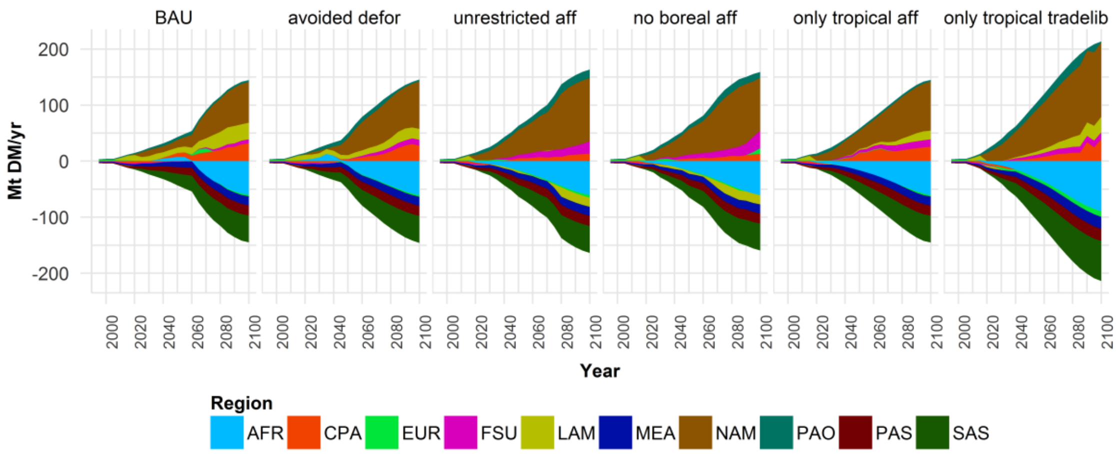

Figure S7 Net trade in livestock products. Values > 0: Exports, Values < 0: Imports.

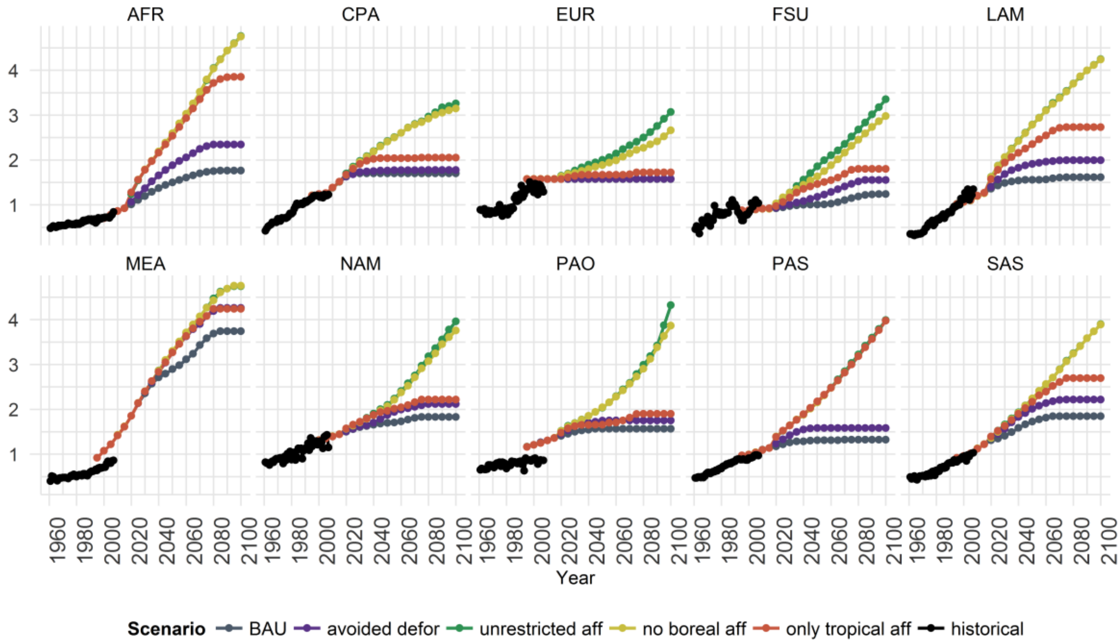

Figure S8 Land use intensity (tau). Values represent the regional agricultural land-use intensity and show the assumed crop yield development through investments into yield-increasing R&D. They are indexed to a global average of 1 in 1995. Historical values are from Dietrich et al. (2012), where also a more detailed description of the approach can be found.

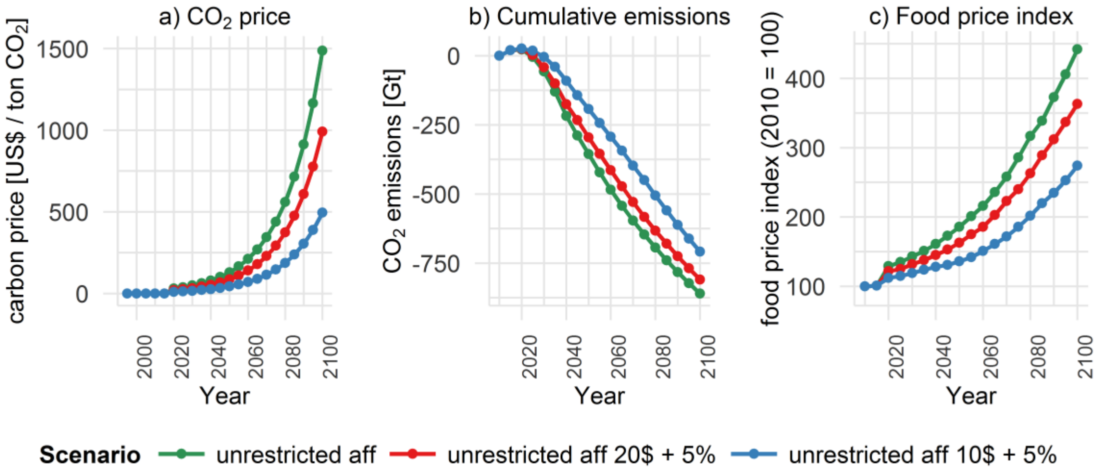

Figure S9 Carbon price sensitivity of the food price index. While we assumed a price on CO_2 emissions starting at 30 US$ per ton of CO_2 in 2020 and increasing by 5% per year, we here also show the food price response for scenarios of unrestricted afforestation with CO_2 prices starting at 10$ and 20$.

