Land and water for agriculture
-
future prospects and trade-offs

vorgelegt von

Diplom-Physiker

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der Technischen Universität Berlin
zur Erlangung des akademischen Grades
Doktor der Naturwissenschaften,
Dr. rer. nat.

genehmigte Dissertation

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Tag der wissenschaftlichen Aussprache: 3. September 2015

Berlin 2015

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Summary

Land and water are among the most vital resources for human wellbeing and terrestrial life in general. Therefore, sustainable management strategies for land and water need to be developed in order to sustain a growing world population and important natural ecosystems. Agricultural activity for food, material, and energy production constitutes the most profound anthropogenic influence on global land and water resources. Thus, agricultural land and water-use is a key determinant of the sustainability of resource management strategies.

The main research objective of this thesis is to explore, how agriculture might affect global land and water resources in the future. In individual studies, projections of agricultural land and water-use are developed, considering important drivers such as population growth, economic development, bioenergy demand, terrestrial climate change mitigation strategies, and sustainable water-use strategies. In a synthesis, the results of the individual studies are combined in order to answer the following research questions: (1) How do different agricultural strategies compare in terms of environmental implications for land and water resources and required transformation of the system? (2) How large is the operating space for land and water management strategies in agriculture?

Methodologically, this thesis relies on the Model of Agricultural Production and its Impacts on the Environment (MAgPIE). MAgPIE can be used to derive scenarios of future agricultural activity by considering socioeconomic drivers and biophysical constraints in a cost optimization framework. For this thesis, the model was extended by a detailed water sector that features an improved representation of water availability, irrigation infrastructure, non-agricultural water demand, and environmental water requirements.

Results indicate that protecting freshwater ecosystems from degradation due to agricultural activity can be achieved without fundamental trade-offs in terms of cropland expansion into terrestrial ecosystems. Terrestrial climate change mitigation strategies will likely require a fundamental transformation of the agricultural system. Environmental consequences differ between mitigation strategies. While afforestation and strategies to avoid land-use change emissions can help to protect or even create important terrestrial ecosystems, large-scale bioenergy production can put a severe threat to terrestrial and freshwater ecosystems. Pronounced trade-offs between land and water for bioenergy production suggest that it is crucial to consider both, land and water-use, when aiming at sustainable bioenergy production.

Based on the here presented results, there is hope that the operating space for agriculture is large enough to support a variety of management strategies. On the one hand, resource constraints still leave ample space for increasing land and water inputs for agriculture. On the other hand, the system seems flexible enough to allow for increasing food production with a reduced environmental footprint in terms of land and water appropriation.

Zusammenfassung

Land und Wasser gehören zu den wichtigsten Voraussetzungen für menschliches Wohlergehen und landbasiertes Leben. Daher braucht es nachhaltige Strategien zur Land- und Wassernutzung, um eine wachsende Weltbevölkerung zu versorgen und wichtige natürliche Ökosysteme zu erhalten. Landwirtschaftliche Nahrungsmittel-, Material- und Energieproduktion stellt den tiefgreifendsten menschlichen Einfluss auf globale Land- und Wasserressourcen dar. Landwirtschaftliche Land- und Wassernutzung ist daher ein essentieller Faktor für die Nachhaltigkeit von Ressourcenmanagementstrategien.

Das hauptsächliche Ziel dieser Dissertation ist es zu untersuchen, wie Landwirtschaft in Zukunft globale Land- und Wasserressourcen beeinflussen könnte. In einzelnen Studien werden Projektionen von landwirtschaftlicher Land- und Wassernutzung entworfen, wobei wichtige Einflussfaktoren wie Bevölkerungswachstum, ökonomische Entwicklung, Bioenergienachfrage, landbasierte Strategien zur Klimawandelvermeidung und nachhaltige Wassernutzungsstrategien berücksichtigt werden. In einer Synthese werden die Resultate der einzelnen Studien kombiniert um folgende Forschungsfragen zu beantworten: (1) Wie verhalten sich verschiedene landwirtschaftliche Strategien in Bezug auf ökologische Folgen für Land- und Wasserressourcen und benötigte Veränderungen des Systems? (2) Wie groß ist der Spielraum für Strategien zur landwirtschaftlichen Land- und Wassernutzung?

Zur Untersuchung dieser Forschungsaufgaben wird das „Modell für Landwirtschaft und ihre Auswirkungen auf die Umwelt“ (MAgPIE) angewendet. Mit MAgPIE werden Szenarien zukünftiger landwirtschaftlicher Aktivität entworfen. Dabei werden sozioökonomische Treiber und biophysikalische Randbedingungen in einem Kostenoptimierungsverfahren berücksichtigt. Für diese Dissertation wurde das Modell um einen detaillierten Wassersektor erweitert, der die Implementierung von Wasserverfügbarkeit, Bewässerungsinfrastruktur, nicht-landwirtschaftlichem Wasserbedarf und ökologischem Wasserbedarf verbessert.

Die Ergebnisse legen nahe, dass der Schutz von Frischwasserökosystemen vor Degradation durch landwirtschaftliche Aktivitäten ohne fundamentale Auswirkungen auf Ackerlandausdehnung in landbasierte Ökosysteme erreicht werden kann. Landbasierte Strategien zur Klimawandelvermeidung werden wahrscheinlich eine fundamentale Transformation der Landwirtschaft erfordern. Dabei unterscheiden sich die ökologischen Folgen verschiedener Vermeidungsstrategien. Aufforstung und Strategien zur Vermeidung von Emissionen durch Landnutzungsänderungen können helfen, wichtige landbasierte Ökosysteme zu erhalten oder sogar zu erschaffen. Großskaliger Anbau von Bioenergiepflanzen dagegen kann Land- und Frischwasserökosysteme ernstzunehmenden Gefahren aussetzen. Die ausgeprägte Austauschbeziehung zwischen Land und Wasser für die Bioenergieproduktion legt nahe, dass es für nachhaltige Bioenergieproduktion entscheidend ist, sowohl die Land-, als auch die Wassernutzung zu berücksichtigen. Basierend auf den hier vorgestellten Ergebnissen gibt es Hoffnung, dass der Spielraum für

die Landwirtschaft groß genug ist um die Anwendung verschiedenster Strategien zu erlauben. Einerseits lassen Ressourcenbeschränkungen reichlich Spielraum für erhöhte Land- und Wassernutzung in der Landwirtschaft. Andererseits scheint das System flexibel genug zu sein, um eine Erhöhung der Nahrungsmittelproduktion bei gleichzeitiger Verringerung des ökologischen Fußabdrucks in Bezug auf Land- und Wassernutzung zu erreichen.

Chapter I

Introduction

Markus Bonsch

Content of Chapter I: Introduction

1 Overview

Human alteration of natural ecosystem dynamics for food acquisition dates back to preindustrial times. While the impact of early human societies on natural systems was local and transitory, the activities of todays industrialized societies have "become so pervasive and profound that they rival the great forces of Nature" (Steffen et al., 2007). Since the beginning of the industrialization around 1800 AD, humans have fundamentally altered the composition of the atmosphere, terrestrial land cover, and global water and nitrogen cycles (Rockström et al., 2009; Steffen et al., 2007). This transformation has pushed the Earth system into planetary terra incognita (Steffen et al., 2007) and could lead to abrupt or irreversible environmental changes that might severely affect human wellbeing (Rockström et al., 2009). Therefore, modern societies need to develop resource management strategies that ensure the sustainability of Earth's life support system (Steffen et al., 2007).

Land and water are among the most vital resources for human wellbeing. The fact that humans cannot survive without access to clean drinking water has led the United Nations (2010) to declare the access to water for drinking and sanitation as a human right. Further direct human water-use involves water for manufacturing and electricity production (Flörke et al., 2013) while direct human land-use comprises settlement and infrastructure areas. The vast majority of global land and water resources however supports the growth of plants that produce a gross energy equivalent of around 1100 Exajoule (EJ) per year (Haberl et al., 2013). A considerable fraction of about 300 EJ per year of this biomass resource is appropriated by human agriculture for food, fibre and energy production (Haberl et al., 2007). Therefore, the sustainability of management strategies for land and water resources crucially depends on land and water-use in the agricultural sector.

Agricultural production has managed to keep pace with a tripling of food demand since 1960 (FAOSTAT, 2013) that was largely triggered by population growth and economic development (Valin et al., 2014). While cropland was expanded by ~10 % (FAOSTAT, 2013), the majority of the production increase was due to crop genetic improvements, fertilizer application and irrigation (Pingali, 2012). Especially irrigation water application has played a crucial role in boosting agricultural yields and has thereby helped to spare land resources (Rosegrant et al., 2009).

Apart from continued population growth and economic development (O'Neill et al., 2014), the future is likely to bring about new challenges for agriculture. Anthropogenic climate change may alter agro-ecological conditions (Schmidhuber and Tubiello, 2007) while avoiding climate change may require land for carbon sequestration (Smith et al., 2013). Bioenergy is expected to play an important role in the future energy mix (Rose et al., 2014) and may compete for land and water resources (Berndes, 2002; Popp et al., 2014). Finally, the protection of natural ecosystems and the associated ecosystem services may require a

reduction of the human appropriation of land and water (Gerten et al., 2013; Rockström et al., 2009).

This dissertation explores how different future socioeconomic development pathways and management strategies may affect agricultural land and water-use at the global scale. The focus is on terrestrial climate change mitigation strategies and on strategies that aim at protecting freshwater ecosystems from degradation due to human interference. All considered management strategies are evaluated with respect to the required transformation of the agricultural system and with respect to environmental consequences for land- and freshwater ecosystems. Special consideration is given to the trade-offs between water for irrigation and land requirements for agricultural production.

The remainder of the introduction first provides background information on the current state and the future drivers of global land and water-use. Subsequently, the research questions that will be addressed in this thesis are presented, followed by a description of the methodology that is used for the analysis. The introduction concludes with an outline of the remainder of this thesis.

2 Current state of global land resources

Humanity appropriates land for different purposes, mainly for food and material production, housing and infrastructure (K.-H. Erb et al., 2007). Natural resource flows on human dominated lands are strongly altered by management practises such as soil sealing, fertilizer and pesticide application, mechanization and irrigation (Foley, 2005; Scalenghe and Marsan, 2009).

Anthropogenic land-use comes at the expense of natural terrestrial ecosystems that also provide a number of important ecosystem services that are essential for human wellbeing. Replacing and degrading those ecosystems can have a variety of adverse effects including habitat loss, biodiversity decline, carbon emissions, soil erosion, and increased frequency and severity of flood events (Bradshaw et al., 2007; Foley et al., 2007).

Land is a finite resource, constrained by the extent of terrestrial land surface. The suitability of land for human land-use, especially for agriculture, crucially depends on local factors such as climatic conditions, slope and soil quality (Bandyopadhyay et al., 2009). Around three quarters of the terrestrial land surface, including most of the high quality land for agriculture, have to date to various extents been modified by direct human action (Figure 1) (K.-H. Erb et al., 2007).

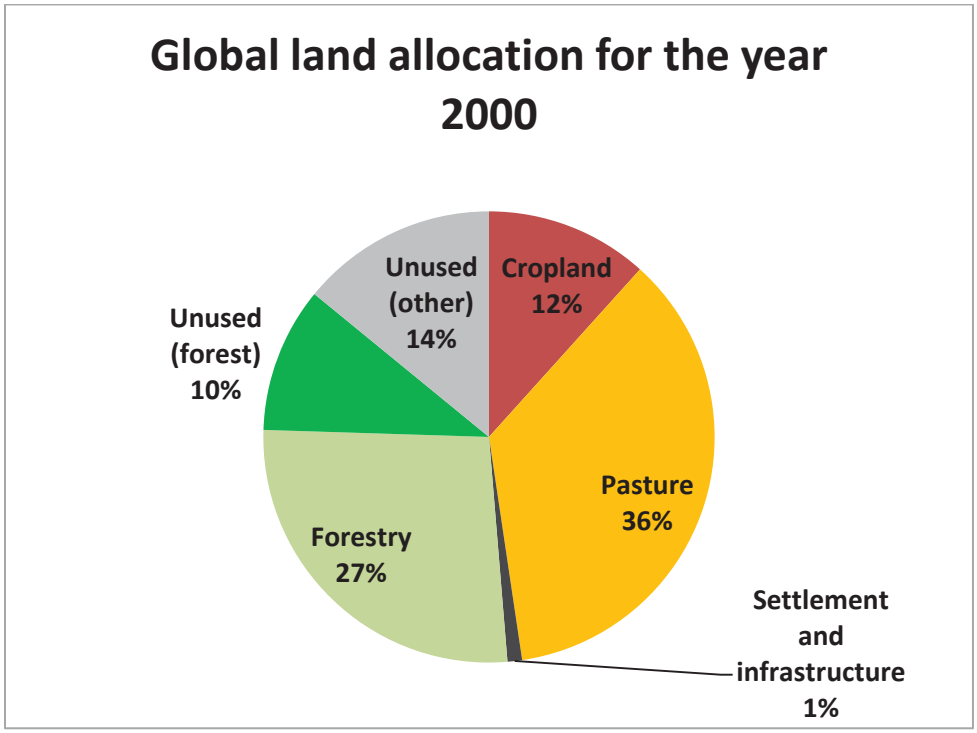

Figure 1 Global land allocation for the year 2000. Data for all categories except for Unused (forest) from Erb et al. (2007). Unused (forest) corresponds to primary forest area from the Global Forest Resources Assessment by the FAO (2010).

Settlement and infrastructure (S&I) areas provide essential services including housing and the potential for economic activity. S&I areas are characterized by artificial land cover such as buildings or roads (K.-H. Erb et al., 2007) and involve a substantial modification of the natural land cover. Especially in urban centers, the extent of vegetation cover is low and soils are largely sealed (Scalenghe and Marsan, 2009). Associated negative environmental impacts include flood risk, biodiversity decline, and changes in local climate (Urban heat islands) (Grimm et al., 2008; Scalenghe and Marsan, 2009). The extent of S&I areas at the global scale is however small (~1%, Figure 1) (K.-H. Erb et al., 2007). Therefore, S&I areas do not play a prominent role in global land-use studies (Schmitz et al., 2014).

Agricultural areas (cropland and pasture) are used for food and material production. Cropland supports the production of food and material crops while pasture supports livestock production through grazing and hay production. Agricultural areas cover around 48% of the global land surface and thus constitute the largest anthropogenic land-use class (Figure 1) (K.-H. Erb et al., 2007). Agriculture is vitally important for human wellbeing since it is responsible for the major part of the global food production. The cereals rice, wheat, and maize are the most important sources of food for human consumption and livestock products provide around 83% of the average global per capita animal protein intake (Food and Agriculture Organization of the United Nations, 2013). Thus, agricultural land, especially cropland, is closely related to food security and hunger alleviation (Godfray and Garnett, 2014).

Management practises on cropland are optimizing nutrient and water inputs for maximizing crop growth via large scale fertilizer application and irrigation (Foley, 2005). This requires the annual application of fertilizer containing 100 Tg of reactive nitrogen and contributes significantly to the doubling of global natural reactive Nitrogen sources of around 200 Tg per year (Fowler et al., 2013). Reactive nitrogen causes several environmental damages including water pollution, the destabilization of aquatic ecosystems (Grizetti et al., 2011), air pollution (Moldanova et al., 2011), climate change, and ozone depletion (IPCC, 2013). The significance of water management on cropland is reflected in the 30-40% of global agricultural production that originate from irrigated cropping systems (Rost et al., 2008). Negative consequences include the exploitation of valuable freshwater resources (see Chapter I, Section 3), and soil salinization (Foley, 2005). Biodiversity on intensively managed cropland is impaired (Jangid et al., 2008) and soils are prone to degradation and erosion (Foley, 2005).

The environmental implications of intensively managed pasture areas are in principle similar to those of cropland. However, pasture areas are often used extensively (K.-H. Erb et al., 2007) and valued as species rich ecosystems (Alkemade et al., 2013). Therefore, the average environmental footprint of pasture areas is likely lower than that of cropland. Livestock grazing is sometimes even a prerequisite for sustaining valuable pasture ecosystems (Oba et al., 2000).

Forestry area comprises all forest land used for wood production. Forestry areas cover a large portion (around 27%, Figure 1) of the terrestrial land surface. The category is, however, heterogeneous and includes dedicated forest plantations as well as forests used extensively for logging (K. H. Erb et al., 2007). Although forestry areas cannot rival the ecological value of primary forests, they offer important ecosystem services such as biodiversity conservation and carbon sequestration. Therefore, forestry is often seen less critical than cropping and can even play a crucial role in conservation efforts (Barlow et al., 2007).

In summary, current scientific evidence suggests that human land-use is affecting the majority of the global land surface. Natural systems have been replaced by systems under varying degrees of human management. Agricultural land, especially cropland, is the most important human influence on the land system when considering extent and environmental footprint. It has to be noted, however, that our knowledge about the current state of global land resources is subject to large uncertainties. There are two main sources of land-use and land cover data: census statistics collected by the Food and Agricultural Organisation of the United Nations (FAOSTAT, 2013) and an increasing number remote sensing datasets from satellite observations. Both data sources are subject to significant uncertainties. Grainger et al. (2008) have shown remarkable inconsistencies in global forest data from different Forest Resource Assessments by the FAO that make it difficult to establish scientific evidence for global deforestation trends from census based data. McCallum et al. (2006) have compared

four global satellite derived land cover datasets. While the datasets also differ in the global extent of different land cover categories, uncertainties about the spatial distribution are even more pronounced (Ellis et al., 2013). Finally, there are substantial differences between national aggregates of land-use data from census statistics and remote sensing (K.-H. Erb et al., 2007; Fritz et al., 2010).

3 Current state of global water resources

There is plenty of water on the "Blue Planet" Earth. Around 97.5% of the global water resource is however salt water that is not suitable for drinking and supporting terrestrial vegetation growth (Oki and Kanae, 2006). Renewable global freshwater resources are governed by the global hydrological cycle (Figure 2). ***Precipitation*** over land areas is the only natural, renewable source of freshwater. When the water reaches the ground, there are four main ways ahead. ***Evaporation*** from soils or other land cover returns a part of the water to the atmosphere. Another part is taken up by plants and transpired, thus supporting vegetation growth (***transpiration***). Third, a part of rainwater percolates to deeper layers of the soil, contributing to ***groundwater*** recharge. Finally, the remainder of the rainwater constitutes surface ***runoff*** and enters rivers and lakes. Runoff thus contributes to the water flow in rivers towards the sea (***discharge***).

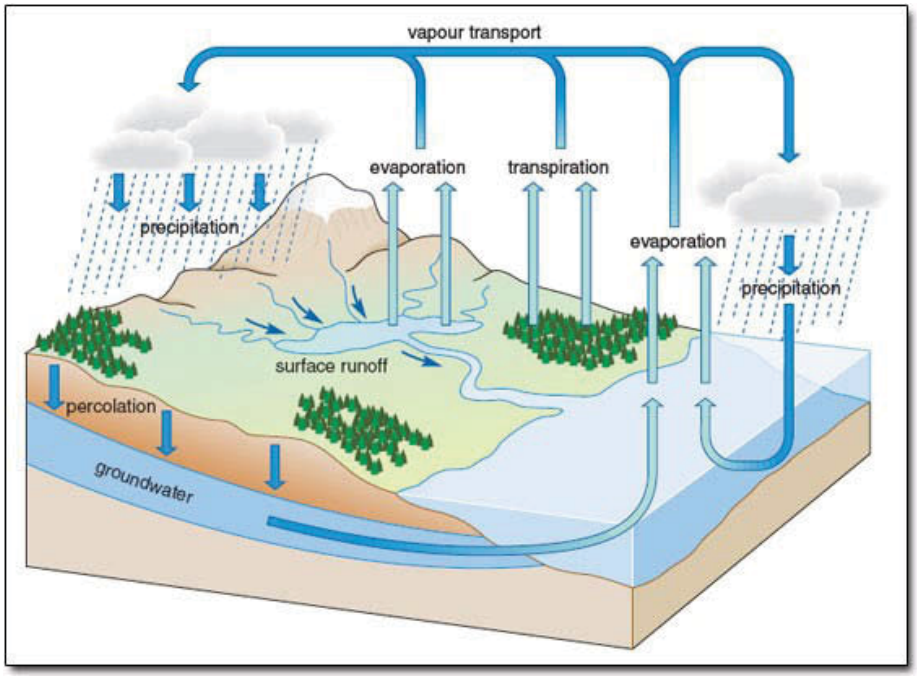

Figure 2 The global water cycle. Figure from Houghton (2004).

Apart from precipitation, natural sources of freshwater include water stored in glaciers and as fossil groundwater. These are, however, not renewable since they constitute a stock of water that was established up to several thousand years ago (Oki and Kanae, 2006). In

coastal areas, humans can gain access to freshwater via desalination of seawater at substantial energy requirements and costs (Elimelech and Phillip, 2011).

For assessments of the human appropriation of water resources, a distinction between “blue” water and “green” water is made (Rost et al., 2008). Green water use refers to the water that evaporates or transpires on land that humans use for agricultural production (cropland and pasture). Blue water use refers to direct human water abstraction from rivers, lakes and aquifers. While human green water use is restricted to agriculture, human blue water use comprises water for domestic purposes (drinking and sanitation), industrial water-use (manufacturing and thermoelectric power production), and agriculture (irrigation and livestock production) (Flörke et al., 2013).

Global estimates of water availability and demand are prone to large uncertainties (Table 1). It can be said, however, that agriculture is the dominant human water-use sector. The majority of global water resources used for agriculture is green water on cropland and pasture areas (Table 1). But even when considering blue water resources only, water withdrawals for agriculture are a factor 2-3 higher than domestic and industrial water withdrawals (Table 1). Only a part of the water withdrawn is consumptively used. The rest flows back to the river system and is available for downstream use after proper wastewater treatment (Flörke et al., 2013). Return flows are especially high for domestic and industrial water-use, so that the dominance of the agricultural sector is even more pronounced when considering consumptive blue water use (Rost et al., 2008).

The global renewable blue water resource is determined by global runoff since desalination does not play a significant role at the global scale (Table 1). At an aggregate global level, water seems to be an abundant resource since only 6-12% of global runoff is withdrawn by humans (Table 1). Water is, however, heterogeneously distributed around the globe, so that there are highly water stressed areas, especially in China, South Asia and the United States (Figure 3) (Oki and Kanae, 2006). It is estimated that over 1 billion people live in areas exposed to high water stress (Oki and Kanae, 2006) and that over 700 million people lack access to safe drinking water (WHO/UNICEF, 2014).

Table 1 Estimates for global water availability and human water-use from the literature.

	Literature estimates (corresponding time period)	Sources and comments
Precipitation	96000 - 118000 km^3yr^{-1} (1979-1999)	Biemans et al. (2009). Comparison of seven global gridded precipitation datasets
Runoff (surface runoff + groundwater recharge)	42000 – 60000 km^3yr^{-1} (1985 – 1999)	Haddeland et al. (2011). Uncertainty across 11 global models. Does not include precipitation uncertainty.
Evapotranspiration	60000 – 85000 km^3yr^{-1} (1985 – 1999)	Haddeland et al. (2011). Uncertainty across 11 global models. Does not include precipitation uncertainty.
Desalination	~14 km^3yr^{-1} (2008)	Schiermeier (2008).
Water storage in large reservoirs	6100 – 8300 km^3 (2000)	Biemans et al. (2011). Data from two different global dam databases.
Human green water use (evapotranspiration over cropland and pasture)	15000 – 15900 km^3yr^{-1} (1971 – 2000)	Rost et al. (2008). Uncertainty from different precipitation datasets.
Blue water use in agriculture (withdrawals)	2200 – 3800 km^3yr^{-1} (2000)	Wisser et al. (2008). Uncertainty from different climate datasets and patterns of irrigated areas.
Blue water use in the domestic, industrial and electricity sectors (withdrawals)	1250 – 1430 km^3yr^{-1} (2010)	Flörke et al. (2013). Comparison of two different model estimates.

Environmental implications of human water-use differ between green and blue water use. Green water use on agricultural areas is not very different from water-use in natural land ecosystem. Anthropogenic land cover change can lead to changes in evapotranspiration and runoff compared to natural vegetation and can thus affect blue water availability. The effect is however highly uncertain and depends strongly on the type of land cover that is altered by humans (Berndes, 2002; Haddeland et al., 2011).

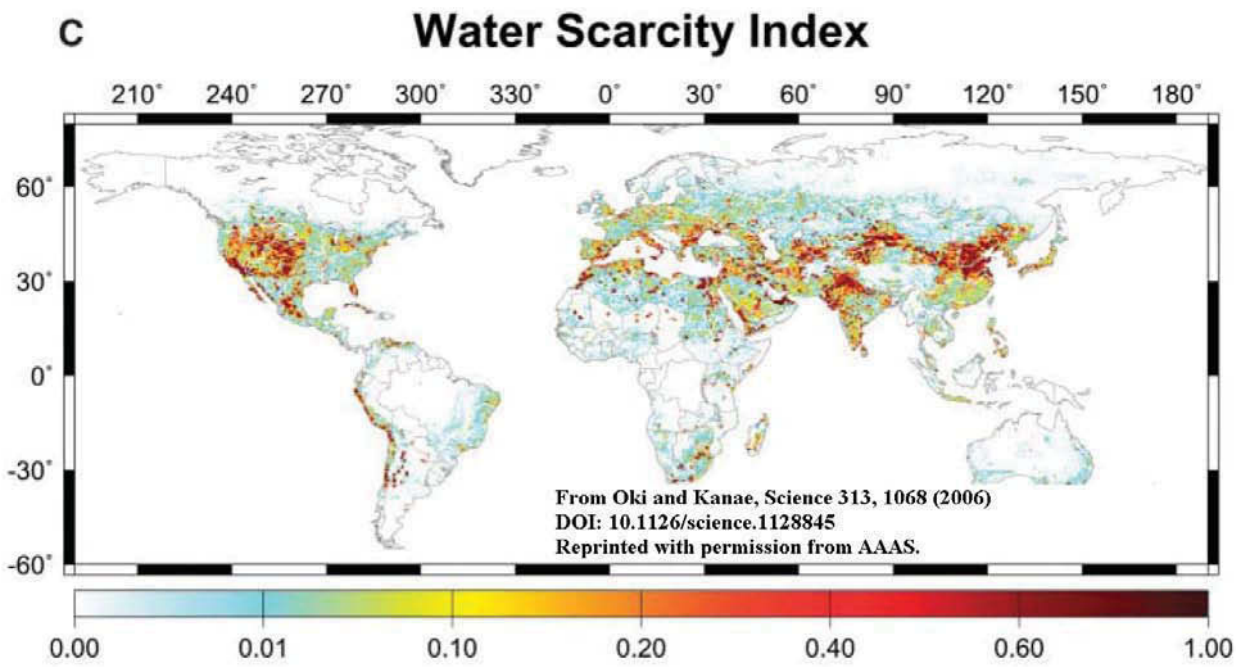

Figure 3 Water scarcity index for the year 2000. Reproduced from Oki and Kanae (2006). The index is calculated as (W-S)/Q, where W is annual human water withdrawal, S is water available from desalination, and Q is the annual renewable freshwater resource (precipitation – evapotranspiration). High values indicate water scarce regions.

Human pressure on blue water resources has negative implications for water quantity and quality. High human water demand leads to the depletion of non-renewable fossil groundwater resources (Gleeson et al., 2012; Zastrow, 2014), associated with decreasing water tables, reductions in river discharge, and vegetation loss (Glazer and Likens, 2012). Human blue water abstraction and alteration of river flows by dam construction have substantial impacts on the global water cycle and freshwater ecosystems. The flow volume and timing in many rivers worldwide is significantly altered by human activity and several large rivers run dry before they reach the sea (Falkenmark and Molden, 2008; Grafton et al., 2012). According to Hoekstra et al. (2012) human water-use exceeds the sustainably allowed level in more than 50% of large river systems globally. Wada et al. (2014) have estimated that humans use 570 km^3yr^{-1} of water from unsustainable sources, constituting around 30% of total consumptive human blue water use. Even if water is returned to the river system after use, chemicals from industrial production and agriculture can have a strong impact on water quality (Hoekstra, 2014). Taken together, these interferences have severe impacts on a variety of ecosystems that provide essential services to humans. Sediment trapping in reservoirs leads to sinking river deltas and endangers the livelihoods of half a billion people worldwide (Syvitski et al., 2009). Across developing regions, 10% of undernourished people depend directly on freshwater ecosystem services and the estimated value of freshwater ecosystem services in Uganda alone exceeds $600 million (UNESCO, 2009). These ecosystem services are threatened by human activity because freshwater biodiversity is rapidly declining (Dudgeon et al., 2006) and the degradation of freshwater ecosystems leads to erosion, impaired water purification capabilities, and the loss of culturally and recreationally important landscapes (Poff et al., 2010; UNESCO, 2009).

In summary, water scarcity in many regions worldwide impairs the livelihoods of millions of people. Human wellbeing is affected by direct water shortages for drinking, sanitation and food production, as well as the degradation of freshwater ecosystems that provide essential ecosystem services. Agricultural blue water use is the most important anthropogenic influence on global water resources when considering quantity and environmental footprint. Data uncertainties about key determinants of the state of global water resources are high.

4 Future prospects: Drivers of agricultural land and water-use

Agricultural land and water-use is influenced by a variety of drivers. These include socioeconomic factors like population growth and economic development, and biophysical factors such as resource availability and climatic conditions (Godfray and Garnett, 2014). Moreover, new sectors will increasingly compete for land and water resources. Most prominently, attempts to mitigate climate change may require land and water for carbon sequestration and bioenergy production (Smith et al., 2013). In the following, the prospects for the main drivers of agricultural land and water-use according to Godfray et al. (2014) are presented.

4.1 Population growth and economic development

Global food demand has almost tripled since 1960 (FAOSTAT, 2013). This increase was on the one hand driven by population growth from around 3 billion people in 1961 to 6.9 billion in 2010 (Figure 4). On the other hand, per capita food demand increased from 2193 kcal cap^{-1} d^{-1} to 2851 kcal cap^{-1} d^{-1} (FAOSTAT, 2013). Income and general economic development are the main drivers of per capita food consumption (Alexandratos et al., 2012).

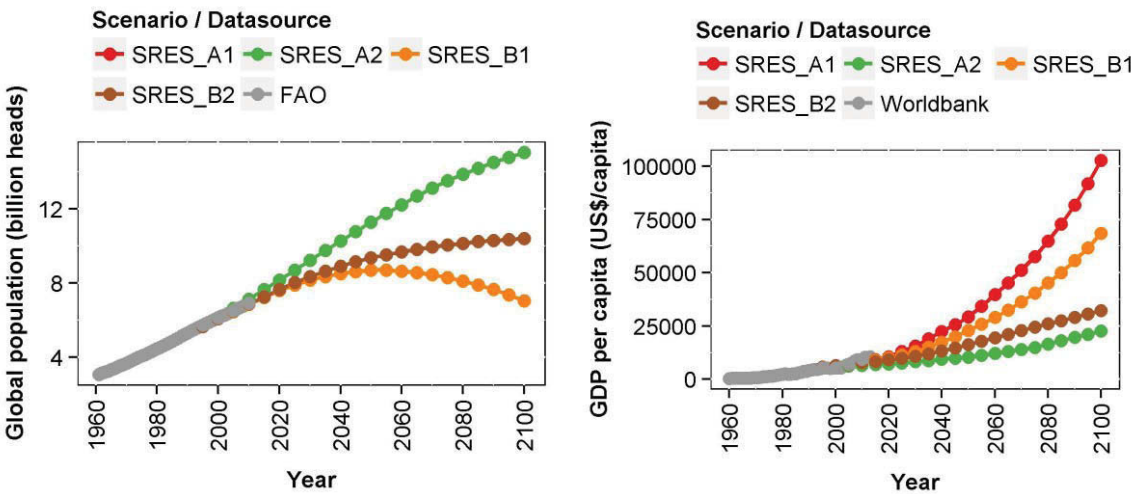

Figure 4 Global population (left) and gross domestic product (GDP, right). Future scenarios from the SRES scenario family (IPCC, 2000). Historical data on population from FAOSTAT (2013). The SRES_A1 population projection is similar to the SRES_B1 projection and is hidden behind the yellow line. Historical data for GDP from the World Bank (2014).

The exact future development of population and the world economy is of course not known. The Intergovernmental Panel on Climate Change (IPCC) has however published population and gross domestic product (GDP) scenarios that aim at spanning a wide range of plausible future developments (IPCC, 2000). While there are substantial differences between the four scenarios (Figure 4), they agree that population and GDP will continue to grow, at least until the middle of the century. This will increase food demand, but also water demand for sanitation and industrial production (Alcamo et al., 2007; Flörke et al., 2013). Thus, future population growth and economic development are likely to put substantial additional pressure on land and water resources for agriculture.

4.2 Climate Change

The last century has seen an increase in global mean temperature by 0.65-1.06 °C (IPCC, 2013). The IPCC's fifth assessment report considers it *extremely likely* that humans contributed to global warming through the emission of greenhouse gases. Anthropogenic climate change is *likely* to further increase the global mean temperature by 0.3-4.8 °C until the end of the century depending on future greenhouse gas emissions (IPCC, 2013). The potential implications for agriculture are twofold. First, local temperature changes may alter the conditions for plant growth and can affect agricultural yields. Second, changes in precipitation patterns and intensities can influence water availability for agriculture. A recent survey of yield projections under climate change until the end of the century finds that almost 80% of the studies project yield decreases up to 100% while 20% of the studies suggest yield increases up to 50% (IPCC, 2014a). Even under the highest emission scenario (RCP8.5) that can lead to temperature increases above 4 °C until the end of the century, global crop models do not agree on the direction of the trend in yields for major food crops (Figure 5) (Rosenzweig et al., 2014). Especially the magnitude of the beneficial effect of higher atmospheric carbon dioxide (CO_2) concentration is a major source of uncertainty.