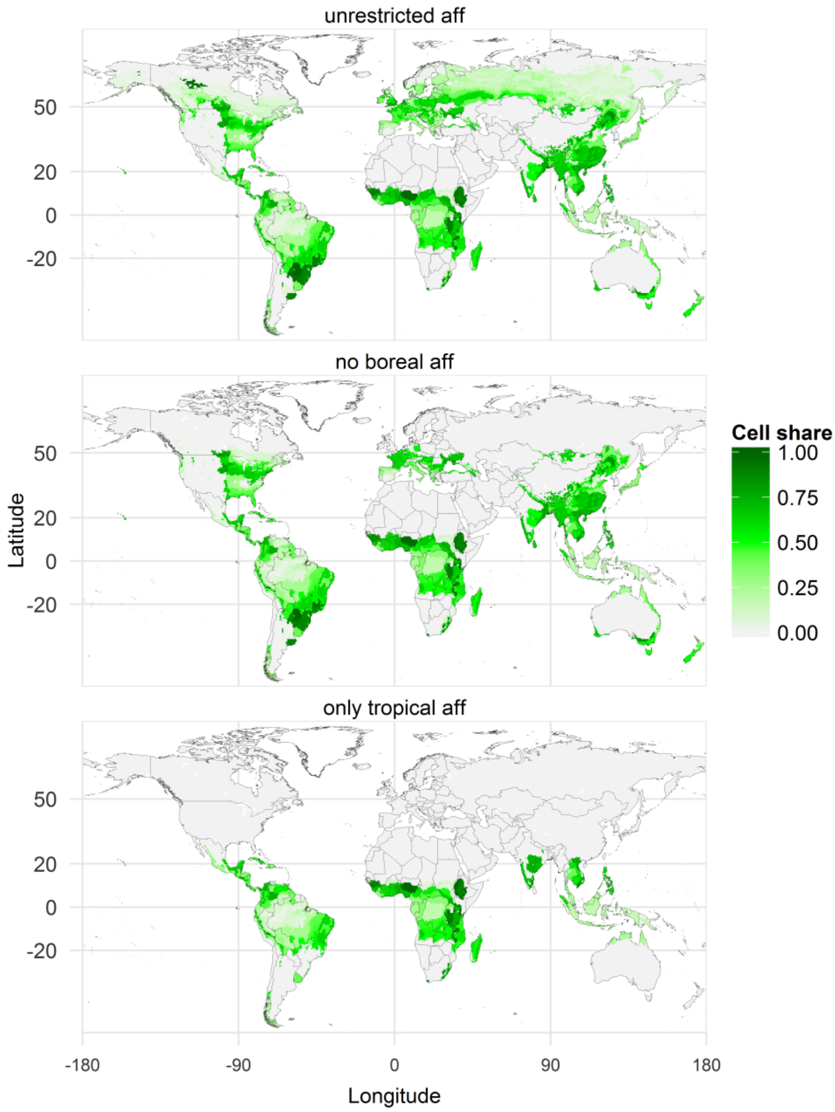

Figure S10 Maps of afforested area in the *unrestricted*, *no boreal*, and *only tropical* afforestation scenarios. Depicted are the shares of grid cells that were newly afforested between 2010 and 2100.

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V Key results

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1. Conversion of forests and other natural habitats likely to continue

This thesis research has indicated that under business as usual, it is very likely that forests will continuously be lost as a result of the growing demand for food and other agricultural commodities. Chapter II has assessed tropical forest losses as a consequence of trade liberalisation and under different policy options. This study foresees substantial cropland expansion and deforestation. The reference case of the study predicts that deforestation would occur on 140 Mha in Latin America, 64 Mha in Sub-Saharan Africa and 24 Mha in Pacific Asia between 2010 and 2050 without further trade liberalisation. Further trade liberalisation would boost deforestation, especially in Latin America, and increase deforested area to 180 Mha. Chapter III has assessed potential loss of forests and other natural vegetation, with a special focus on conservation of priority areas in Africa and Latin America. This study has also suggested substantial loss of natural vegetation until the middle of the century.

1.1. Land-use change is persistent threat to climate and biodiversity

Land-use change and forest loss that are associated with increasing demand can cause additional greenhouse gas emissions and pose a threat to valuable natural habitats. In the event of no additional pasture intensification and under further trade liberalisation, as assumed in Chapter II and one scenario of Chapter III, land-use change would lead to cumulative emissions of more than 100 Gt CO_2 by the middle of the century. This is a substantial share of the cumulative CO_2 emission budget of less than 1,000 Gt CO_2 that remains for the period between 2011 and 2100 to likely limit warming to 2°C (IPCC, 2014). Chapter III, which has more closely examined the spatial pattern of land-use change, has evidenced that the occurrence of land-use change is also projected for areas which are vital for biodiversity conservation. In the reference case of the study, approximately 23 Mha of natural vegetation would be lost within conservation international biodiversity hotspots (Mittermeier *et al.*, 2004) in Latin America, and about 14 Mha in Sub-Saharan Africa.

2. Policies offer potential to mitigate climate and conserve biodiversity

Several policies have been suggested whereby land-use could contribute to climate change mitigation and conserve biodiversity. This thesis has reviewed a selection of policies that are relevant for the development of land-use change on the global level. It has specifically assessed the consequences of expanding protected areas and pricing land-use change emissions in addition to considering the influence of technological change and the associated boosts in yields and productivity.

2.1. Protected areas could save forests and emissions

Protected areas significantly reduce land conversion (Andam *et al.*, 2008; Beresford *et al.*, 2013; Geldmann *et al.*, 2013), and this thesis underpins their importance for the conservation of intact forest ecosystems and their co-benefit in avoiding land-use change emissions. Chapter II has analysed a scenario in which the share of protection of intact and frontier forests increased over time. This reduced total deforestation in the tropics from

264 Mha in the no-policy case to 152 Mha. Furthermore, it concurrently decreased CO_2 emissions from land-use change by 40%. As Chapter III has demonstrated, protected areas are essential to stopping, or at least reducing, the loss of conservation priority areas. Land-use change in Latin America and Africa could cause the loss of large shares of biodiversity hotspots, especially if the growing demand for food is not accompanied by strong increases in livestock production intensity. The study also emphasises the importance of good planning for location. Designation of protected areas should be founded on where they are most needed and where there is a high number of endemic species which are also threatened by land-use change. Protection of frontier forests, for instance, has had only a mild effect on converted areas in Latin America, as these areas were hardly converted without protection. A protection of biodiversity hotspots, in contrast, has halted the loss of these areas but has also reduced overall land conversion in Latin America.