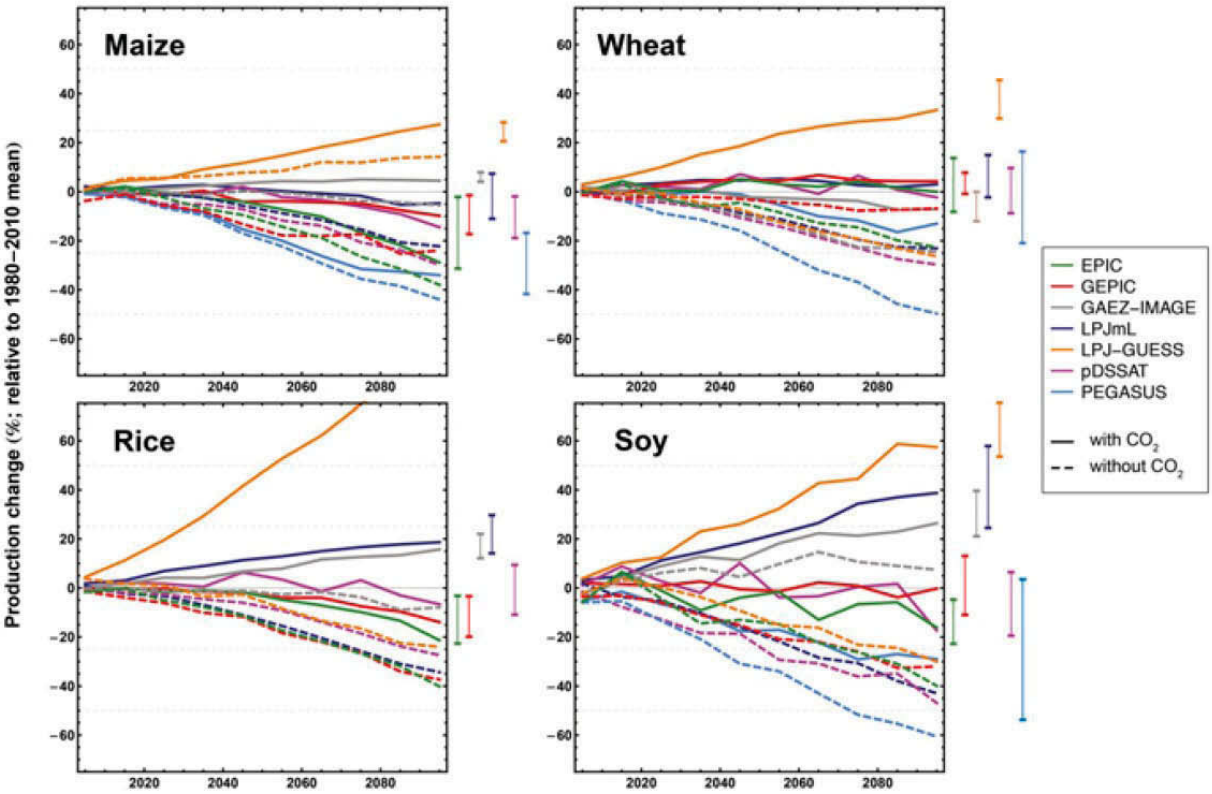

Figure 5 Climate impacts on the production of four important food crops under an RCP8.5 scenario. Lines depict the relative change in decadal mean production for each crop model, based on current agricultural lands and irrigation patterns (median across five different Global Circulation Models). Simulations with CO2 fertilization effect (solid lines) and without CO2 effect (dashed lines). Bars show the range of all Global Circulation Models for each crop model including CO2 fertilization effects. Figure adopted from Rosenzweig et al. (2014) with permission from PNAS.

The influence of climate change on water resource availability is spatially diverse and varies considerably between climate models (IPCC, 2014a). In the Mediterranean and southern Africa, water availability is likely to be reduced while other water stressed areas will see an increase in runoff in the future (IPCC, 2014a). Arnell et al. (2014) have found that climate change impacts on human exposure to water stress are largely masked by climate model uncertainty at the global scale.

In summary, climate change can have substantial impacts on land and water resources for agriculture by changing yield and water availability patterns. The quantification of these impacts is however difficult because of the large associated uncertainties.

4.3 Climate Change mitigation

Mitigating dangerous climate change will require a stabilization of greenhouse gas concentrations in the atmosphere. Anthropogenic greenhouse gas (GHG) emissions mainly comprise carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) (IPCC, 2014b). The most important driver of anthropogenic climate change is CO_2 emissions from fossil fuel combustion and cement production (27.9 – 33 Gt CO2 yr^{-1} in the period 2002 - 2011) (IPCC, 2013). Carbon emissions from land-use change are prone to large uncertainties and add

another 0.4 – 6.2 Gt CO2 yr^{-1} (IPCC, 2013), accounting for 1.1-18.3% of total anthropogenic CO2 emissions. The warming potential of Methane emissions in 2010 was equivalent to 7.8 ± 1.6 Gt CO2 yr^{-1}, and nitrous oxide emissions in 2010 amounted to 3.1 ± 1.9 Gt CO_2 equivalent per year (IPCC, 2014b). The agricultural sector is the most important source of non-CO_2 GHG emissions and accounted for 56% of total non-CO_2 emissions in 2005 (IPCC, 2014b). Following Smith et al. (2013), the main mitigation options in the agricultural sector can be summarized as follows.

Improved management of soils, fertilizer and livestock systems

A reduction of agricultural N_2O emissions can be achieved through better soil and fertilizer management while reducing CH_4 emissions will require a better management of livestock systems with a special focus on enteric fermentation emissions and manure management (IPCC, 2014b; Smith et al., 2013). These measures have limited direct influence on agricultural land and water-use.

Reductions in carbon losses from biota and soils

Significant amounts of carbon are stored in soils and vegetation, especially in forests. Protecting carbon rich land ecosystems can significantly reduce agricultural emissions (Smith et al., 2013). Such measures will require restricting the expansion of agricultural areas into carbon rich natural ecosystems (IPCC, 2014b) and may thus require a significant transformation of agricultural land-use with respect to business as usual conditions.

Enhancement of carbon sequestration in biota and soils

Afforestation and forest restoration aim at increasing the land area under forest cover in order to exploit the high carbon storage potential of forests (Smith et al., 2013). Thus, negative emissions from afforestation and reforestation can decrease the mitigation pressure on other sectors that are difficult and costly to decarbonize, e.g. the transport sector (IPCC, 2014b). Realizing carbon sequestration through afforestation may require large amounts of land and may thus significantly alter agricultural land-use.

Provision of biomass for energy generation

A central challenge for successful mitigation policies will be the substitution of fossil fuels with less carbon intensive energy carriers (IPCC, 2014b). Bioenergy from dedicated energy crops, residues, and forestry products is a promising renewable energy carrier with low carbon intensity (Smith et al., 2013). Bioenergy can be used for heat and electricity production, as well as for liquid fuel production and can thus contribute to decarbonizing the transport sector (Gerbens-Leenes et al., 2009). If combined with carbon capture and storage, bioenergy can even remove CO_2 from the atmosphere (Azar et al., 2006). Large scale production of dedicated bioenergy crops can put substantial additional pressure on land and water resources (Berndes, 2002; Popp et al., 2014).

In summary, climate change mitigation may require a substantial transformation of the agricultural sector. Especially bioenergy production and afforestation may compete for land and water resources.

4.4 Resource scarcity and sustainability concerns

Land and water are finite resources. Resource scarcity may therefore pose an additional challenge to future agriculture. Even if physical resource limits are not binding, sustainability criteria may limit human land and water appropriation. These considerations have led to the proposal of "planetary boundaries" for land, water, and other resources that define the "safe operating space" for humanity (Rockström et al., 2009).

Estimates of the availability of land for cropland expansion vary considerably among different studies (Eitelberg et al., 2014). The lowest estimates indicate that there is almost no room for further cropland expansion while the highest estimates suggest that a threefold increase in global cropland could be feasible (Eitelberg et al., 2014). The large spread between estimates indicates that agricultural land is not primarily restricted by physical resource availability, although physical boundaries may be reached locally. More important, unmanaged land provides a number of important ecosystem services that are threatened by anthropogenic land-use change (see 2 Current state of global land resources). Thus, there have been calls to stop expanding agriculture in order to sustain important natural ecosystems (Foley et al., 2011). Perceived scarcity of land is reflected by increasing land prices and trends towards land acquisition by countries and private investors, also referred to as "land grabbing" (Cotula et al., 2009).

Global renewable water resources available for human use in rivers, lakes and aquifers are estimated at 12500 – 16300 km^3yr^{-1} (Gerten et al., 2013). This is more than twice the amount of current annual human water withdrawals of 3450 – 5230 km^3yr^{-1} (Flörke et al., 2013; Wisser et al., 2008), indicating that water resources are abundant. Water is, however, distributed very heterogeneously around the globe and cannot be transported easily. Therefore, water scarcity is a crucial problem in many places, especially in India, China and the USA (Figure 3) while water resources are abundant elsewhere. Moreover, natural ecosystems and the life contained within them are nowadays widely viewed as a legitimate user of water (Vörösmarty et al., 2013). Sustaining important freshwater ecosystems requires the provision of environmental flows that assure sufficient quantity, quality and timing of water for the environment (Poff et al., 2010). Even if physical water availability does not impose a hard constraint, it can therefore be advisable to restrict human water-use in order to protect environmental flows. Taking into account environmental flows and other sustainability criteria reduces global available water resources from 12500 – 16300 km^3yr^{-1} to 1100 – 5500 km^3yr^{-1} (Gerten et al., 2013). It is estimated that around 30 % of current human water-use originates from unsustainable surface- and groundwater (Wada and Bierkens, 2014).

In summary, physical scarcity of land and water resources may constrain agricultural expansion in many places worldwide. Furthermore, the protection of important land and freshwater ecosystems may require limiting agricultural activity, even below the current state.

5 Research questions

5.1 Main research objective

The background section of this thesis has highlighted that global land and water resources are subject to substantial pressure from human activity, mainly from agriculture. Moreover, it has been discussed that several drivers may alter agricultural land and water-use in the future. Therefore, the design of sustainable management strategies requires information about the future impact of global agriculture on land and water resources. This thesis aims at providing such information by addressing the research question:

How will agriculture affect global land and water resources in the future?

Making accurate predictions about the future evolution of the agricultural system is impossible for two reasons. First, the system is not deterministic: if we would know, how the future will unfold, we would adapt our management strategies accordingly and thus change the expected future status of the system (Schwanitz, 2013). Second, our knowledge about the current status of the system and the underlying dynamics is prone to large uncertainties (see Sections 2 and 3 of the current chapter). Therefore, scenario analyses are performed in Chapters II to V in order to explore the potential influence of different external drivers and management decisions on future agricultural land and water-use. Synthesizing the individual studies allows us to identify the most crucial drivers and to quantify the expected challenges for agriculture associated with different management strategies.

The overarching research objective is therefore addressed by answering six research questions that are presented in the following sections. Research questions one to four are each closely connected to one individual study from Chapters II to V while the last two research questions are addressed by synthesizing the results across all studies.

5.2 What are the land-use implications of policies that aim at reducing emissions from land-use change (Chapter II)?

Avoiding carbon emissions from land-use change is promoted as a cost-effective climate change mitigation measure (Kindermann et al., 2008). Especially policies that reduce emissions from deforestation and forest degradation (REDD) are seen as an important terrestrial mitigation option (Nepstad et al., 2013). Positive side-effects of REDD policies include biodiversity conservation, water regulation, erosion protection, and other ecosystem services that are provided by undisturbed forest ecosystems (Bradshaw et al., 2007; Foley et al., 2007).This thesis explores the potential land-use implications of different

terrestrial carbon emission reduction policies that incentivize emission reductions via a global carbon tax. The focus is on the difference between a REDD policy that only prices emissions from deforestation and a comprehensive terrestrial carbon policy that embraces emissions from all terrestrial systems including pasture and non-forest natural vegetation.

5.3 How may terrestrial carbon sequestration policies influence agricultural land-use (Chapter III)?

The contribution of the land-use sector to climate change mitigation is not limited to emission reductions. Land management strategies can additionally exploit the carbon storage potential of plants and soils to sequester carbon and create negative emissions. There are two main mechanisms for terrestrial carbon sequestration (Tavoni and Socolow, 2013): afforestation and bioenergy with carbon capture and storage (BECCS). Afforestation is a low-tech strategy where trees are planted for carbon sequestration. BECCS on the other hand relies on highly advanced technologies to capture the carbon during the conversion of bioenergy crops to final energy and store it underground. This thesis investigates the land-use implications of afforestation and BECCS individually and in a combined setting. Thereby, it helps to improve our understanding of the land-use challenges and potential environmental implications associated with terrestrial carbon sequestration.

5.4 Do we have to expect a conflict between sustainable water management strategies and sustainable land management strategies for agriculture (Chapter IV)?

Current levels of human water-use lead to widespread degradation of ecosystems and endanger associated livelihoods (Dudgeon et al., 2006; Syvitski et al., 2009). Therefore, environmental flow protection policies are proposed that reserve sufficient amounts of water to sustain freshwater ecosystems (Pahl-Wostl et al., 2013; Smakhtin et al., 2004). At the same time, the continuing decline of pristine land ecosystems has led to the call for halting agricultural expansion (Foley et al., 2011). Potential conflicts between these two management goals arise from the substitutability of land and water resources in food production, but have not been studied at the global scale so far. This thesis provides a first quantitative estimate of the potential implications of environmental flow protection policies on global agricultural land and water-use. In doing so, it advances the understanding of the land-water nexus and provides a new perspective on the sustainability of water management strategies.

5.5 How may large-scale bioenergy production affect global land and water resources (Chapter V)?

Bioenergy can replace fossil fuels in the transport sector and remove carbon dioxide from the atmosphere if combined with carbon capture and storage technologies (Creutzig et al., 2014). Therefore, integrated assessment modelling studies project that bioenergy may provide up to 50% of the future primary energy demand (Rose et al., 2014). Such high bioenergy shares will require large-scale production of 2nd generation bioenergy crops that

may require significant amounts of land and irrigation water. Chapter V of this thesis investigates two large-scale bioenergy production scenarios in order to explore associated land and water requirements and potential trade-offs. In the first scenario, bioenergy crops can be produced in rainfed and irrigated production systems while in the second scenario, irrigated bioenergy production is prohibited. Thus, this analysis improves our understanding of the water requirements for large-scale irrigated bioenergy production and provides insights into the feasibility and land-use implications of water saving bioenergy strategies.

5.6 How do different agricultural strategies compare in terms of environmental implications for land and water resources and required transformation of the system (Synthesis across Chapters II to V)?

One important step towards sustainable strategies for agriculture is an analysis of the environmental implications of different management options. Another important aspect is the societal challenges of agricultural management strategies. Among others, these can include concerns about food security and sovereignty (Fader et al., 2013), distributional effects due to shifts in comparative advantages (Phan, 1988), and the necessity of implementing new legal or regulatory frameworks (Pahl-Wostl et al., 2013). Investigating societal challenges of agricultural policies in detail is highly complex and is therefore beyond the scope of this thesis. However, the transformation of the agricultural system that is required to achieve a certain management goal can provide an idea about the societal challenges that we have to expect. A strategy that requires a fundamental transformation of the system is likely to be more difficult to achieve than a strategy that requires only minor adjustments. Moreover, strong transformations of agriculture increase the risk of unexpected side-effects that might compromise the provision of essential services, e.g. food security. In Chapter VI of this thesis, the individual results from Chapters II to V are synthesized with a focus on the environmental implications for global land and water resources and on the required transformation of the agricultural system. Thus, this thesis provides information on how desirable different strategies for terrestrial climate change mitigation and sustainable water management are from an environmental perspective and how challenging the implementation may be.

5.7 How large is the operating space for land and water management strategies in agriculture (Synthesis across Chapters II to V)?

It is impossible to accurately predict the future of agricultural land and water-use. Moreover, based on today's knowledge, it is not even possible to foresee all potential future challenges and to come up with strategies to address them (Stirling, 1994). In this context of high uncertainty, it is a crucial question, how large the operating space for agricultural land and water management strategies is. Is agricultural resource use tightly constrained by biophysical constraints and inevitable socioeconomic processes? Or are different future pathways feasible that allow for the adaptation of management strategies to unforeseen challenges? This thesis contributes to answering this question by exploring the range of potential future agricultural land and water-use across all scenarios from Chapters II to V.

Thereby, it provides information about the potential operating space for agricultural land and water management strategies and about the adaptive capacity of the system to unforeseen challenges.

6 Methodology

In order to tackle the research questions that were outlined in the previous section, quantitative scenarios of agricultural land and water-use in the future are developed and evaluated. The scenario analysis relies on a computer model approach that captures important dynamics of the agricultural system, including socioeconomic drivers such as population growth and economic development, as well as biophysical constraints such as resource availability and agricultural yields. This setup can be viewed as an "integrated modelling framework" (Sohl and Claggett, 2013) consisting of a modularized mathematical representation of individual processes that affect agricultural land and water-use. It makes use of high performance computers to link individual modules in real time during the simulation process. The advantage of this approach is the ability to integrate various drivers of agricultural land and water-use and their complex, non-linear interactions in a consistent modelling framework.

6.1 Model description

This analysis makes use of the Model of Agricultural Production and its Impacts on the Environment (MAgPIE) (Bodirsky et al., 2012; Dietrich et al., 2013a, 2013b; Krause et al., 2013; Lotze-Campen et al., 2010, 2008; Popp et al., 2012, 2011a, 2011b, 2010; Schmitz et al., 2012). A mathematical description of the core model can be found in Bodirsky et al. (2012) and the associated supplementary information (http://www.biogeosciences.net/9/4169/2012/bg-9-4169-2012-supplement.zip).

The goal of MAgPIE is to fulfil demand for agricultural products at minimum costs. Demand for agricultural products at the level of 10 world regions is estimated exogenously via econometric analysis considering population growth, economic development and dietary preferences (Bodirsky et al., under review, 2012). The production function takes land, water, yield information and monetary inputs into account. A distinctive feature of MAgPIE is the endogenous simulation of future yield improvements through technological progress (Dietrich et al., 2013b). Spatial allocation of production to individual simulation units is based on comparative advantages under biophysical constraints (land and water availability), rotational constraints (Lotze-Campen et al., 2008) and trade constraints (Schmitz et al., 2012). MAgPIE simulations in this analysis are performed on a spatial level of 500 (Chapter III), 600 (Chapter II), and 1000 (Chapters IV, V) simulation units respectively.

For the analyses conducted in this thesis, the water representation within MAgPIE has been refined with respect to water availability, irrigation costs, and non-agricultural water demand (Chapter IV).

In order to simulate the effects of terrestrial climate change mitigation policies, MAgPIE contains modules for estimating carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) emissions as well as a module that allows for pricing these emissions in global carbon tax regimes. CO_2 emissions are calculated by combining simulated information about land-use changes with carbon densities (Popp et al., 2012). The calculation of N_2O and CH_4 emissions involves combining simulated information on agricultural activity with emission factors derived from literature (Bodirsky et al., 2012; Popp et al., 2010).

MAgPIE can simulate scenarios with flexible time steps from 1995 to 2100 using recursive dynamic optimization. The optimization problem is solved for each time period separately and the optimization results of the previous period serve as a starting point for the optimization in the following time period. In contrast to a perfect foresight approach, the future consequences of decisions are not known within the optimization and expectations have to be approximated from the information available in the current time step and a predefined time horizon of 30 years.

MAgPIE is closely linked to other models that are being developed at the Potsdam Institute for Climate Impact Research. Spatially explicit biophysical information including crop yields, water availability, and carbon densities are obtained from the LPJmL model (Bondeau et al., 2007; Müller and Robertson, 2013). On the socioeconomic side, MAgPIE receives scenarios for bioenergy demand and greenhouse gas prices from the energy-economy model REMIND (Bauer et al., 2012; Klein et al., 2014; Leimbach et al., 2010; Luderer et al., 2013).

6.2 Model testing and validation

Testing, whether the applied methodology is adequate for the specific research question under consideration is an integral part of any scientific assessment (Jakeman et al., 2006). If a study, such as this thesis, relies on computer models, the question of adequate methodology is often referred to as model validity (Barlas, 1996). Several methodologies and philosophical paradigms exist for testing model validity (Barlas and Carpenter, 1990). Traditionally, model results are compared to real system behaviour. In the case of discrepancies, the model is “proven” wrong and adjustments need to be made before it can be used. Consequently, positive verification of a model is impossible, but the credibility increases with the number of tests that have been successfully passed. In the context of scenario analyses with the MAgPIE model, this approach is unfortunately not applicable since it is impossible to obtain real world data about the future for comparison (Schwanitz, 2013). This is a common issue for land-use models and – up to now – there has been no agreement in the scientific community about a validation concept for model based land and water-use projections (Sohl and Claggett, 2013). Literature about the subject emphasizes that model validity in the context of unknown and unpredictable behaviour of the real system corresponds to a gradual process of building confidence in a model via a variety of tests and measures (Barlas, 1996; Jakeman et al., 2006; Schwanitz, 2013). The credibility of the analyses presented in this thesis mainly rests on three pillars that are often demanded

in proposals for model validation concepts (Barlas and Carpenter, 1990; Jakeman et al., 2006; Schwanitz, 2013; Sohl and Claggett, 2013):

Documentation of important assumptions and limitations

A clear documentation of important model assumptions and known limitations enables the reader to judge on his/her own if the methodology is adequate for the purpose and what interpretation of the results is appropriate. In this thesis, the underlying assumptions of each individual study are documented and discussed. In Chapters IV and V, a dedicated section "Assumptions and limitations" is devoted to the task of highlighting the most important assumptions and associated limitations.

Comparison of model results to historical data

Even though no real-world data for comparison exists for the future, model results in the historical period (1995 – 2010) can be compared to historical data. While this does not directly imply that future scenarios are plausible, it helps in building confidence in the model's ability to reproduce key features exhibited by the real system. Within this thesis, comparison to historical data is pursued throughout the individual studies and considers important simulation results such as cropland, yield evolution and agricultural water demand.

Sensitivity analysis with respect to crucial model parameters

Complex models such as MAgPIE rely on a large number of parameters that are often associated with substantial uncertainties. Ideally, the influence of all uncertain parameters should be tested in order to arrive at an uncertainty estimate of the simulation results. This is, unfortunately, impossible due to the large number of parameters and a lack of uncertainty information for individual parameters. Still, the sensitivity of model results with respect to variations in important parameters can be tested in order to provide a lower estimate of the output uncertainty. Within this thesis, sensitivity analyses are conducted throughout the individual studies and their implications for the interpretation of results are discussed.

7 Structure of the dissertation

Apart from the current chapter, the Introduction, this thesis contains five chapters. Chapters II to IV contain one article, published in a peer reviewed scientific journal, each. The effort of publishing the articles in Chapters II to V has been undertaken in close collaboration with colleagues from the Potsdam Institute for Climate Impact Research. The author's contribution to each of the articles is documented in the *Statement of Contribution* section at the end of this thesis. Each of the articles forms a self-contained chapter featuring its individual layout and references.

Chapter II compares the land-use implications of a REDD policy that applies a global carbon tax on emissions from deforestation to a comprehensive terrestrial mitigation policy that embraces emissions from all land conversion activities.

Chapter III investigates afforestation and bioenergy in combination with carbon capture and storage as terrestrial carbon sequestration options with a special focus on land-use implications.

Chapter IV provides insights about the consequences of environmental flow protection policies that aim at preventing freshwater ecosystem degradation. Focusing on land-use changes associated with such scenarios allows to gain insights about potential trade-offs between freshwater and terrestrial ecosystems.

Chapter V extends the trade-off analysis between water and land for agricultural production by considering large-scale bioenergy production as a driver of agricultural activity.

Chapter VI summarizes the key findings of this thesis and provides a synthesis across the results according to the research questions that were outlined in Section 5 of the current chapter. Furthermore, an outlook for further research opportunities is presented.

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Chapter II

Land-use protection for climate change mitigation *

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* published in Nature Climate Change Volume 4, Issue 12, Pages 1095-1098, December 2014, DOI: 10.1038/nclimate2444

Content of Chapter II: Land-use protection for climate change mitigation

nature climate change

LETTERS

PUBLISHED ONLINE: 17 NOVEMBER 2014 | DOI: 10.1038/NCLIMATE2444

Land-use protection for climate change mitigation

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Land-use change, mainly the conversion of tropical forests to agricultural land, is a massive source of carbon emissions and contributes substantially to global warming[1-3]. Therefore, mechanisms that aim to reduce carbon emissions from deforestation are widely discussed. A central challenge is the avoidance of international carbon leakage if forest conservation is not implemented globally[4]. Here, we show that forest conservation schemes, even if implemented globally, could lead to another type of carbon leakage by driving cropland expansion in non-forested areas that are not subject to forest conservation schemes (non-forest leakage). These areas have a smaller, but still considerable potential to store carbon[5,6]. We show that a global forest policy could reduce carbon emissions by 77 Gt CO_2, but would still allow for decreases in carbon stocks of non-forest land by 96 Gt CO_2 until 2100 due to non-forest leakage effects. Furthermore, abandonment of agricultural land and associated carbon uptake through vegetation regrowth is hampered. Effective mitigation measures thus require financing structures and conservation investments that cover the full range of carbon-rich ecosystems. However, our analysis indicates that greater agricultural productivity increases would be needed to compensate for such restrictions on agricultural expansion.

Driven mainly by the fertilizing effects of increased levels of CO_2 in the atmosphere, the land system has been a terrestrial sink for carbon in recent decades[2]. However, the role of land for sequestering carbon is counteracted, as the carbon emissions from land-use and land-cover change accounted for approximately 12% of all anthropogenic carbon emissions from 1990 to 2010[3]. The future development of forest area is uncertain, but deforestation is projected to persist as a significant emission source in the absence of new forest conservation policies, especially under increasing demand for agricultural commodities. Compared to climate change mitigation options in the energy and transport sector, recent research has indicated low opportunity costs and significant near-term mitigation potential through reducing deforestation, promoting avoided deforestation in tropical countries as a cost-effective mitigation option[7].

Despite the general scientific agreement on environmental benefits of forest conservation, and although the United Nations Framework Convention on Climate Change (UNFCCC) has affirmed the potential role of forests in stabilizing the global climate, no global action has yet emerged to conserve natural forests. Several issues have so far prevented the development of conservation mechanisms supported under the UNFCCC (ref. 8). In particular, the design of financing mechanisms[4], but also environmental and socio-political concerns associated with REDD (Reduced Emissions from Deforestation and Degradation) and its variations are being intensively discussed[9,10]. One key issue for the implementation of REDD is how to address leakage of emissions[11]. Without full participation of all countries in a forest conservation scheme, emission reductions in one location could result in increased emissions elsewhere, as agricultural expansion, the main driver for deforestation, could just be displaced rather than avoided[12].

However, carbon leakage is not only relevant in the context of regionalized forest protection efforts. Another risk associated with a global REDD scheme that so far has not been quantified in the literature is the shift of land-use pressures to non-forest ecosystems (non-forest leakage) simply because they are the only remaining resource for agricultural expansion[13]. Such ecosystems may also be rich in carbon. First, areas under natural vegetation other than forests, such as shrublands and savannas, can also store considerable amounts of aboveground carbon, especially in Africa, but also in Latin America and Asia[6]. Second, carbon-rich soils also play a major part in the terrestrial carbon balance and have to be taken into consideration[5,14]. Grasslands and pastures, unlike cropland, maintain a permanent vegetation cover and, therefore, have a high root turnover, leading to substantial soil organic carbon storage[15]. For this reason, carbon stocks decline strongly after land is converted from grasslands and pastures to cropland[5]. Finally, agricultural activity can reduce carbon sequestration by preventing regrowth of natural vegetation on abandoned agricultural land[16].

In contrast to the current political discussion, which focuses only on REDD implementation, recent global modelling assessments have focused on the implementation of a global terrestrial carbon policy covering all regions and land types[17,18]. To avoid the negative consequences of a global forest conservation policy, a profound understanding of potential implementation failures, such as leakage into land types other than forests, is needed.

Here, we estimate land-use and associated carbon dynamics for different global terrestrial carbon policies at global and regional scale using the land-use optimization model MAgPIE (Model of Agricultural Production and its Impacts on the Environment—see Methods)[19]. Biophysical inputs for MAgPIE, such as agricultural yields, carbon densities and water availability, are derived from a dynamic global vegetation, hydrology and crop growth model, the Lund–Potsdam–Jena model for managed Land (LPJmL; refs 20,21). LPJmL provides the climate- and CO_2-driven changes in carbon densities, agricultural productivity and water availability of a 2 °C scenario (RCP2.6) to drive MAgPIE simulations. For this study, we assume ambitious mitigation policies with different contributions of the land-use sector in three scenarios: no terrestrial carbon policy in the reference scenario (Ref); a global terrestrial land-use policy that considers carbon emissions from deforestation

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LETTERS NATURE CLIMATE CHANGE DOI: 10.1038/NCLIMATE2444

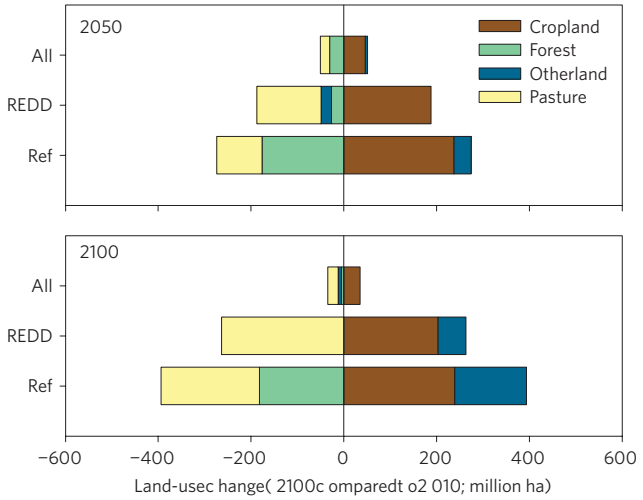

Figure 1 | Change in global land pools. The upper figure shows changes from 2010 to 2050 and the lower figure changes from 2010 to 2100 for the reference case (Ref) without land-use mitigation, a terrestrial land-use policy that considers carbon emissions from deforestation only (REDD) and a terrestrial carbon policy that accounts for emissions from all land types (All).

only in the REDD scenario; a global terrestrial carbon policy introduced by a universal carbon tax on greenhouse gas emissions from all terrestrial systems in the All scenario. To account for uncertainty in climate projections, we compute changes in carbon densities, agricultural productivity and water availability for the implementation of the RCP2.6 scenario in five different global circulation models (GCMs). We generally report mean values across all GCMs, while single GCM outputs and standard deviations can be found in Supplementary Table 1. In addition to the default scenarios with different GCM inputs, we perform sensitivity analyses with crucial exogenous parameters (demand for agricultural products, costs for agricultural yield increases and tax on terrestrial carbon emissions) to test the stability of our results in terms of cumulative carbon emissions (see sensitivity analysis in the Supplementary Information). It is important to note that the land-use model not only embraces the calculation of emissions from deforestation and other land-use change, but also the uptake of carbon from regrowth of secondary natural vegetation on abandoned agricultural land and carbon dynamics driven by climate change and CO_2 fertilization. In contrast to the mitigation of carbon emissions from land-use change, carbon uptake is not rewarded financially in our scenarios, as we focus in this study on protection policies. The MAgPIE model has been validated intensively for land-use, agricultural yield and land carbon dynamics and reproduces historical trends well (see also the validation section in the Supplementary Information). In addition, the ability of LPJmL to simulate global terrestrial carbon dynamics has been demonstrated in several previous studies[21,22].

Our reference scenario (Ref) without any terrestrial carbon policy is parameterized according to the 'SSP2' storyline of the shared socio-economic pathways[23] (see more detail in Methods). Our model results show that agricultural production increases are mainly realized by intensification on existing agricultural land (Supplementary Fig. 1) as well as by agricultural land expansion. In 2010, global cropland area was 1,454 million ha, pasture land area 3,079 million ha, global forest area 4,144 million ha and global other land area 4,229 million ha (see also Supplementary Fig. 2). At the global level, cropland increases by 237 million ha until the year 2050 and by 239 million ha until 2100, compared to 2010 (Fig. 1). Cropland area expands in developing and emerging regions, including countries of the Middle East and Africa (MAF), countries of Latin America and the Caribbean (LAM) and Asian countries, with the exception of the Middle East, Japan and Former Soviet Union states (ASIA), whereas it decreases in OECD90 countries (OECD; Supplementary Fig. 3). As a consequence, agricultural land is abandoned in the developed regions, as well as in LAM and MAF, where less pasture land is needed owing to more intensified livestock production systems that require less roughage for ruminant feed. Therefore, abandoned land increases by 154 million ha globally until 2100. According to this scenario, global land-use change emissions accumulate to 173 Gt CO_2 over the twenty-first century (Fig. 2a). Because of regrowth of secondary natural vegetation, 84 Gt CO_2 is sequestered on abandoned agricultural land up to 2100 (Fig. 2b).

Subsequently, we estimate the impacts of two different terrestrial land-use policies on land-use and carbon dynamics. Consistent with previous findings[17], a global terrestrial carbon policy (All scenario), introduced by a universal carbon tax on greenhouse gas emissions from all terrestrial systems, halts land-use change and associated carbon emissions, but decreases carbon uptake from regrowth on abandoned land (29 Gt CO_2 until 2100). However, if a terrestrial land-use policy considers carbon emissions from deforestation only (REDD scenario), forest loss is stopped whereas cropland expansion is reduced only marginally (cropland expansion of 203 million ha until 2100) compared to the Ref scenario (239 million ha) without any land-use policy. Such a policy restricts the areas available for cropland expansion, forcing agricultural expansion to switch to less suitable land. This also incentivizes intensification of existing croplands, leading to improved agricultural management and higher investments in yield-increasing technology (Supplementary Fig. 1). Under the REDD scenario, additional pasture land of 51 million ha is lost until 2100 compared to the Ref scenario, mainly in Africa and Latin America. At the same time, abandoned agricultural land area is reduced by 94 million ha compared to the Ref scenario. The reason is that less agricultural land is abandoned in Africa and Latin America if production cannot be extended into forested areas, and more land with non-forest natural vegetation is lost in Asia and Africa. Under the REDD scenario, carbon emissions from land-use change accumulate to 96 Gt CO_2, which is approximately 55% of the land-use-change-related emissions in the Ref scenario without any land-based mitigation. In addition, less agricultural land is taken out of production, thereby decreasing the uptake potential of secondary natural vegetation regrowth on abandoned land to 55 Gt CO_2.

Climate impacts such as precipitation and temperature changes and CO_2 fertilization based on RCP2.6 affect the carbon dynamics of the terrestrial system in all scenarios. Globally, carbon uptake due to climate change and CO_2 fertilization of 178 Gt CO_2, 176 Gt CO_2 and 180 Gt CO_2 can be attributed to the Ref, REDD and All scenarios, respectively, until 2100 (Fig. 2c). In all scenarios, highest carbon uptake driven by climate change and CO_2 fertilization can be observed until the mid-century as RCP2.6 peaks at 490 ppm CO_2 and then declines[24]. As a consequence of land-use change and carbon uptake, we conclude that the land system could contribute most to climate change mitigation if all ecosystems were to be included in a terrestrial land-use policy (All), taking up 191 Gt CO_2 until 2100 (Fig. 2d). In comparison, if only forest conservation measures are considered (REDD), the carbon uptake would be 55 Gt CO_2 lower compared to All, mainly owing to leakage effects into non-forest ecosystems and associated carbon emissions. Lowest net carbon uptake of 88 Gt CO_2 can be observed in the reference scenario without any land-use policy (Ref).

Our study shows that until 2050, without any land-use policy (Ref), land-use change would contribute about 13% to the global budget of 1,000 Gt CO_2 that must not be exceeded if global warming is to be limited to 2 °C with 66% likelihood[25], and about 7% if forest conservation measures are considered (REDD).