2.2. Pricing land-use change emission could reduce deforestation and incentivise afforestation

A price on land-use change emissions and a reward for carbon dioxide removal could render deforestation economically unattractive and induce afforestation. In this way, land use could shift from a net source of greenhouse gas emissions to a net sink. The study informing Chapter II has assessed the implications of two CO_2 price trajectories on tropical deforestation. Both scenarios – the case where the price reaches 12.5 USD in 2050 and the case where it exponentially increases to roughly 90 USD in 2050 – result in a significant decrease in deforestation, and therefore in land-use change emissions. Especially in the second case, hardly any deforestation occurs after 2020, when the price surpasses 10 USD. This reveals that avoided deforestation is a cheap mitigation option. In the study entitled 'Afforestation to mitigate climate change: impacts on food prices under consideration of albedo effects' (Chapter IV), which assumes the CO_2 price to start at 30 USD in 2020 and increase by 5% per year, land-use change emissions are completely avoided. In the other scenarios of this study, the CO_2 price additionally incentivised afforestation. As a result, almost 2,600 Mha would be afforested by the end of the century and would sequester about 860 Gt of CO_2. Even if afforestation were limited to the tropics, where afforestation has the highest cooling effect, more than 500 Gt of CDR could be achieved.

2.3. Necessity of agricultural intensification

Agricultural intensification will be necessary to fulfil the expected growth in food demand. The modelling of this thesis assumed a population growth and a higher per-capita consumption on average. A combination of agricultural area expansion and intensification could satisfy this consequential increase in demand. MAgPIE endogenously modelled investments into yield-increasing R&D, and all simulation strongly relied on this option. Scenarios with stronger emission reductions were associated with higher rates of crop intensification. For instance, the high CO_2 price scenario in Chapter II, which concluded in the lowest overall deforestation emissions, also resulted in one of the highest productivity increases. Chapter III assessed the sensitivity of grazing intensification, while cropland yields were consistent across scenarios. In Africa, where domestic demand is projected to more than triple by 2050, low rates of pasture intensification were associated with much higher losses of natural vegetation. Especially in cases involving additional pressures, such as afforestation as modelled in Chapter IV, high rates of agricultural intensification are needed.

2.4. Strong influence of agricultural trade

The pace of international trade liberalisation is a key determinant in the future of land-use. The assessment of Chapter II has indicated that a reduction of trade barriers can induce higher deforestation in tropical countries. Regions with relatively low production costs have a comparative advantage at producing and increase their production. However, this induces agricultural expansion at the expense of pristine natural forests. Trade liberalisation must therefore be accompanied by policies that conserve precious ecosystems.

The openness of regional agricultural markets also determines the effectiveness of domestic policies. Higher grazing productivity in Latin America, as in the analysis of Chapter III, hardly reveals any land saving. Instead, the increase in production was followed by an increase in exports and did not reduce the total agricultural area.

Agricultural trade liberalisation also influences food prices. Chapter IV has considered the consequences of strong climate change mitigation via afforestation, which would prompt food prices hikes due to competition with food production. When afforestation is restricted to tropical regions and does not affect food production in temperate regions, an increase in trade could buffer food price increases globally.

2.5. Land-based climate change mitigation could affect food prices

Global climate change poses several problems to society, and its mitigation might also have negative societal effects. Assessments of policies that aim to reduce emissions in the land-use sector should therefore take into account not only the effectiveness of the policy in reducing emissions but also its influence on society. This thesis emphasises that despite offering immense CDR potential, afforestation can have severe consequences for food prices and could reduce the self-sufficiency of regions in food production. The study of Chapter IV has concluded that large-scale afforestation of over 2,500 Mha following the introduction of a global price on land-use emissions could quadruple global food prices. Even though the price effect could be reduced by limiting afforestation to the tropics – where its cooling effect is highest – it could still substantially increase prices in those regions.

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VI Discussion & Outlook

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1. Results put in perspective and political implications

Through the Paris Agreement, governments have committed to limit global warming to 2°C compared to the pre-Industrial era and to strive to restrict it to 1.5°C. To fulfil these goals net anthropogenic emissions should amount to zero in the second half of the century (UNFCCC, 2015), and it is imperative to immediately reduce emissions in all sectors (Rogelj *et al.*, 2015; Millar *et al.*, 2017; Su *et al.*, 2017). Consequentially, the land-use sector is already crucial in the Intended Nationally Determined Contributions (INDCs) (Forsell *et al.*, 2016). This thesis underpins the immense importance and high potential of land use and agriculture in reducing emissions. The research of this thesis has also depicted scenarios which halt deforestation and phase out emissions related to land-use change without compromising the food production that is necessary to feed a growing population. Through afforestation efforts, land use could also contribute to CDR from the atmosphere. A downside, however, is that the increased pressure on the land system might lead to a surge in food prices.

1.1. Cessation of deforestation needed – policies at hand

Probably the first and most critical step in exploiting the mitigation potential of land use while reducing the decline in biodiversity is to stop the on-going deforestation in the tropics (Kindermann *et al.*, 2008). As this thesis has revealed, failure to implement adequate policies is likely to enable continued deforestation in response to heightened demand for food and an expansion of agricultural areas, which could thus threaten vital habitats and cause substantial greenhouse gas emissions. A crucial focus here is the assessment of two policies: an expansion of protected areas (Chapters II and III) and a price on carbon emissions from land-use change (Chapters II and IV). Both policies have proven to be fairly useful for reducing deforestation, emissions and areas that are important for biodiversity conservation. The effectiveness of protected areas at reducing the loss of natural vegetation, however, is critically dependent on their location, the timing, and the occurrence of leakage effects (Ewers & Rodrigues, 2008; Joppa & Pfaff, 2011; Pouzols *et al.*, 2014; Kehoe *et al.*, 2017). Of course, neither the effective expansion of protected areas nor the implementation of a carbon price is easily achieved, but it is ultimately mostly a question of political will rather than a lack of suitable policies to transform current deforestation into an expansion of global forest area.

1.2. Costs of mitigation

Climate change has already substantially impacted human well-being and ecosystems worldwide, and this trend will intensify with time (IPCC, 2014). The social costs of carbon – the economic costs of one ton of CO_2 equivalent (CO_2eq) emitted into the atmosphere – are substantial (van den Bergh & Botzen, 2014; Marten *et al.*, 2015; Moore & Diaz, 2015; Hsiang *et al.*, 2017; Nordhaus, 2017). Climate change mitigation, on the other hand, provides positive co-benefits, for instance regarding global air quality (Rao *et al.*, 2016). In addition, any delay of climate change mitigation will hinder the achievement of the 2°C target and increase its costs (den Elzen *et al.*, 2010; Jakob *et al.*, 2012; Rogelj *et al.*, 2013; Luderer *et al.*, 2016).