The results of our study emphasize that land-use policies should cover all land types to avoid non-forest leakage effects. Beyond the

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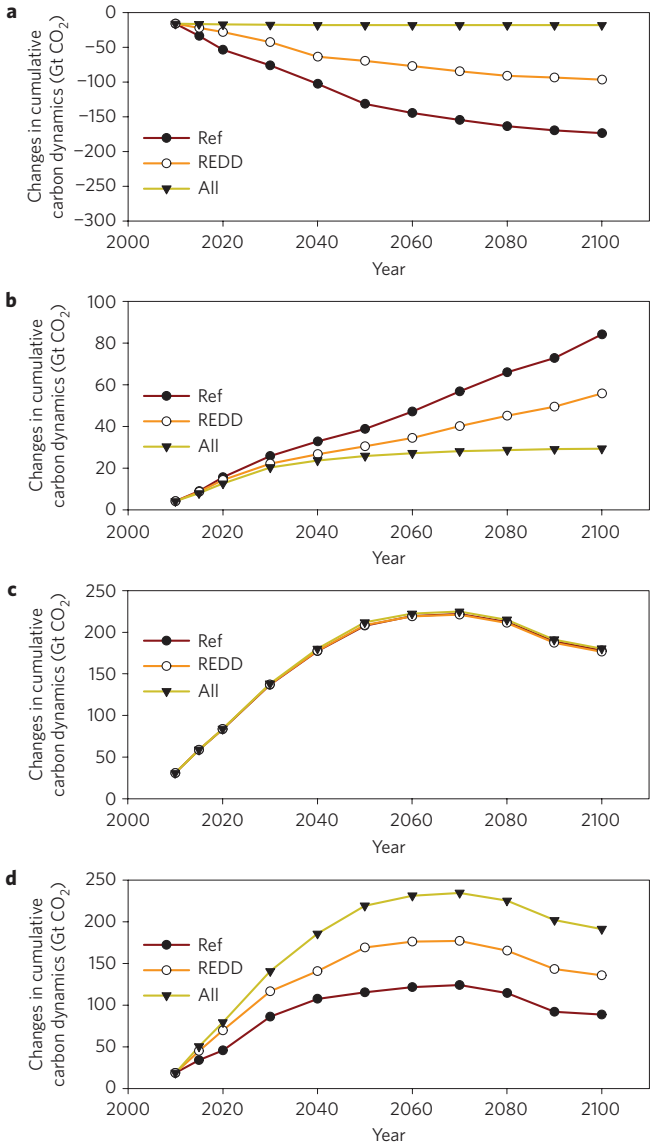

Figure 2 | Cumulative global carbon dynamics over the twenty-first century. a–d, Mean changes in carbon dynamics are calculated for all scenarios and across five GCMs for carbon losses due to land-use change (**a**), carbon uptake due to regrowth of secondary natural vegetation on abandoned agricultural land (**b**), carbon dynamics driven by climate change and CO_2 fertilization under RCP2.6 (**c**), and net carbon dynamics (**d**). Positive values represent terrestrial carbon sequestration, whereas negative values indicate loss of terrestrial carbon to the atmosphere.

importance of controlling land-use dynamics for climate change mitigation, which were analysed here, such policies should also account for other environmental assets, such as biodiversity. Land-use policies provide a huge opportunity to protect biodiversity as a co-benefit of maintaining forests[26]. But, as our analysis shows, forest protection policies such as REDD can lead to displacement of pressures, resulting from increasing demand for agricultural products, to less productive, non-forest ecosystems perceived to contain lower carbon levels. Those ecosystems, such as the tropical savannas of the Brazilian Cerrado, that nevertheless can support great levels of biodiversity or are home to endemic species of high conservational value can become increasingly threatened under such incomplete policies[13,27,28].

Implementing a global terrestrial carbon policy that includes all land types would have the largest benefits for both climate change mitigation and the protection of pristine landscapes. However, the implementation of such a scheme may be regarded as optimistic, given the slow progress in recent international negotiations. If a land-use policy that embraces all land types is considered politically impossible to implement, a simpler and more easily achievable approach to minimize the risks of any forest conservation scheme would be to identify and protect non-forest ecosystems of high value for carbon and biodiversity. So, if a forest conservation mechanism comes into operation, financing structures would have to be implemented which ensure that conservation investment is spread over the range of ecosystems not covered by REDD funding[13].

Our analysis indicates that higher agricultural productivity increases would be needed to compensate for reduced land availability for agricultural use (Supplementary Fig. 1). Generally, preserving ecosystems while enhancing agricultural production is a central challenge for sustainability[11]. Restrictions to agricultural expansion due to land conservation may affect land-use competition, with substantial effects on agricultural production costs and food prices[17,29,30]. And even if REDD is currently seen as a low-cost climate mitigation option, additional costs for the implementation and verification of REDD projects[7], as well impacts on downstream economic values of current land uses, including employment and wealth generated by processing and service industries[9], could occur. These possible impacts need to be balanced against positive effects on CO_2 reductions. More efficient land management and major technological innovations in agriculture have the potential to prevent a global shortage of productive land[29], decrease carbon emissions from land-use change and enhance uptake of carbon from regrowth of secondary natural vegetation on abandoned agricultural land (see sensitivity analysis in the Supplementary Information). Large production increases are possible from, for example, closing yield gaps, but they will require considerable changes in nutrient and water management as well as shifting productivity frontiers to meet sustainability challenges[31]. On the other hand, demand-side measures such as changes in diet towards less products of animal origin can have 'land sparing' effects[32] which reduce the pressure from agricultural expansion on forests and other land (see Supplementary Fig. 4 and sensitivity analysis in the Supplementary Information). In contrast to such processes helping to reduce land-use pressure, enhanced competition in the land system could emerge due to financial rewards for the regrowth of natural vegetation (afforestation), mainly at the expense of pasture areas[33].

Methods

MAgPIE is a mathematical programming model projecting spatially explicit land-use dynamics in ten-year time steps until 2100 using recursive dynamic optimization[19]. The objective function of MAgPIE is the fulfilment of exogenously calculated food and livestock demand, defined for ten world regions (Supplementary Fig. 9 and Table 3), at minimum costs under socio-economic and biophysical constraints. Major cost types in MAgPIE are factor requirement costs (capital, labour, fertilizer and other inputs), land conversion costs, transportation costs to the closest market, investment costs for yield-increasing technological change and costs for carbon emission rights[29,34]. Whereas socio-economic constraints such as trade liberalization and forest protection are defined at the ten-region scale, biophysical constraints such as crop and pasture yields, carbon density and water availability, derived from the dynamic global vegetation model LPJmL (refs 20,21), as well as land availability, are introduced at the grid-cell level (0.5° longitude/latitude). The cost-minimization problem is solved through endogenous variation of spatial production patterns (intra-regionally and inter-regionally through international trade), land expansion and yield-increasing technological change (TC).

MAgPIE features land-use competition based on cost-effectiveness between food and livestock production and land-use-based mitigation such as avoided deforestation. Available land types are cropland, pasture, forest and other land (for example, non-forest natural vegetation, abandoned land, desert). Grid-cell-specific carbon densities for the different carbon stocks (vegetation, soil, litter) of the various land types are based on LPJmL simulations and IPCC guidelines for National Greenhouse Gas Inventories (IPCC 2006). MAgPIE

calculates carbon emissions as the difference in carbon stocks (vegetation, litter and soil) between simulated time steps (more information in the Supplementary Information). Carbon uptake in MAgPIE occurs if regrowth of natural vegetation takes place on abandoned agricultural land (more information in the Supplementary Information). Mitigation of carbon emissions is stimulated by an exogenous tax on terrestrial carbon emissions. The carbon tax is multiplied by carbon emissions to calculate carbon emission costs, which enter the cost-minimizing objective function of MAgPIE. Therefore, stopping land-use change is an economic decision when emissions from land-use change are priced. In contrast, carbon uptake due to regrowth of natural vegetation is not rewarded financially in MAgPIE.

Our socio-economic assumptions are based on the Shared Socio-economic Pathways (SSPs) for climate change research[23]. In this study we choose SSP 2, a 'Middle of the Road' scenario with intermediate socio-economic challenges for adaptation and mitigation. Food, livestock and material demand is calculated using the methodology described in ref.35 and the SSP 2 population and gross domestic product projections (~65 EJ yr^{-1} in 2100, Supplementary Fig. 4). The SSPs do not incorporate climate mitigation policies by definition. Carbon tax (~US$1,500 per tonne of CO_2 in 2100, Supplementary Fig. 5) in our study is aimed at ambitious climate change mitigation (~RCP 2.6 in 2100). The carbon tax has a level of US$30 per tonne of CO_2 in 2020, starts in 2015 and increases nonlinearly at a rate of 5% per year. For consistency, MAgPIE simulations include temperature, precipitation and CO_2 trends and corresponding impacts on agricultural yields, water availability and carbon stocks in vegetation under a RCP2.6, derived by LPJmL. To account for uncertainty in climate projections for RCP 2.6, in this study we use climate data of the five GCMs: HadGEM2-ES, IPSL-CM5A-LR, MIROC-ESM-CHEM, GFDL-ESM2M and NorESM1-M.

Received 10 April 2014; accepted 21 October 2014; published online 17 November 2014

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Acknowledgements

The research leading to these results has received funding from the European Union's Seventh Framework Program under grant agreement no. 282846 (LIMITS), no. 265104 (VOLANTE) and no. 603542 (LUC4C). Funding from Deutsche Forschungsgemeinschaft (DFG) in the SPP ED 178/3-1 (CEMICS) is gratefully acknowledged.

Author contributions

A.P. designed the overall study; F.H. and M.B. carried out the MAgPIE model runs. A.P. wrote the manuscript with important contributions from F.H., B.L.B., C.M. and M.B.; A.P., F.H., M.B. and B.L.B. analysed the results; F.H., I.W., B.L.B., M.B., J.P.D., A.P., M.S., A.B. and H.L-C. contributed in developing and improving the MAgPIE model; C.M. and S.R. provided biophysical input data from LPJmL; all authors discussed and commented on the manuscript.

Additional information

Supplementary information is available in the online version of the paper. Reprints and permissions information is available online at www.nature.com/reprints. Correspondence and requests for materials should be addressed to A.P.

Competing financial interests

The authors declare no competing financial interests.

nature climate change

SUPPLEMENTARY INFORMATION

DOI: 10.1038/NCLIMATE2444

Land-use protection for climate change mitigation

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1. Additional results

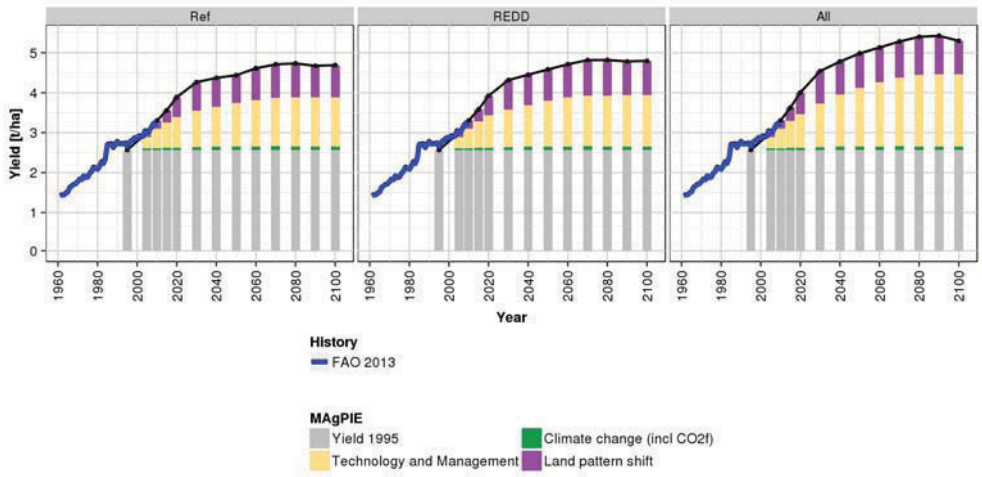

Figure SI-1: Global required agricultural yields [t/ha] aggregated across all crops for all 3 scenarios. The blue line shows historical data for agricultural yields (FAO 2013). The black line shows simulated global agricultural yields in MAgPIE. Grey bars represent agricultural yields in 1995. The colored bars indicate the contribution of different drivers to agricultural yield dynamics: climate change including CO_2 fertilization effects (green), improved agricultural management and technological change (yellow) and shifts in agricultural land use patterns due to e.g. increased agricultural trade or shifting cultivation mainly towards more high yielding feed crops to fulfill increasing demand for livestock products (purple).

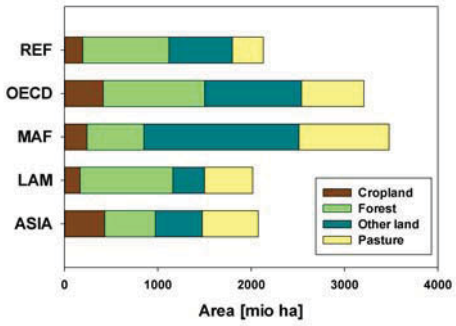

Figure SI-2: Regional land pools [mio ha] for the year 2100. Results are shown are for five aggregate regions: (1) OECD90 countries (OECD), (2) reforming economies of Eastern Europe and the Former Soviet Union (REF), (3) countries of the Middle East and Africa (MAF), (4) countries of Latin America and the Caribbean (LAM) and (5) Asian countries with the exception of the Middle East, Japan and Former Soviet Union states (ASIA. Brown bars represent cropland, yellow bars forest, green bars other land and blue bars pasture.

SUPPLEMENTARY INFORMATION DOI: 10.1038/NCLIMATE2444

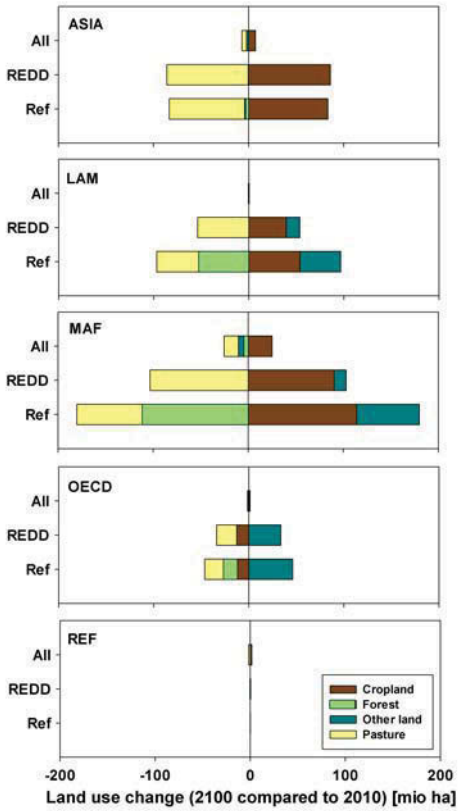

Figure SI-3: Change in regional land pools in million ha from 2010 to 2100 for the reference case (Ref) without land use mitigation, a terrestrial land use policy that considers carbon emissions from deforestation only (REDD) and a terrestrial carbon policy that accounts for emissions from all land types (All). Results are shown are for five aggregate regions: (1) OECD90 countries (OECD), (2) reforming economies of Eastern Europe and the Former Soviet Union (REF), (3) countries of the Middle East and Africa (MAF), (4) countries of Latin America and the Caribbean (LAM) and (5) Asian countries with the exception of the Middle East, Japan and Former Soviet Union states (ASIA).

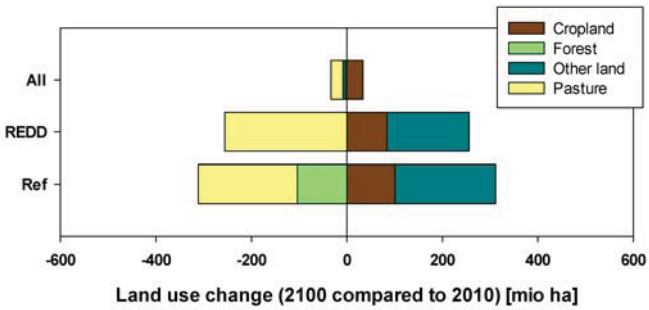

Figure SI-4 Change in global land pools in million ha from 2010 to 2100 for a demiterian scenario in which the consumption of livestock products in all countries is restricted to a maximum of 15% of caloric demand, which is equivalent to half of current consumption in Western countries (Bodirsky et al. 2014).

		Land use				Carbon dynamics			
		Cropland	Forest	Pasture	Other	Land total	LUC	Regrowth	CC & CO_2f
Ref	HadGEM2_ES	209,14	-192,91	-170,78	154,58	-14,85	-180,54	83,07	82,36
	IPSL_CM5A_LR	223,58	-125,17	-249,41	152,00	73,19	-145,05	85,40	133,21
	MIROC_ESM_CHEM	242,50	-160,69	-213,96	131,15	13,02	-159,10	68,54	103,22
	GFDL_ESM2M	252,76	-221,50	-190,62	158,37	180,05	-196,76	81,01	295,45
	NorESM1_M	268,01	-207,88	-235,48	174,35	191,61	-187,74	102,75	276,82
	Mean	***239,19***	***-181,63***	***-212,05***	***154,09***	***88,61***	***-173,15***	***84,17***	***178,25***
	SD	***23,30***	***38,83***	***32,06***	***15,49***	***94,37***	***21,18***	***12,24***	***100,36***
REDD	HadGEM2_ES	178,35	-0,09	-201,00	22,28	30,43	-100,81	52,14	78,55
	IPSL_CM5A_LR	187,84	-0,08	-294,69	107,05	100,27	-103,71	69,18	134,18
	MIROC_ESM_CHEM	227,93	-0,08	-303,42	76,49	44,86	-96,94	42,60	99,13
	GFDL_ESM2M	209,61	-0,18	-219,59	9,14	240,60	-105,53	51,96	294,57
	NorESM1_M	214,79	-0,05	-298,47	83,17	263,07	-76,47	63,33	276,54
	Mean	***203,70***	***-0,10***	***-263,43***	***59,63***	***135,84***	***-96,65***	***55,94***	***176,55***
	SD	***20,24***	***0,05***	***49,05***	***41,93***	***109,33***	***11,63***	***10,56***	***101,48***
All	HadGEM2_ES	27,19	-2,95	-14,00	10,76	89,91	-25,14	32,36	82,70
	IPSL_CM5A_LR	29,61	-5,27	-21,22	2,38	161,11	-14,99	38,20	137,44
	MIROC_ESM_CHEM	29,38	-7,28	-16,62	-5,84	105,16	-13,27	15,01	103,41
	GFDL_ESM2M	43,67	-2,81	-39,88	-1,02	300,86	-20,45	24,11	297,20
	NorESM1_M	44,45	-7,73	-20,46	-16,74	299,71	-18,01	37,40	280,32
	Mean	***34,86***	***-5,21***	***-22,44***	***-2,09***	***191,35***	***-18,73***	***29,42***	***180,31***
	SD	***8,46***	***2,32***	***10,18***	***10,17***	***102,92***	***4,68***	***9,81***	***101,31***

Table SI-1: Change in global land pools in million ha from 2010 to 2100 and cumulative carbon dynamics [Gt CO_2] until 2100 for all scenarios and considered GCMs, as well as the mean and standard deviation across all GCMs.

 DOI: 10.1038/NCLIMATE2444

2. Sensitivity analysis

In order to test the stability of our results in terms of cumulative carbon emissions, we perform sensitivity analyses with crucial exogenous parameters. Besides the sensitivity tests for several GCMs (see Table SI-1), we conducted a sensitivity analysis for (a) demand for livestock products, (b) costs for agricultural yield increases and (c) tax on terrestrial carbon emissions (see Table SI-2). We compare the mean value over GCM specific results for cumulative carbon emissions until 2100 of the sensitivity analyses (a, b and c) to the results of the SSP2 default case presented in the main paper (also mean value over GCM specific results).

a) Demand for livestock products (SSP2 food low)

To test the sensitivity of demand-side measures, such as changes in diets towards less products of animal origin, we included a demiterian scenario in which the consumption of livestock products in all countries is restricted to a maximum of 15% of caloric demand, which is approximately equivalent to half of current consumption in OECD countries (Bodirsky et al. 2014). Here, especially in the REF and REDD scenario, lower carbon emissions from deforestation and other land use change (LUC) and more uptake of carbon from regrowth of secondary natural vegetation on abandoned agricultural land (Regrowth) can be observed. In general, reductions in food demand lower the pressure on the land system but do not change the general findings of our study as highest net carbon uptake (Land total) can be observed in the ALL scenario.

b) Agricultural yield increases (SSP2 tc costs low, SSP2 tc costs high)

In order to increase total agricultural production, MAgPIE can endogenously decide to either invest in yield-increasing technology or increase cropland through land expansion (Popp et al. 2011, Dietrich et al. 2014). To test the stability of our results with respect to the trade-off between land expansion and yield-increasing technological change, we vary the parameters of the cost function for technological change from their default values to the low and the high end of their uncertainty range (see Dietrich et al. 2014). Lower costs for agricultural yield increases (*SSP2 tc costs low*) would lead to higher agricultural yields and lower therefore the pressure on the land system in all scenarios with less emissions from deforestation and other land use change (LUC) and more uptake of carbon from regrowth of secondary natural vegetation on abandoned agricultural land (Regrowth), especially in REF and REDD. All scenarios in *SSP2 tc costs low* show similar net carbon uptake (Land total). In turn, higher costs for land-use intensification (*SSP2 tc costs high*) increase, especially in REF and REDD, emissions from deforestation and other land use change (LUC). Therefore, net carbon uptake (Land total) is lower in all scenarios of *SSP2 tc costs high* compared to *SSP2 default*.

c) Tax on terrestrial carbon emissions (SSP2 C Price low)

In the *SSP2 default* scenarios, the carbon tax has a level of 30 $\$/tCO_2$ in 2020, starts in 2015 and increases by 5% yr^{-1}. For *SSP2 C Price low*, the level of the carbon tax is 5 $\$/tCO_2$ in 2020 respectively. The range for the sensitivity analysis is based on Kriegler et al (2013). Cumulative carbon emissions in REF, REDD and ALL are similar compared the *SSP2 default* case, which indicates that our findings are also valid at substantially lower carbon prices.

DOI: 10.1038/NCLIMATE2444

		Land total	LUC	Regrowth	CC & CO_2f
Ref	*SSP2 default*	*-88,6*	*173,8*	*-84,2*	*-178,2*
	SSP2 food low	-157,9	115,1	-94,7	-178,3
	SSP2 tc costs low	-202,0	78,4	-100,8	-179,6
	SSP2 tc costs high	31,1	296,9	-87,9	-177,9
	SSP2 C Price low	-88,6	173,8	-84,2	-178,2
REDD	*SSP2 default*	*-135,8*	*96,6*	*-55,9*	*-176,6*
	SSP2 food low	-183,8	74,6	-80,6	-177,7
	SSP2 tc costs low	-217,7	61,2	-99,8	-179,1
	SSP2 tc costs high	-92,0	143,6	-59,6	-176,0
	SSP2 C Price low	-119,6	124,3	-67,2	-176,7
All	*SSP2 default*	*-191,4*	*18,4*	*-29,4*	*-180,3*
	SSP2 food low	-191,5	18,5	-29,7	-180,3
	SSP2 tc costs low	-209,3	16,5	-44,6	-181,2
	SSP2 tc costs high	-175,4	26,6	-20,9	-181,1
	SSP2 C Price low	-139,5	85,1	-46,7	-177,9

Table SI-2: Sensitivity analysis for (a) lower demand for livestock products (SSP food low) (b) lower (SSP tc costs low) and (b) higher (SSP2 tc costs high) costs for agricultural yield increases and (c) lower tax on terrestrial carbon emissions (SSP2 C Price low) in comparison to the SSP2 default case. The sensitivity is tested for cumulative carbon emissions ($GtCO_2$) from deforestation and other land use change (LUC), cumulative uptake of carbon from regrowth of secondary natural vegetation on abandoned agricultural land (Regrowth) and cumulative carbon dynamics driven by climate change and CO_2 fertilization (CC & CO_2f). Cumulative carbon dynamics until 2100 are reported as mean values across all GCM specific results.

3. Model validation

In order to test our results, we compare regional MAgPIE projections for cropland and pasture with historical land use data (Klein Goldewijk et al., 2011) (Figures SI-5, SI-6). Deviations of regional MAgPIE cropland in 1995 from historical data stay below 8% and deviations in regional pasture area are below 12%. The near term trend in the MAgPIE cropland projections is in general similar to historical trends except for ASIA where cropland growth in MAgPIE stagnates until 2020. Near term pasture dynamics in ASIA, MAF, OECD and REF are similar to the historical pattern. LAM exhibits slightly declining pasture area in the future while in the historical period, increases were observed.

We report yield growth due to improved agricultural management and technological change by calculating a regional yield index (1995=100) (Figure SI-7). Due to investments into agricultural research and development, the regional yield level in our projections increases on average by ~15 - 20 points per decade in MAF depending on the scenario. In ASIA, yields increase by 8 to 10 points per decade and LAM exhibits yield increases by 5 to 7 points per decade. In OECD and REF, average yield increases stay below 4 points per decade. Historical data from Dietrich et al. (2012) shows yield increases due to technological change by ~14 points per decade after 1960. Fischer & Edmeades (2010) find that yields for the important food crops maize, rice and wheat increased at about 8 to 16 points per decade between 1988 and 2007. Historical corn yield levels in the USA increased at ~14 points per decade between 1960 and present (Egli, 2008). Thus, our productivity pathway is largely compatible with the historical trend.

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Finally, we compare global CO_2 emissions from land use change with the outcome of a model comparison covering four estimates for carbon emissions from LULCC (Houghton et al. 2012). CO_2 emissions in 2005 are very close to mean emissions reported by Houghton et al. The trend in emissions simulated by MAgPIE is similar to historical trends, reporting in general a decrease of emissions over time (Figure SI-8).

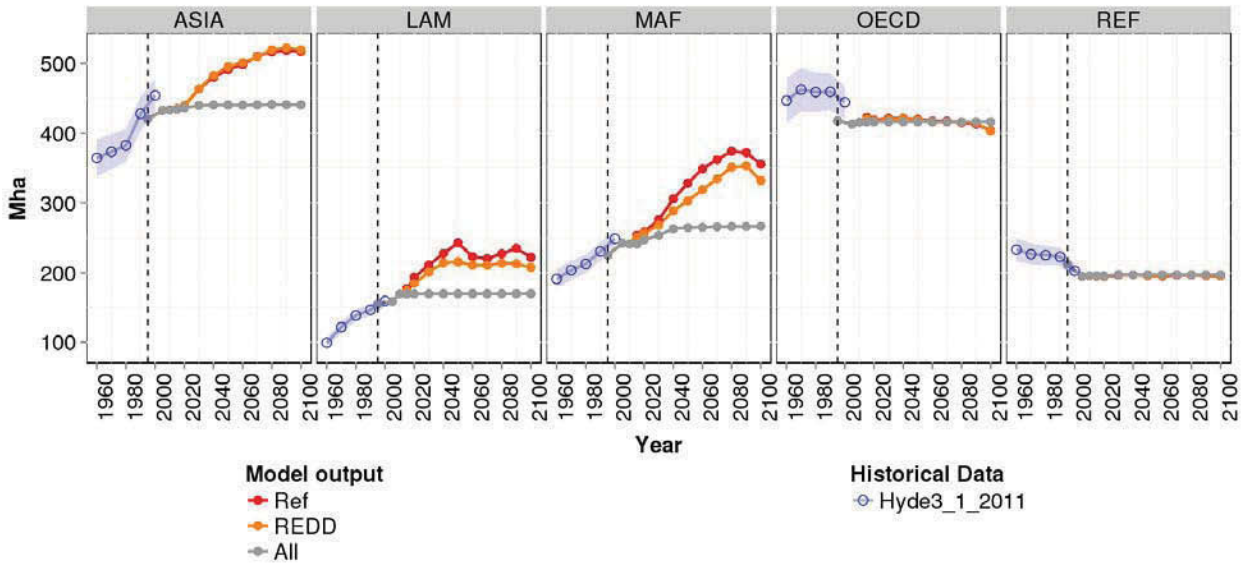

Figure SI-5 Regional cropland development under the Ref (red line), REDD (orange line) and All (grey line) scenarios until 2100 (mean over 5 GCMS). Estimates of historical cropland by Klein Goldewijk et al., 2011 for comparison. A vertical dashed line marks the start of the simulation period.

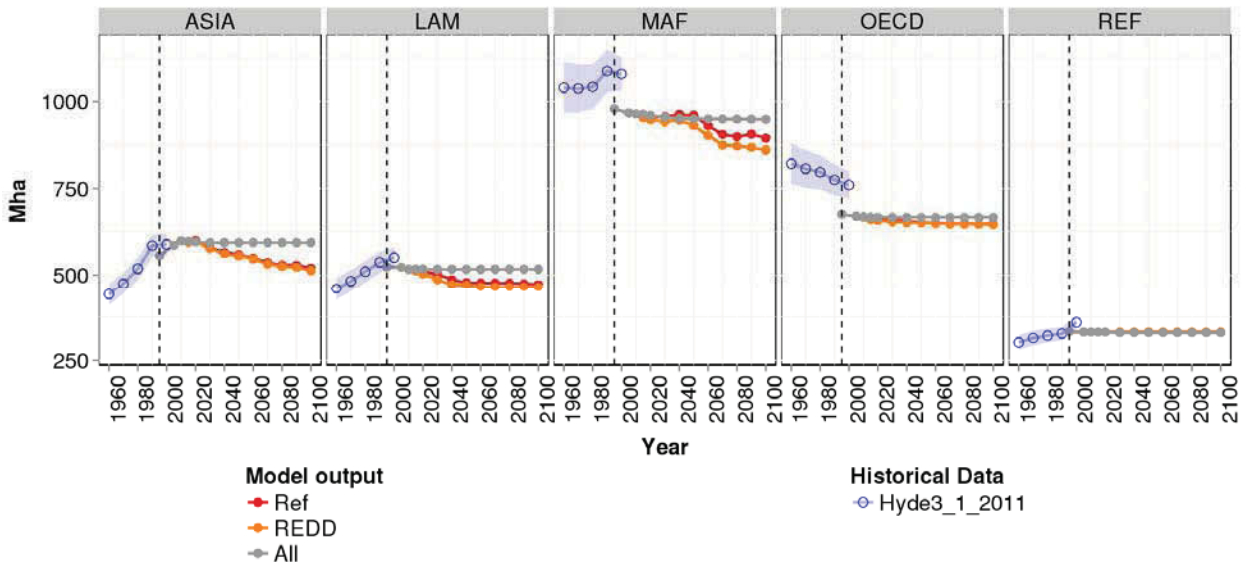

Figure SI-6 Regional pastureland development under the Ref (red line), REDD (orange line) and All (grey line) scenarios until 2100 (mean over 5 GCMS). Estimates of historical pasture area by Klein Goldewijk et al., 2011 for comparison. A vertical dashed line marks the start of the simulation period.

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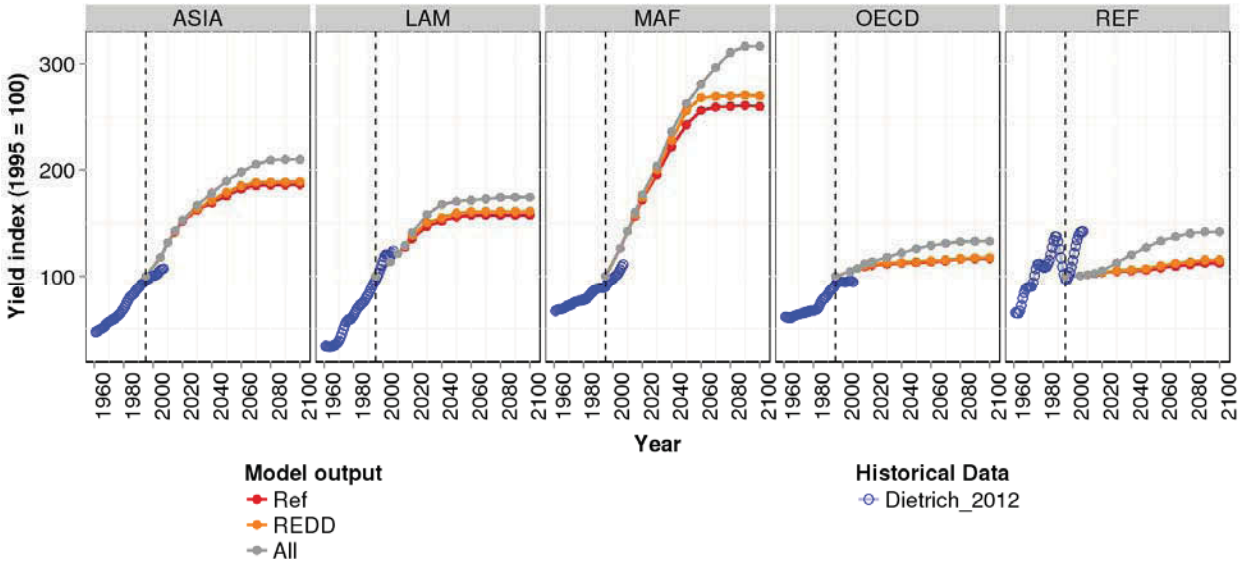

Figure SI-7 Regional yield index (1995 = 100) for the REF (red line), REDD (orange line) and ALL (grey line) scenarios until 2100 (mean over 5 GCMs). Increases over the simulation period reflect investments into yield increasing technological change (TC). Historical data from Dietrich et al. (2012). A vertical dashed line marks the start of the simulation period.

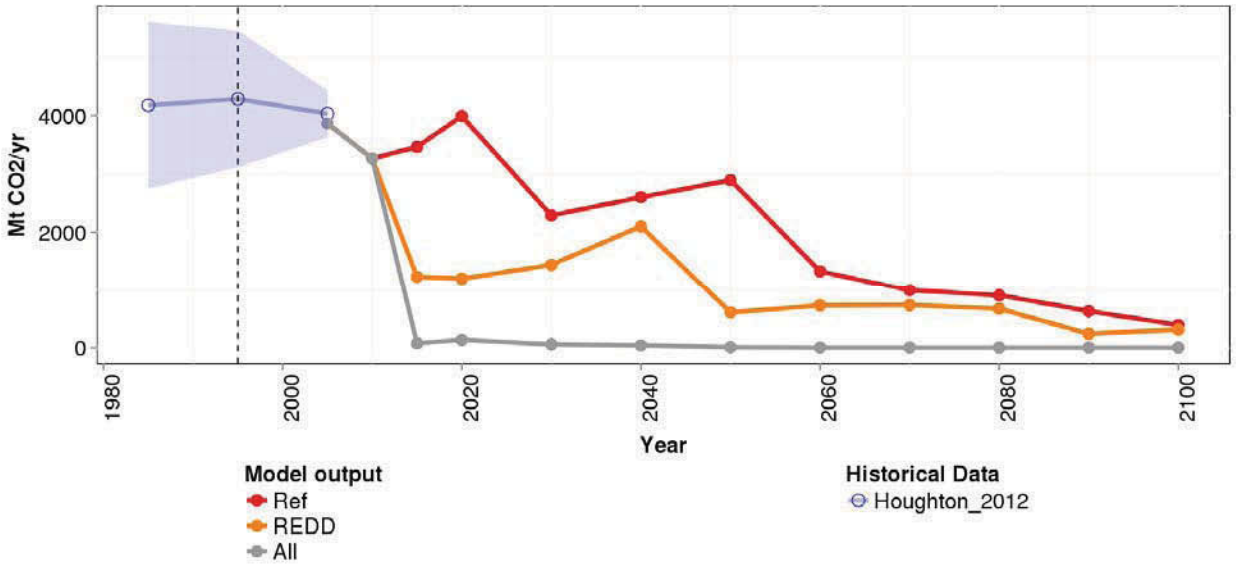

Figure SI-8: Global carbon emissions from land use change for the Ref (red line), REDD (orange line) and All (grey line) scenarios until 2100 (mean over different GCMS) compared to historical data from Houghton et al. (2012). The blue shaded region corresponds to the uncertainty range of the Houghton_2012 estimate. The vertical dashed line marks the start of the simulation period.