However, as this thesis has demonstrated, climate change mitigation in the land-use sector could also connect with adverse effects. Ambitious mitigation through afforestation could foster competition with food production and thereby increase food prices. Similar concerns have been raised for bioenergy production before (Ajanovic, 2011; Mueller *et al.*, 2011). A large-scale afforestation program could result in as much as 2,500 Mha of additional forests, which is more than the current area of arable land (FAOSTAT, 2016a), as well as drastic increases in food prices. Prospects for land-based CDR coincide with a likely heightened demand for agricultural products from an expanding population (Gerland *et al.*, 2014) with increased per-capita consumption (Kearney, 2010; Alexandratos & Bruinsma, 2012; Pradhan *et al.*, 2013). Despite the lack of exploitation of many possibilities to reduce food demand, such as a reduction of food waste (Bajželj *et al.*, 2014), higher reliance on irrigation (Jägermeyr *et al.*, 2016) or spatially optimised production (Koh *et al.*, 2013), an achievement of both sufficient agricultural production and land-based climate change mitigation will remain challenging. In order to minimise the social effects of afforestation, the process should be directed at areas with the highest effectiveness, namely the tropics. As Chapter IV has revealed, this could already substantially reduce the effect on food prices, as could more liberalized trade. Nevertheless, in order to avoid social disruption by afforestation, food price effects must be closely monitored and potentially subject to compensation, especially in tropical developing countries with high expenditure shares on food (FAO, 2015).

2. Uncertainties in modelling the future of land use

This thesis has largely relied on land-use modelling to assess the influence of policies on land use, food prices and biodiversity. This section discusses certain limitations of modelling the future and highlights potential challenges and pitfalls when building a model. This discussion remains rather generic, however, as the limitations of the individual studies have already been reviewed in the respective chapters.

The modelled time horizons of the individual studies of this thesis are varied but extend to the end of the century. Reviewing the history of the last 80 years – roughly the timespan until 2100 – several turnovers have strongly influenced global land use. The reunification of Germany, for instance, led to considerably lower harvests in East Germany (Niedertscheider *et al.*, 2014). Additionally, the Nagorno-Karabakh conflict between Armenia and Azerbaijan (1991–1994) prompted a strong abandonment of agricultural fields, which were not re-cultivated (Baumann *et al.*, 2015). This suggests that armed conflicts and socio-economic and institutional changes have a significant impact on land use, yet it is difficult to imagine that the German reunification and its effects on land use could have been projected 80 years prior to World War II and the country's division. Such relatively abrupt and unexpected land-system changes have been common throughout in history – even after long, steady periods – and can limit the predictability of future land-use change (Müller *et al.*, 2014).

Clearly, neither general land-use modelling for coming decades nor the results of this thesis can offer precise forecasts. Nevertheless, modelling can be a useful exercise. A popular aphorism by the statistician George Box says that 'all models are wrong but some are useful' (Box, 1979). If constructed to the best of one's knowledge and validated thoroughly against reality, models can be valuable tools for informing decision-makers of plausible futures, assessing the effect of policies, and enlarging scientific knowledge.

2.1. Challenges of modelling land use

The modelling of complex systems, such as the global land-use system, poses numerous challenges. Sohl and Claggett (2013) have suggested that 'one of the biggest challenges facing the LULC modeler is to reduce the complexity of the LULC system to a practical level while still representing the important processes affecting system behaviour'. Representing the key processes in the model is challenging for several reasons. First, the underlying processes and their interplay might not be fully understood or may be too complex to be represented in the model, which could result in a structurally invalid model. Such a model might even produce decent results over a limited and validated time span, but for the wrong reasons. Second, even with knowledge of general interdependencies, precise numbers to quantify and parameterise them in the model might be missing. This is a common problem in global land-use modelling, where certain elements in the model, such as costs, must be estimated on the basis of limited data. In addition, land-use models commonly rely on external data and other model results that could contain mistakes. For instance, the FAOSTAT database, which is a main data source for the MAgPIE model, is known to contain inconsistent values (Yu & Abler, 2014).

Another potential problem regarding long-term land-use modelling is that it makes projections for a state that can differ drastically from the present but must assume that observed relationships persist also outside calibration conditions. This is the case for the relationship between investments in R&D and expected yield increases (Dietrich *et al.*, 2014). It is likely that further yield increases can be achieved over the next decade on the global level, but for single regions and in the long term, both a stagnation of yields and major technological breakthroughs are not completely implausible scenarios (Ray *et al.*, 2012, 2013; Grassini *et al.*, 2013). Lastly, even if the structures of processes are known and correctly quantified, they might be wrongly implemented in the model as a result of programming mistakes, which are often referred to as 'bugs'. When creating or advancing a land-use model or interpreting its results, one should be aware of these potential shortcomings.

2.2. Realism of scenario assumption

Scenarios have been integral in this thesis to study the effects of different policies and other assumptions on future development. These scenario assumptions, however, also include a degree of simplification. This specifically regards the two policies with the most prominent representation in the scenario set-up of the three papers: an expansion of protected areas and a carbon price on land-use change emissions.

Increased protection of tropical intact and frontier forests, comprising all such forests by the middle of the century, formed one scenario of Chapter II. Chapter III assumed an establishment of protected areas in biodiversity hotspots and frontier forests in Latin America and Africa. Both studies entailed two strong assumptions. First, protected areas were assumed to completely avoid land-use change inside their boundaries. While protected areas have proven to significantly reduce deforestation and conversion to agriculture, their effectiveness is far from perfect (Andam *et al.*, 2008; Nelson & Chomitz, 2011; Beresford *et al.*, 2013; Heino *et al.*, 2015; Spracklen *et al.*, 2015). Second, the assumed extent of these protected areas is rather large compared to the current size of protected areas and the Aichi target of expanding protected areas to 17% of the terrestrial area (CBD, 2010). Given that in order to be effective, protected areas must be gazetted and proper management and enforcement are necessary on the ground and cause conflicts (Soares-Filho *et al.*, 2014; Tollefson, 2015), a rapid expansion of effectively protected areas is unlikely. These scenarios should therefore not be interpreted as highly likely representations of the future, but they do depict possible effects of these rather extreme cases.