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4. Additional model description

4.1. MAgPIE (Model of Agricultural Production and its Impact on the Environment)

The global land-use model MAgPIE (Lotze-Campen et al., 2008, 2009, Popp et al. 2010) is a recursive dynamic optimization model with a cost minimization objective function, which has been coupled to the grid-based dynamic global vegetation model LPJmL (Bondeau et al., 2007, Müller and Robertson 2014). MAgPIE takes regional economic conditions such as demand for agricultural commodities and production costs as well as spatially explicit data on potential crop yields, carbon densities, water availability (from LPJmL) and land into account and derives specific land use patterns, yields and total costs of agricultural production for each grid cell. Due to computational constraints all spatially explicit data with 0.5 degree resolution (59,199 grid cells) is aggregated to 600 simulation units in this study based on a hierarchical bottom-up clustering algorithm (Dietrich et al. 2013).

MAgPIE simulates land-use activities using five land-use classes: cropland, pasture, forest, urban areas (which is a static area), and other land. Input data for each land-use class enters MAgPIE on simulation unit level (Krause et al. 2013). Cropland input is calculated according to the methodology described in Fader et al. (2010) from the MIRCA2000 dataset (Portmann, Siebert, and Döll 2010). Pasture, forest and urban land is largely based on work by Erb et al. (2007). Applying a residual approach, Erb et al (2007) define pasture as the difference between the total area of a grid cell and the area allocated to the sum of the other land use classes, which results in a substantially larger extent of global grazing land as indicated by the FAO definition "Permanent meadows and pastures" (FAOSTAT 2013). To harmonize on FAO data, cellular pasture area based on 4 grazing suitability classes from Erb et al. (2007) is counted up, starting with the most suitable grazing class, until the regional FAO area is matched. Cell specific excess pasture area from there is added to the forest pool for the grazing suitability classes 1 and 2 (aboveground NPP more than 200 g C/m2) and to other land for the unproductive classes 3 and 4. Forest inputs contain the forestry category by Erb et al. (2007), unused pasture area as explained above and those parts of the unused category from the Erb et al. (2007) dataset that are covered by forest according to Bryant et al. (1997) and Greenpeace International (2005). Finally, other land comprises all land area in each grid cell that is not part of any other category. Based on this initialization, in 1995, global cropland area is 1445 mio ha, pasture land area 3262 mio ha, global forest area 4235 mio ha and global other land area 3963 mio ha. Since not all available land is suitable for cropping due to terrain- and agro-edaphic constraints (Fischer et al. 2002), we used the suitability index from Fischer et al (2002) to restrict land that can be converted to cropland (Krause et al. 2013).

LPJmL computes potential irrigated and non-irrigated yields for each crop within each grid cell as an input for MAgPIE. In case of pure rain-fed production, no additional water is required, but yields are generally lower than under irrigation. In addition, LPJmL has been applied a priori to simulate cell specific available water discharge under potential natural vegetation and its downstream movement according to the river routing scheme implemented in LPJmL. Then, if a certain area share is assumed to be irrigated in MAgPIE, additional water for agriculture is taken from available discharge in the grid cell. MAgPIE endogenously decides on the basis of minimizing the costs of agricultural production where to irrigate which crops.

The objective function of the land use optimization model is to minimize total cost of production for a given amount of regional food and bioenergy demand. Regional food energy demand is defined for an exogenously given population in 10 food energy categories. All demand categories are estimated separately for 10 world regions (see Figure SI-9, Table SI-2) and have to be met by the world crop production. Additionally, the regions have to produce a certain share of their demand domestically to account for trade barriers (Schmitz et al., 2012).

Four categories of costs arise in the model: production costs for livestock and crop production, yield increasing technological change costs, land conversion costs and intraregional transport costs. Production costs, containing factor costs for labour, capital, and intermediate inputs, are taken from GTAP (Narayanan and Walmsley, 2008). In order to increase total agricultural production, MAgPIE can either

DOI: 10.1038/NCLIMATE2444

intensify production (yield-increasing technology, shifting from rainfed to irrigated crop production systems) or increase cropland through land expansion. The endogenous implementation of technological change (TC) is based on a surrogate measure for agricultural land use intensity (Dietrich et al., 2012). Investing in TC leads not only to yield increases but also to increases in agricultural land-use intensity, which in turn raises costs for further yield increases. The other alternative for MAgPIE to increase production is to expand cropland into non-agricultural land (Krause et al., 2009; Popp et al., 2011). Cropland expansion involves land conversion costs for every unit of cropland, which account for the preparation of new land and basic infrastructure investments. Land conversion costs are based on country-level marginal access costs generated by the Global Timber Model (GTM) (Sohngen et al., 2009). Moreover, land expansion in MAgPIE is restricted by intraregional transport costs which accrue for every commodity unit as a function of the distance to intraregional markets and the quality of the infrastructure. The data set is based on GTAP transport costs (Narayanan and Walmsley, 2008) and a 30 arc-second resolution data set on travel time to the nearest city (Nelson, 2008).

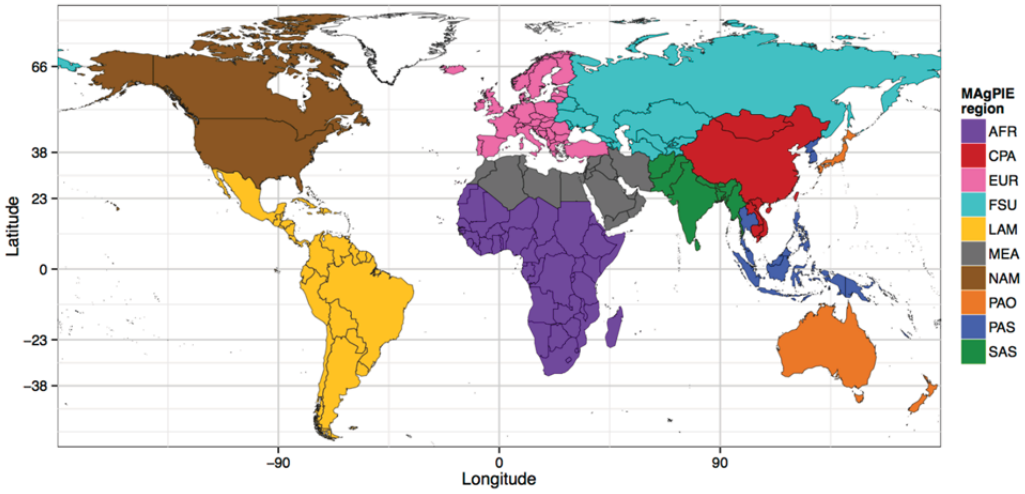

Figure SI-9: MAgPIE economic world regions.

MAgPIE	Region	SSP
AFR	Sub-Saharan Africa	MAF
CPA	Centrally planned Asia including China	ASIA
EUR	Europe including Turkey	OECD
FSU	States of the former Soviet Union	REF
LAM	Latin America	LAM
MEA	Middle East/North Africa	MAF
NAM	North America	OECD
PAO	Pacific OECD including Japan, Australia, New Zealand	OECD
PAS	Pacific (or Southeast) Asia	ASIA
SAS	South Asia including India	ASIA

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Table SI-3. Abbreviations and names of the 10 economic world regions in MAgPIE, and mapping to the 5 SSP regions used in figure SI-1

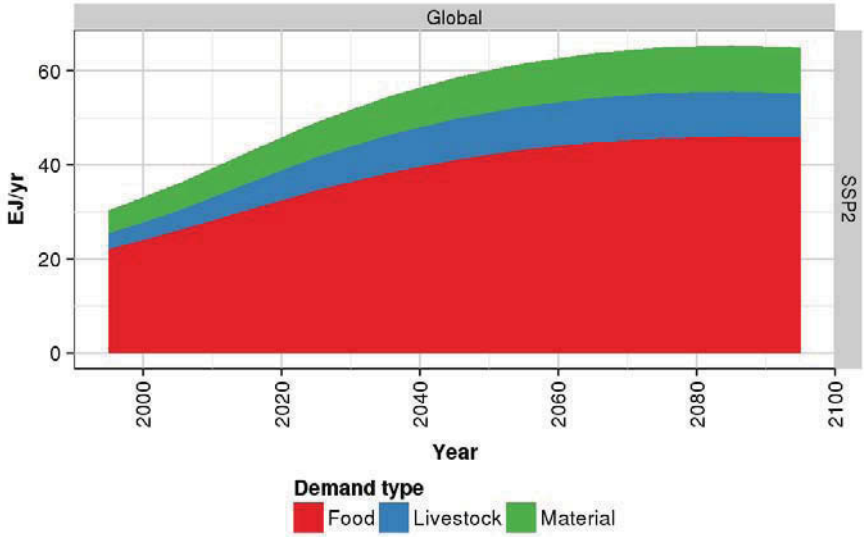

Figure SI-10: Time-series of food, livestock and material demand (based on population and GDP projections; IIASA 2013)

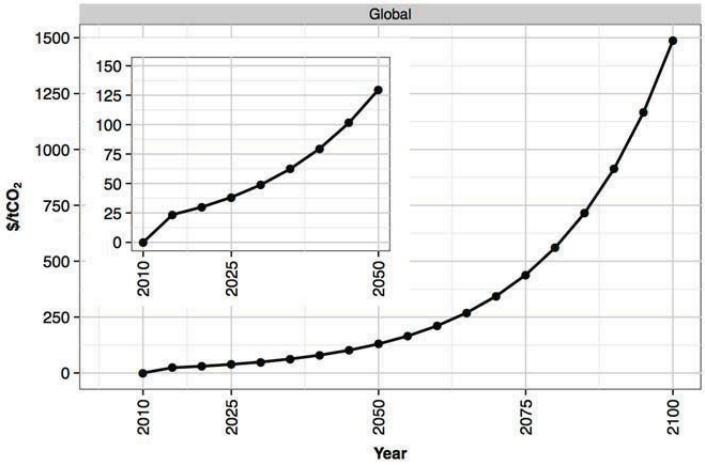

Figure SI-11: Time-series of assumed global tax on terrestrial carbon emissions between 2015 and 2100 in $/t$CO_2$. The carbon tax has a level of 30 $/t$CO_2$ in 2020, starts in 2015 and increases non-linearly at a rate of 5% per year (based on Kriegler et al. 2013).

DOI: 10.1038/NCLIMATE2444

4.2. Detailed model description on carbon stocks and carbon dynamics

4.2.1. Carbon stocks

Simulated land types in MAgPIE are cropland, pasture, forest and other land (e.g. non-forest natural vegetation, abandoned land, desert). Each of them has cell-specific carbon densities for vegetation, soil and litter pools (see Figure SI-12). In the following we describe the parameterization for the carbon pools of each land type:

Forest - Forests are considered as natural vegetation in MAgPIE. Hence, vegetation, litter and soil carbon densities of forests are derived from the global hydrology and vegetation model LPJmL (Bondeau *et al* 2007, Müller and Robertson 2013) at grid-cell level. LPJmL simulates carbon densities (vegetation, litter and soil) for natural ecosystems under consideration of grid cell-specific temperature, precipitation and CO_2 concentration.

Other land - Vegetation, litter and soil carbon density of non-forest natural vegetation and deserts is derived from LPJmL at grid-cell level. Vegetation, litter and soil carbon densities of abandoned agricultural land are dynamic over time (see below).

Cropland – Vegetation carbon densities of cropland were derived by LPJmL, while litter carbon density is assumed to be 0. Due to soil management, croplands have lower soil organic matter than natural vegetation. To account for this, carbon stored in croplands is reduced depending on the climate zone by 20-52% (IPCC 2006; Chapter 5; Table 5.5) relative to natural soil carbon estimated by LPJmL (Bodirsky et al. 2012).

Pasture - Vegetation and litter carbon densities of pasture are derived from LPJmL at grid-cell level. As pastureland and natural vegetation have a similar level of soil organic matter (IPCC 2006; Chapter 6; Table 6.2) soil carbon densities of pastureland at the grid cell level were estimated using the natural vegetation carbon pools of LPJml.

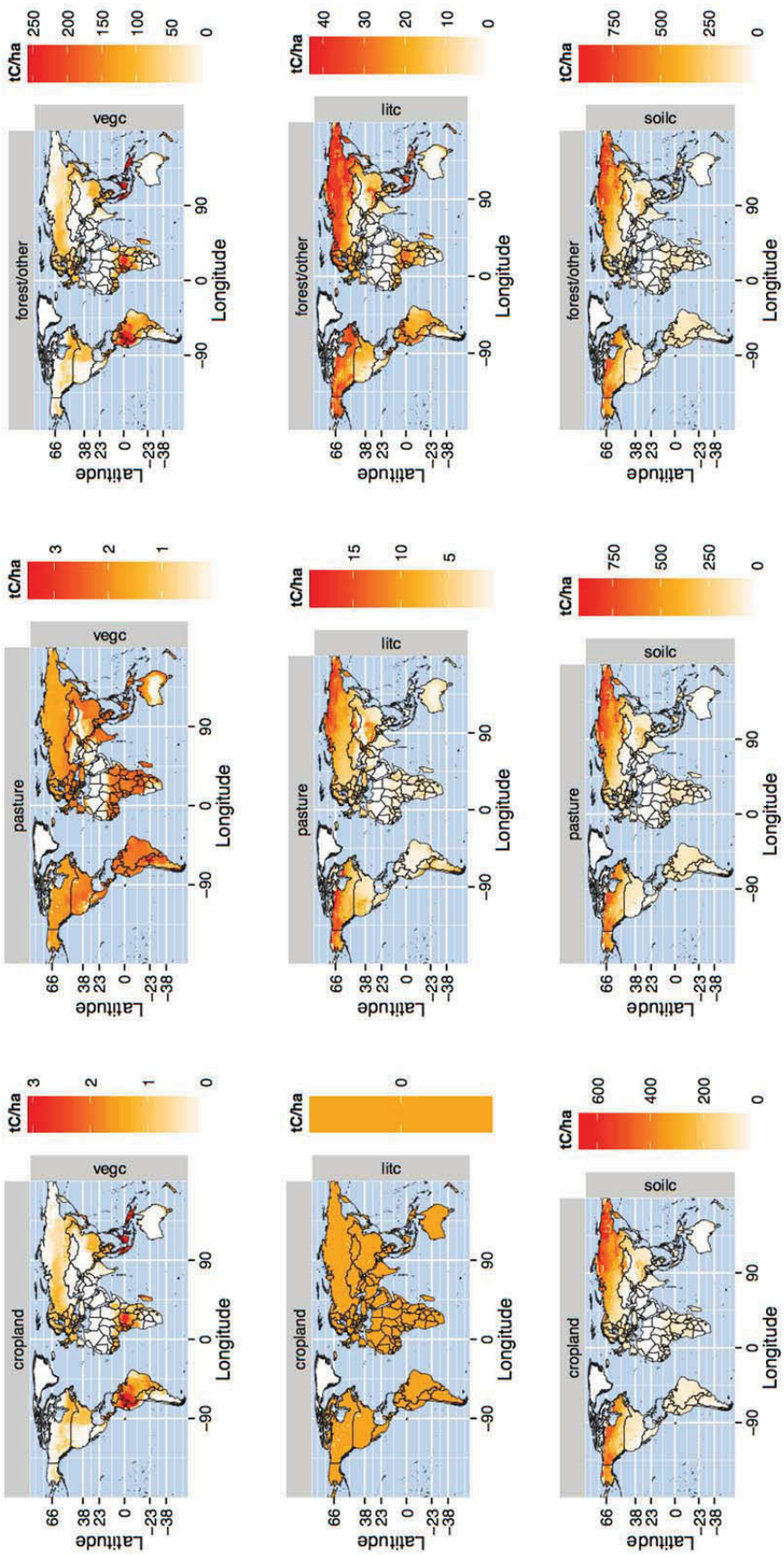

Figure SI-12: Carbon densities in 1995 (tC/ha) in MAgPIE simulated by LPJmL for each land type (cropland – left row; pasture – middle row; forest other natural vegetation – right row) and for each carbon pool (vegetation carbon - upper row; litter carbon – middle row; soil carbon – lower row).

DOI: 10.1038/NCLIMATE2444

4.2.2. Carbon dynamics

In MAgPIE, carbon emissions from land use change occur if the carbon content of the previous land use activity exceeds the carbon content of the new land-use activity, but carbon stocks can also be affected by changing climatic conditions. Carbon emissions become negative if carbon stocks increase, for instance through regrowth of natural vegetation or CO_2 fertilization due to increased levels of CO_2 in the atmosphere.

Regrowth of natural vegetation

If agricultural land is abandoned, regrowth of natural vegetation takes place. Here, vegetation carbon density increases over time along S-shaped growth curves (Figure SI-13). The vegetation carbon density growth curves are based on a Chapman-Richards volume growth model (Murray and von Gadow 1993, Gadow and Hui 2001), which is parameterized using vegetation carbon density of natural vegetation (Figure SI-14) and climate region specific Mean Annual Increment (MAI) and MAI culmination age (IPCC 2006).