Several scenarios in Chapters II and IV assumed a globally established price on emissions from forest clearing. In the study on afforestation (Chapter IV), this price additionally incentivised CDR through forest plantations. Two preconditions are necessary to realise this scenario: the political decision and a system for monitoring, reporting and verification (MRV). Currently, no greenhouse gas pricing exists on the global level, and existing regional taxing or emission trading systems do not include land-use change emissions (World Bank Group & ECOFYS, 2016). Even though efforts to monitor emissions from deforestation or reforestation could be lower than for agricultural N_2O and CH_4 emissions, since it could largely rely on remote sensing, transaction costs could be substantial depending on the level of accuracy (Böttcher *et al.*, 2009). Lastly, the monitoring of deforestation emissions does not address who would be liable to pay for emissions that are caused, for instance, by illegal deforestation on public land. Presently, implementation of a global scheme for pricing of land-use change emissions does not seem particularly likely in the near future; however, a linking of different national systems could have a similar effect. The modelled scenarios reveal the effect of emission pricing in general, whether this is done with a tax on the global level or several individual systems at the national level.

3. Potential developments for modelling the future

As discussed earlier, models are simplified representations of reality. In order to study emerging research questions new models are being build, or existing models are modified to fulfil the specific research needs. However, not all issues can be addressed within a short period of time, as they require more resources and longer model development, and they have therefore been excluded from the context of this thesis. These aspects are roughly in line with the four general suggestions that Sohl and Claggett (2013) have made to improve the utility of land-use models: (1) more transparency through provision of model code and documentation, (2) use of scenarios to assess uncertainty, (3) advance model in a way most relevant to stakeholders and (4) better validation and quantification of uncertainty.

It is also important to note that simply adding components to the model is not always the appropriate solution. One must ask beforehand whether the gain in information content via new model components is worth the increase in model complexity (Sohl & Claggett, 2013). A higher number of modelled processes usually corresponds to higher uncertainty and increases error propagation (van Delden *et al.*, 2011). A model with higher complexity and more variables and parameters is also at greater risk of overfitting if the model is calibrated. Bryngelsson and Lindgren (2013) have argued that more detailed models will probably perform better for shorter time spans where many processes matter, while for modelling longer time horizons more detail does not outweigh the decrease in transparency. This is consistent with the argument by Engström *et al.* (2016a) that models which are simple – and therefore transparent – might be sufficient to make predictions about future land use, at least on the continental and global scales.

The following sections highlight several aspects that could be considered in future MAgPIE development in accordance with the aforementioned remark that a more complex model is not necessarily superior.

3.1. Spatial resolution of biogeophysical data

Spatial data in the MAgPIE model is commonly included on a geographical resolution of 0.5° x 0.5°. The land-use pattern in the initial year and the yield information from LPJmL are, for instance, integrated on this level. For the optimisation, this information together with the socio-economic data is then aggregated to a lower number of simulation units, e.g. 1,000 clusters. After solving the model, the calculated land-use change since the initial year is disaggregated again to the 0.5° resolution by a relatively simple algorithm. The total loss in forest area in one cluster, for example, is allocated to all cells containing forest. With research questions that focus on the spatial patterns of land use on the local level, such as in the assessment in Chapter IV, this approach reaches its limit.

In recent years, multiple new geographical datasets have been published that could be used to update the land-use pattern and increase the spatial resolution of the model. Especially new global land-cover datasets, which are based on remote sensing information, are constantly being published. The Climate Change Initiative (CCI) of the European Space Agency has recently provided an annual global land cover time series for the years 1992 to 2015 at a resolution of 300m (ESA, 2017). In 2013, a global dataset of forest cover in 2000 and 2012 at a spatial resolution of 30m was released (Hansen *et al.*, 2013).

On the regional and national levels, several further high-resolution land-cover datasets are available. Corine provides land-cover information for the EU between 1990 and 2012 on a resolution of 100m (EEA, 2013). Moreover, the Multi-Resolution Land Characteristics (MRLC) Consortium has offered land-cover information for the USA on a 30m resolution (Homer *et al.*, 2015), and the Africover project has produced land-cover information for several African states (GLCN, 2013). Such data, which is not available on the global level, would have to be embedded in a framework that allows for the usage of geographical information at different scales. This would be especially valuable for assessments that focus on smaller geographical regions, e.g. on single countries. Based on global, regional and national data, Fritz *et al.* (2015) have created a map of global croplands. Many of the newly published datasets present land-cover information at a much higher resolution than currently included in the MAgPIE model and/or for several points in time. However, MAgPIE models land-use rather than land-cover, so the datasets cannot be used directly but would instead require substantial pre-processing. Examples of newly released global land-use datasets includes the following: HYDE 3.2 (Klein Goldewijk *et al.*, 2016), which distinguishes intensively used pasture and less intensive rangeland; LUH2 (Hurtt *et al.*), which is based on the HYDE data and future projections from the integrated assessment community; and a downscaled dataset on the level of 30" based on LUH (Hoskins *et al.*, 2016).

It is neither possible nor meaningful to merely increase the spatial resolution of input data but solve the model with the same number of simulation units as before. Two distinct approaches exist to increase the spatial representation in the model. One possibility is to increase the number of simulation units, but computational constraints limit this option. Despite drastic advancements in speed and memory availability in recent years, both these factors still limit the complexity of models. In order to avoid this, the MAgPIE model could either be converted into a linear optimisation problem – thus allowing for a higher number of variables – or simulations for the world regions could be computed independently and in parallel.

The other option is to separate the allocation procedure on different spatial levels into distinct model components. The LUMOCAP Policy Support System (van Delden *et al.*, 2010) uses such a hierarchical approach. Total production is calculated on the EU-27 level and a spatial interaction and distribution model allocates broad land use to countries and regions while a cellular automata model determines it for a resolution of 1 km x 1 km. In a last step, a crop-choice model specifies the crop type within the agricultural areas. This approach offers several advantages. For example, it keeps the core of the model slim so that applications that do not require a high spatial resolution can be computed more quickly. At the same time, it could take into account that different processes have a varying influence at different spatial resolutions. On a more local level, effects of proximity (for instance to roads and existing agriculture) might play a significant role, which a cellular automata model could better represent. A challenging but important determination is the appropriate spatial level for the different model components.

3.2. Resolution of socio-economic data and regional policies

An integration of yield and land-use information on a higher spatial resolution into the MAgPIE model alone would not automatically improve the accuracy of model predictions. An update to the demand side would likely be necessary as well to achieve this.

Socio-economic data in the MAgPIE model is currently represented on the level of 10 world regions. This comprises the demand side of the model also information on costs for production. To more accurately represent processes that influence the spatial pattern of agricultural production, this information should be included at a higher resolution. Main sources for this information are the FAOSTAT and GTAP databases. The FAOSTAT (2016b) database reports information on current agricultural production and consumption on the level of countries, and the latest GTAP version distinguishes 140 regions (Aguiar *et al.*, 2016). This offers the ability to include this data on a much higher geographical level. A dynamic regionalisation of the model would enable its customisation for specific research questions.

Other regional and national policies, besides the protected areas that are considered within this thesis, also influence land-use. Two examples, one targeting energy production and one focusing on land-use conservation, are as follows: (i) The Renewable Energy Sources Act (EEG) and its feed-in tariff system, which strongly influenced the area that was used for maize for energy production in Germany (Gömann *et al.*, 2010; Peterson *et al.*, 2011) and (ii) Brazil's Forest Code, which is central to land-use management in Brazil and requires landowners to designate a certain share of their property for natural vegetation (Stickler *et al.*, 2013; Soares-Filho *et al.*, 2014). As such national and sub-national policies are in place in many regions, they also influence land-use significantly on the global level. To achieve more robust projections of future land use, it is therefore important to also consider the effects of such regional policies, especially when studies focus on single regions. A finer level on which results should be reliable requires a finer level of the policies under consideration.

3.3. Temporal resolution and modelled time span

So far, the MAgPIE model has been initialised for the year 1995 and then computed in time steps of 5 or 10 years until, at maximum, the year 2100. This could be improved in two respects, namely the temporal resolution and the modelled time span.

Increasing the temporal resolution entails shortening the modelled time steps. Yearly time steps, for instance, would offer an ability to assess the effects of inter-annual yield variability. Climate variability is responsible for a substantial portion of yield variability, with differing trends worldwide (Ray *et al.*, 2015; Iizumi & Ramankutty, 2016), but in contrast to long-term changes in climate, its effect on yields are currently not considered. Yet, yield variability can be decisive for food prices and can add to the pressures that derive from afforestation, as modelled in Chapter IV. Drought-induced production decreases, for instance, were partially responsible for wheat price hikes and amplified the food price effect of bioenergy production (Mueller *et al.*, 2011).

A longer, historically modelled time span would help to assess the performance of the model. FAOSTAT (2016b) data for main agricultural characteristics are available from the 1960s onwards. However, this data is far from complete, as initially fewer countries were reporting, and it is still far from perfect today in view of its many inconsistencies (e.g. twice as much pork was produced in China than was consumed (Yu & Abler, 2014)). Nevertheless, use of historical data for a longer time span would allow for eliminating some uncertainty by prescribing selected parameters (such as the yield development over time), and thereby assess the accuracy of other modelled parameters. However, it remains challenging that there is no consistent, global land-use dataset based on observational data for this time span. The popular HYDE database (Klein Goldewijk *et al.*, 2011) relies on remote sensing data for around the year 2000 only; data for previous years was generated according to a heuristic that considered soil suitably and proximity to coasts and rivers. A validation against this dataset would probably show a good result if it considered similar factors in allocating land use, irrespective of real historical land-use patterns.

3.4. Improved representation of demand

Regional caloric demand is exogenously read into the MAgPIE model. It is pre-calculated based on a time-dependent regression on historical consumption and per-capita income and is computed for the future by the use of GDP (Dellink *et al.*, 2017) and population (KC & Lutz, 2017) projections (Bodirsky *et al.*, 2015). The livestock share in total consumption is determined in the same way, while the food composition within the vegetal and meat share is assumed to remain constant over time. This ensures consideration of the main components that drive long-term food demand, namely population and income (Tilman *et al.*, 2011; Alexandratos & Bruinsma, 2012). However, in reality, the demand for agricultural products is more dynamic than the model currently represents. Whereas food demand is generally considered to be relatively inelastic to a change in prices and the own-price elasticity is typically below one, prices still influence food consumption (Muhammad *et al.*, 2011). In addition, body weight, demographic structure and physical activity level of the population also influence average per-capita food energy requirements (Hiç *et al.*, 2016). Moreover, the composition of diets changes over time. For example, the consumption of sugar and vegetable oils has increased in developing countries in recent decades (Kearney, 2010). Future modelling should take into account not only a variety of drivers that affect the per-capita demand for calories but also the composition of diets.

A uniform elasticity cannot represent the demand response to changing prices since it varies strongly for different commodities. Andreyeva *et al.* (2010) have estimated the price elasticity for food expenditure in the USA based on a meta-analysis. Softdrinks (0.79), beef (0.75) and pork (0.72) present a relatively high elasticity, while it is low for sugar (0.34) and eggs (0.27). Food that is consumed outside of the home, e.g. in restaurants, shows the highest elasticity (0.81). Changing prices of one commodity will influence not only the demand for that good but also complements and substitutes. Canadian poultry and pork demand, for instance, is estimated to rise with an increase in beef price (Pomboza & Mbaga, 2007). Price elasticities also vary strongly between regions. Low-income countries exhibit stronger demand changes in response to price changes than countries with a higher income. While the own-price elasticity for cereals is about -0.5 in the Democratic Republic of Congo, it is almost zero in the USA (Muhammad *et al.*, 2011). This is also explained by low expenditure shares on food in high-income countries, which also leads to a minor income effect for price shocks. Households in high-income countries spend only about 20% on food, of which only about 9% are on cereal products – so, less than 2% of total available income (Muhammad *et al.*, 2011). In addition, only about 15 cents of each dollar that is spent on food in the USA are for farm production (Canning, 2011); all other 85% that determine the price are not directly influenced by changing prices of the primary produce. This is one reason why farm prices are not fully transmitted to the retail level (Lloyd, 2017). In regions with a large share of highly processed goods consumption, a change in the price of primary products can also be compensated for by buying less-processed products or switching to cheaper brands, which would not change the absolute amount consumed. As a result of these factors, the food price elasticity decreases with increasing wealth and will also diminish in importance in currently developing regions.

3.5. Macro-economic effects

Agriculture requires inputs from the manufacturing sector, such as fertilisers or machinery, and is additionally linked to non-agricultural sectors through the provision of food, fibres and fuels. Changes in the agricultural sector therefore influence other sectors, and vice versa. MAgPIE is a PE model of the land-use sector, and it thus represents the agricultural system in significant detail but largely neglects feedbacks from the rest of the economy (Michetti & Zampieri, 2014). As a consequence, certain aspects that might have also been relevant for the results of this thesis could not be quantified.

Changing agricultural prices can have several effects. As discussed, the price of food commodities has two consequences: it directly influences the demand via the price elasticity of demand, and it determines available income, which in turn affects demand. Higher prices can result in higher incomes for farmers and other workforce members in the agricultural sector, who are usually also consumers. In many parts of the world where smallholder agriculture still prevails (Fernandes, 2013; Graeub *et al.*, 2016) and employs a large share of the population, increasing prices can lead to income increases for many and can thereby boost demand. On the other hand, an increase in regional prices would reduce net exports, and thus domestic income (Khan & Ahmed, 2014). Climate change mitigation-related price shocks would probably affect several regions to various degrees.

The PE approach also limits the study of trade effects. Since it does not consider other sectors besides agriculture, the influences on the total trade balance and exchange rates remain unclear. Changes in the exchange rate could alter the trade patterns compared to a model that does not factor in this aspect.

Pricing of greenhouse gas emissions was a prominent scenario assumption in this thesis; however, the effect on the whole economy was neglected. Carbon pricing can become an important source of public income, may affect the revenues of capital goods and could alter the distribution of resources (Siegmeier & Franks, 2017). A greenhouse gas price that fosters afforestation could substantially change regional income levels and income distribution. Landowners in particular would profit from such a policy, which could also generate substantial flows of money from developed emitter countries into less-developed countries that are rich in land.

The findings of Chapter IV also highlight the need for continuous agricultural yield increases if afforestation were to take place on land that was formerly used for agriculture. Technology-driven yield increases could reduce the need for labour in the agricultural sector, thus raising unemployment in the short term. On the other hand, labour shifts from a low-productive agrarian sector to a more productive manufacturing sector could increase total economic output (Kalkuhl & Edenhofer, 2016).

To overcome some of these limitations, PE land-use models can be coupled with top-down CGE models so that, in the perfect case, the solution does not differ from that of an extended CGE model (Palatnik & Roson, 2012). Currently, MAgPIE can be linked to the energy system model REMIND to exchange information on bioenergy demand and prices (Klein *et al.*, 2014), and developments to deepen the inter-linkages are undertaken.

3.6. Validation

Model validation is an integral part of model development and is essential to build trust in model results. Learning from validation is also the most efficient way to improve the predictive power of land-use models (Pontius *et al.*, 2004) and helps to determine the spatial level at which model results are reliable. Notwithstanding its importance, Pontius *et al.* (2004) have identified validation as 'the weakest part of contemporary LUCC modeling'. The availability of new data, such as the aforementioned new land-use datasets or data on the temporal development of irrigated areas (Siebert *et al.*, 2015), potentially enlarges options to validate global land-use models. For MAgPIE, the situation also improves since the temporal overlap of the modelled time span and observed data increases constantly. Nevertheless, the quality assessment of land-use models in general, and of MAgPIE in particular, remains challenging. Lastly, Baldos and Hertel (2013) have stressed that 'successful model validation is also confounded by the fact that agricultural models must predict human behaviour'.

Different data for both calibrating and validating the model are essential for meaningful validation as well as to avoid overfitting. Two approaches are usually applied to avoid this problem: a separation through space and a separation through time (Pontius *et al.*, 2004). Separation through space is commonly employed when classifying remote sensing data. Datasets are then split up into a fraction that is used for training and parameter tuning while another part is used for testing the model. For MAgPIE, this approach is not possible due to the global nature of the model. Separation through time, so a validation of the model with data from time steps that were not used in parameterising the model, was done, also within this thesis. However, several datasets, such as the one on spatial land use (Erb *et al.*, 2007), are available for only one point in time.

The disagreement of modelled and observed data that exists already in the year of model initialisation renders the validation for MAgPIE more challenging than for other land-use models. Rule-based models which allocate new cropland based on suitability usually exhibit perfect agreement with the input land-use dataset in the starting year of the model. Land-use patterns in MAgPIE, in contrast, differ from the input in 1995 due to the optimisation approach, which complicates validation.

Many land-use models work with grid cells that can take only one discrete land-use class, while MAgPIE cells contain different shares of all land-use classes. Established methods to compute the accuracy of predictions, such as the percentage of correctly classified cells or the kappa coefficient, which shows the agreement between categorical datasets relative to what can be expected by chance (van Vliet *et al.*, 2011), can therefore not be utilised. This difference raises new questions of how to judge the quality of the model, such as whether a small change in the wrong direction (e.g. 1% decrease in cropland modelled instead of 2% increase in reality) is worse than a larger mismatch with the correct sign (10% increase vs. 2% increase). Such questions must be addressed prior to the development of a quality criterion for the results on the grid level.

The number of cells that changes over the modelled time period is usually small compared to the number of cells that do not. As a consequence, maps of modelled and observed data will appear similar, and simple statistical measures such as the R^2 will show high agreement. In this case, even a null model predicting no land-use change at all will often perform better than a land-use model at its full model resolution (Pontius & Spencer, 2005). Many studies insufficiently consider this fact, and van Vliet *et al.* (2011) have noted that 'in the evaluation of land-use change models [...] the agreement between the model result and the actual land-use data is meaningless when the amount of change is not considered'.

To deal with these challenges, future MAgPIE validation should focus on the following aspects. First, the validation of the calibration (leading to values and land-use pattern in the starting year of the model) should be clearly distinguished from the validation of the development over time. Second, the quality assessment should also separately consider general trends (e.g. whether the global increase in cropland was correctly predicted) and an evaluation of location accuracy (whether the change was predicted at the right location) (Brown *et al.*, 2005; van Vliet *et al.*, 2016). In doing so, the assessment should consider model results at different spatial aggregation levels and compare results to a null model that predicts pure persistence as well as a random model which assumes even distribution of change (Pontius *et al.*, 2004). This will probably require the definition of a new quality criterion that is adapted to the specific characteristics of MAgPIE. Another proposition is to focus validation efforts on certain main historical developments also known as 'stylized facts' (Baldos & Hertel, 2013). At best, validation should focus on as many model outputs as possible – not only land use but also production quantities, prices, yield developments, emissions and trade volumes.

3.7. Communication of uncertainty and sensitivity analysis

The communication of uncertainty is a topic that strongly relates to model validation, and it is of similar importance. The communication of uncertainty has been subject to heated debates in the context of the IPCC (Budescu *et al.*, 2009, 2012; Swart *et al.*, 2009; Jonassen & Pielke, 2011; Adler & Hirsch Hadorn, 2014), and not least because MAgPIE results have found their way into the IPCC reports this is an important topic. The guidelines to the authors of the third (Moss & Schneider, 2000) and fifth assessment reports (Mastrandrea *et al.*, 2010) list key steps in discussing uncertainty. They advise consideration of all plausible sources of uncertainty and identification of those most likely to affect conclusions. Whenever possible, probabilistic information and ranges should be presented, and the precision of results should be adapted to the level of uncertainty.

A precise quantification of model uncertainty and a calculation of error propagation for MAgPIE is probably impossible given the complexity of the model, its nature as an optimisation model and the lack of probability information for input data. However, a sensitivity analysis, or analysis of model output in response to a variation of model elements (Brown *et al.*, 2013), is possible. It has been demonstrated in Chapter II for land-conversion costs, which strongly influence the amount of land-use change. In contrast to faster models (e.g Engström *et al.*, 2016b) a Monte-Carlo sensitivity analysis, whereby a large number of parameters is varied systematically, is not possible due to the runtime of the model.

The use of scenarios spanning the entire plausible range of developments and the use of ensembles of model runs are other means to address the difficulty of predicting human behaviour (Swart *et al.*, 2009). Scenarios are similar to sensitivity analysis but diverge in that they usually comprise a set of model parameters. Scenario analysis is already a core component of modelling with MAgPIE, which entails parameterisation of, for instance, the shared socio-economic pathways (O'Neill *et al.*, 2017; Popp *et al.*, 2017).

Model intercomparisons help to estimate the possible range of outcomes and offer insight about model behaviour. Since models are built independently to a certain degree, such comparisons can reveal some of the uncertainty that results from the model structure. The AgMIP has compared 10 PE and CGE models and indicated that despite harmonisation of key assumptions, different models predicted different trends for main model outcomes (Von Lampe *et al.*, 2014). Combined with model intercomparisons of global gridded crop models (GGCMs) (Rosenzweig *et al.*, 2014), which produce input to land-use models, such assessments are a valuable source of information for evaluating the uncertainty and should play a prominent role in future research. On the downside, the comparison of model results could put pressure on modelling teams to adjust models in such a way that they show a higher degree of similarity.

3.8. A task without a clear end

Model development of models that aim at representing the complex global land-use system is not a task that will be finalised at some point in time. The availability of new data and new theories can improves and enables continuous advancements. New research questions, changing policies and novel technologies have to be reflected in the model structure. Higher computational resources will facilitate more complex and highly resolved models. Nevertheless, time for model development and model runs will remain a limited resource, so only a selection of possible developments can be realised in the future, and the prioritisation of model developments with respect to the questions at hand will persist as one of the biggest challenges.

Ultimately, land use is determined by individuals – consumers who demand for a certain product or farmers planting a certain crop – and it is impossible to know and model individual decisions and predict land use on this level. This is an important restriction to be aware of when modelling future land use. Fortunately, many research questions do not necessitate this level of detail, and it is sufficient to model the 'average land-user'.

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List of Abbreviations

AFOLU	Agriculture, Forestry and Other Land Use
AgMIP	Agricultural Model Intercomparison and Improvement Project
AR5	Fifth Assessment Report of the IPCC
BECCS	Bioenergy with Carbon Capture and Storage
CCI	Climate Change Initiative
CCS	Carbon Capture and Storage
CDM	Clean Development Mechanism
CDR	Carbon Dioxide Removal
CGE	Computable General Equilibrium
CO_2eq	Carbon dioxide equivalent
FAOSTAT	Food and Agriculture Organization Corporate Statistical Database
GDP	Gross Domestic Product
GGCM	Global Gridded Crop Model
GTAP	Global Trade Analysis Project
GWP	Global Warming Potential
HANPP	Human Appropriation of Net Primary Production
HYDE	History Database of the Global Environment
INDC	Intended Nationally Determined Contributions
IPCC	Intergovernmental Panel on Climate Change
LPJmL	Lund-Potsdam-Jena managed Land (model)
LUH	Land-Use Harmonization project
LULC	Land Use and Land Cover
LULUCF	Land Use, Land-Use Change and Forestry
MAgPIE	Model of Agricultural Production and its Impact on the Environment
MRLC	Multi-Resolution Land Characteristics Consortium
MRV	Monitoring, Reporting and Verification
NPP	Net Primary Production
PADDD	Protected Area Downgrading, Downsizing and Degazettement
PE	Partial Equilibrium
R&D	Research and Development
REDD	Reducing Emissions from Deforestation and forest Degradation
SSP	Shared Socioeconomic Pathway

Statement of Contribution

Chapter II Agricultural trade and tropical deforestation: interactions and related policy options

C. Schmitz, H. Lotze-Campen and A. Popp designed the study. C. Schmitz developed necessary model amendments, conducted the model simulations, analysed the data and drafted the manuscript. U. Kreidenweis contributed to a major revision of the manuscript. To this end, he conducted new model simulations and improved the text and the graphical data representations in response to the reviewer's comments. C. Müller contributed LPJmL model input data. C. Schmitz, H. Lotze-Campen, A. Popp, M. Krause and J.P. Dietrich developed the MAgPIE model. All authors discussed the manuscript.

Chapter III Unbridled international trade eats up the land-savings from pasture intensification and diminishes the potential for conservation

U. Kreidenweis, A. Popp, F. Humpenöder, L. Kehoe and T. Kuemmerle developed the overall study design. U. Kreidenweis created the required new model components, ran the simulations and developed the figures with significant contributions from F. Humpenöder, A. Popp and L. Kehoe. U. Kreidenweis drafted the manuscript. H. Lotze-Campen, A. Popp, B.L. Bodirsky, F. Humpenöder and U. Kreidenweis contributed to the development of the MAgPIE model. All authors discussed and amended the manuscript.

Chapter IV Afforestation to mitigate climate change: impacts on food prices under consideration of albedo effects

U. Kreidenweis and A. Popp designed the research. U. Kreidenweis and F. Humpenöder developed and programmed the central model components. U. Kreidenweis ran the model simulations, drafted the manuscript and designed the figures. U. Kreidenweis, A. Popp, F. Humpenöder and M. Stevanović analysed the model results. H. Lotze-Campen, A. Popp, B.L. Bodirsky, F. Humpenöder, M. Stevanović and U. Kreidenweis contributed to the MAgPIE development. All authors commented on the manuscript.

List of Publications

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Page 69-93:

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