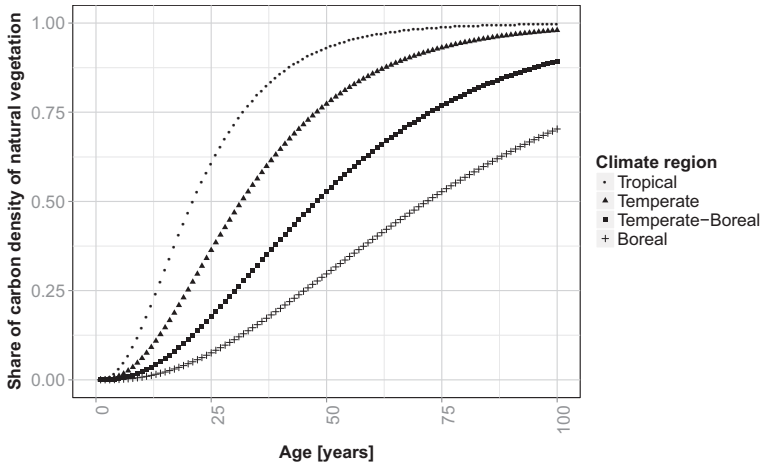

Figure SI-13: Climate region specific S-shaped regrowth curves for a period of 100 years. The vertical axis presents the share of grid-cell specific carbon density of potential natural vegetation in 1995 (Figure SI-14)

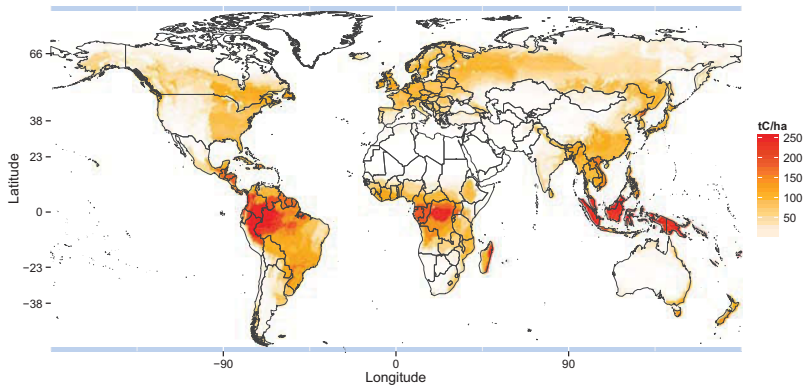

Figure SI-14: Grid-cell specific above- and belowground carbon density of potential natural vegetation in 1995 (tC/ha) derived from LPJmL used as input in Chapman-Richards volume growth model

 DOI: 10.1038/NCLIMATE2444

Litter and soil carbon density of abandoned agricultural land are assumed to increase linearly towards the value of forest/other vegetation (Figure SI-14) within a 20 year's time frame (IPCC 2000). The initial value for vegetation and litter carbon density is assumed to be 0, while the initial value for soil carbon density depends on the former land-use.

Climate change and CO_2 fertilization

Climate impacts in MAgPIE are represented by changes in biophysical inputs like crop yields, carbon densities and water availability, which are derived using the DGVM LPJmL. LPJmL is a dynamic vegetation, hydrology and crop model operating at a global grid (0.5 degree longitude/latitude). LPJmL simulates yields for the most important agricultural crops, and carbon densities (vegetation, litter and soil) for natural ecosystem under consideration of different climatic scenarios. To account for uncertainty in climate projections for RCP 2.6, we use in this study the five GCMs: HadGEM2-ES, IPSL-CM5A-LR, MIROC-ESM-CHEM, GFDL-ESM2M and NorESM1-M (Hempel et al. 2013). To facilitate comparison of MAgPIE results, GCM specific biophysical inputs derived from LPJmL are harmonized for the initial MAgPIE time step. Yield harmonization is achieved by defining a reference GCM (HadGEM2-ES) and multiplication of the relative changes (all time steps divided by initial time step) of all other GCMs with this reference. This method preserves the relative differences and assures that the input data is identical for the initial time step. For carbon densities, this approach leads to a distortion of the temporal dynamics compared to the original data. Therefore, GCM specific differences with respect to 1995 have been added to the 1995 reference value. Resulting negative values are set to 0 and values that exceed the maximum carbon density in the original data have been cut off. Changes in carbon densities are given as a mean (Figure SI-15) and standard deviation (Figure SI-16) across all GCMs for different carbon pools (vegetation, litter and soil) and land types.

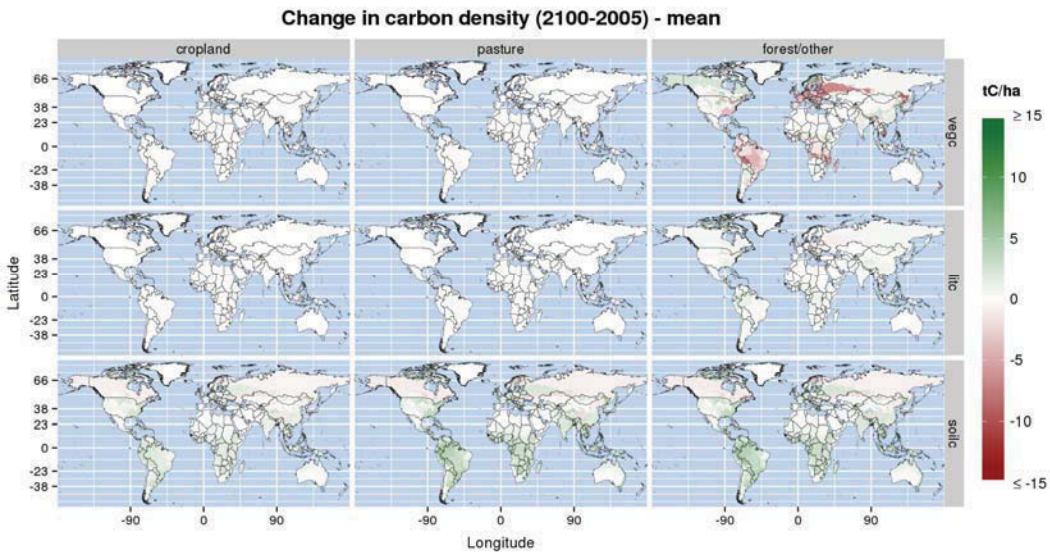

Figure SI-15 Mean change (2100 compared to 2005) in carbon densities (tC/ha) simulated by LPJmL across five applied GCMs (HadGEM2-ES, IPSL-CM5A-LR, MIROC-ESM-CHEM, GFDL-ESM2M and NorESM1-M) based on climate projections for RCP 2.6. For each land type (cropland – left row; pasture – middle row; forest other natural vegetation – right row) and for each carbon pool (vegetation carbon - upper row; litter carbon – middle row; soil carbon – lower row).

DOI: 10.1038/NCLIMATE2444

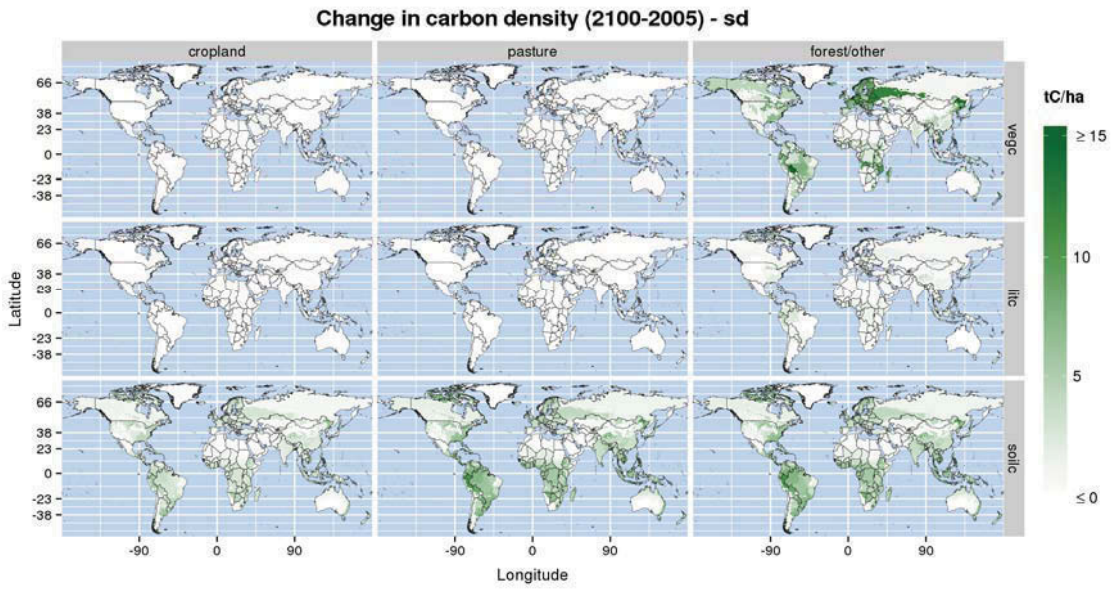

Figure SI-16: Standard deviation (2100 compared to 2005) from mean carbon densities (tC/ha) simulated by LPJmL across five applied GCMs (HadGEM2-ES, IPSL-CM5A-LR, MIROC-ESM-CHEM, GFDL-ESM2M and NorESM1-M) based on climate projections for RCP 2.6. For each land type (cropland – left row; pasture – middle row; forest other natural vegetation – right row) and for each carbon pool (vegetation carbon - upper row; litter carbon – middle row; soil carbon – lower row).

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Chapter III

Investigating afforestation and bioenergy CCS as climate change mitigation strategies *

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* published in Environmental Research Letters Volume 9, Issue 6, 2014-05-01, DOI: 10.1088/1748-9326/9/6/064029

Content of Chapter III: Investigating afforestation and bioenergy CCS as climate change mitigation strategies

OPEN ACCESS
IOP Publishing
Environmental Research Letters
Environ. Res. Lett. 9 (2014) 064029 (13pp)
doi:10.1088/1748-9326/9/6/064029

Investigating afforestation and bioenergy CCS as climate change mitigation strategies

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Received 26 December 2013, revised 5 April 2014
Accepted for publication 7 May 2014
Published 23 June 2014

Abstract

The land-use sector can contribute to climate change mitigation not only by reducing greenhouse gas (GHG) emissions, but also by increasing carbon uptake from the atmosphere and thereby creating negative CO_2 emissions. In this paper, we investigate two land-based climate change mitigation strategies for carbon removal: (1) afforestation and (2) bioenergy in combination with carbon capture and storage technology (bioenergy CCS). In our approach, a global tax on GHG emissions aimed at ambitious climate change mitigation incentivizes land-based mitigation by penalizing positive and rewarding negative CO_2 emissions from the land-use system. We analyze afforestation and bioenergy CCS as standalone and combined mitigation strategies. We find that afforestation is a cost-efficient strategy for carbon removal at relatively low carbon prices, while bioenergy CCS becomes competitive only at higher prices. According to our results, cumulative carbon removal due to afforestation and bioenergy CCS is similar at the end of 21st century (600–700 $GtCO_2$), while land-demand for afforestation is much higher compared to bioenergy CCS. In the combined setting, we identify competition for land, but the impact on the mitigation potential (1000 $GtCO_2$) is partially alleviated by productivity increases in the agricultural sector. Moreover, our results indicate that early-century afforestation presumably will not negatively impact carbon removal due to bioenergy CCS in the second half of the 21st century. A sensitivity analysis shows that land-based mitigation is very sensitive to different levels of GHG taxes. Besides that, the mitigation potential of bioenergy CCS highly depends on the development of future bioenergy yields and the availability of geological carbon storage, while for afforestation projects the length of the crediting period is crucial.

S Online supplementary data available from stacks.iop.org/ERL/9/064029/mmedia

Keywords: climate change mitigation, afforestation, bioenergy, carbon capture and storage, land-use modeling, land-based mitigation, carbon sequestration

Introduction

For ambitious climate change mitigation, huge reductions in greenhouse gas (GHG) emissions are needed (Meinshausen *et al* 2009, Rogelj *et al* 2011, 2013a). Currently, the land-use sector is responsible for 17–32% of global anthropogenic GHG emissions (Bellarby *et al* 2008). There are several measures for reducing GHG emissions in the land-use sector,

such as avoided deforestation or improved agricultural management (Smith *et al* 2013). However, the land-use sector cannot only contribute to climate change mitigation by decreasing GHG emissions, but also by increasing carbon uptake from the atmosphere (Rose *et al* 2012). Recent integrated assessment modeling (IAM) studies focused on afforestation and bioenergy in combination with carbon capture and storage (bioenergy CCS) as land-based mitigation strategies for carbon removal (Tavoni and Socolow 2013). Both strategies make use of the accumulation of carbon in growing biomass through photosynthesis. Bioenergy CCS removes carbon from the atmosphere by capturing the carbon released during the combustion of biomass and storing the captured carbon in geological reservoirs underground. Afforestation detracts carbon from the atmosphere through the managed regrowth of natural vegetation. While carbon removal due to bioenergy CCS can be raised through land expansion *and* yield increases (increase in productivity per unit area), gains in carbon removal due to afforestation are mostly limited to the extensification of forestland.

In the literature, studies focus on the standalone mitigation potential of bioenergy CCS (Azar *et al* 2010, Popp *et al* 2011, Klein *et al* 2014, Kriegler *et al* 2013, Vuuren *et al* 2013) and afforestation (Strengers *et al* 2008, Reilly *et al* 2012) or investigate both at the same time (Wise *et al* 2009, Calvin *et al* 2014, Edmonds *et al* 2013). However, the standalone and combined effects of bioenergy CCS and afforestation on land-use and carbon dynamics have not been analyzed so far with a common methodological approach. Looking at both, the standalone and combined mitigation potential, provides insight into potential trade-offs like competition for land or path dependencies, which are important for the evaluation of afforestation and bioenergy CCS as mitigation strategies. In this study, we use the Model of Agricultural Production and its Impacts on the Environment (MAgPIE), a spatially explicit, global land-use model to analyze three scenarios with different land-based mitigation policies: afforestation, bioenergy CCS and the combination of both. Land-based mitigation in MAgPIE is incentivized by an exogenously given tax on GHG emissions. The trade-off between land expansion and yield increases is treated endogenously in the model. In order to test the stability of our results, we perform sensitivity analyses with the most important exogenous parameters.

Methods and material

Land-use model MAgPIE

MAgPIE is a spatially explicit, global land-use allocation model and projects land-use dynamics in ten-year time steps until 2095 using recursive dynamic optimization (Lotze-Campen *et al* 2008, Popp *et al* 2010). The objective function of MAgPIE is the fulfilment of food, livestock and material demand at minimum costs under socio-economic and biophysical constraints. Demand is based on exogenous future population and income projections (see scenario section), while price-induced changes in consumption are not reflected. Major cost types in MAgPIE are: factor requirement costs (capital, labor and fertilizer), land conversion costs, transportation costs to the closest market, costs for R&D (technological change) and costs for GHG emission rights. For long term investments, like land conversion or R&D, we assume a time horizon of 30 years and an annual discount rate of 7%, which reflects the opportunity costs of capital at the global level (IPCC 2007, chapter 2.4.2.1). While socio-economic constraints like trade liberalization and forest protection are defined at the regional level (ten world regions) (figure S1), biophysical constraints such as crop yields, carbon density and water availability, derived from the global hydrology and vegetation model LPJmL (Bondeau *et al* 2007, Müller and Robertson 2014), as well as land availability (Krause *et al* 2013), are introduced at the grid cell level (0.5 degree longitude/latitude; 59 199 grid cells). Due to computational constraints, all model inputs in 0.5 degree resolution are aggregated to 500 clusters for the optimization process based on a k-means clustering algorithm (Dietrich *et al* 2013). During the optimization process, the cluster level is the finest resolution. The clustering algorithm combines grid cells to clusters based on the similarity of data for each of the ten world regions. If, for instance, two grid cells with similar land patterns are merged into one cluster, the algorithm preserves the land area available by summing up the area of the two grid cells. Investment in R&D in the agricultural sector translating into yield-increasing technological change is a variable in MAgPIE and can therefore endogenously enhance food and bioenergy crop yields (Dietrich *et al* 2014). Finally, the cost minimization problem is solved through endogenous variation of spatial production patterns, land expansion (both at the cluster level) and yield-increasing technological change (at the regional level) (Lotze-Campen *et al* 2010).

MAgPIE features land-use competition based on cost-effectiveness at cluster level among the land-use related activities food, livestock and bioenergy production as well as afforestation. Available land types are cropland, pasture, forest and other land (e.g. non-forest natural vegetation, abandoned agricultural land, desert). The forestry sector, in contrast to the agricultural and livestock sector, is currently not implemented dynamically in MAgPIE. Therefore, timberland needed for wood production, consisting of forest plantations and modified natural forest, is excluded from the optimization, which is about 30% of the initial global forest area (4235 mio ha). In addition, other parts of forestland, mainly undisturbed natural forest, are within protected forest areas, which is about 12.5% of the initial global forest area (FAO 2010). Altogether, about 86% of the world's land surface is freely available in the optimization of the initial time-step.

MAgPIE calculates emissions of the Kyoto GHGs carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) (Bodirsky *et al* 2012, Popp *et al* 2010, 2012). Mitigation of GHG emissions is stimulated by an exogenous tax regime on GHG emissions (see scenario section). The GHG tax is multiplied with GHG emissions in order to calculate GHG emission costs, which enter the cost minimizing objective

Table 1. Regional herbaceous bioenergy yields (GJ ha^{-1}) in 1995 derived from LPJmL (potential yields) and initial bioenergy yields MAgPIE (actual yields). Region specific yields are obtained by calculating the average across all clusters within a region.

	AFR	CPA	EUR	FSU	LAM	MEA	NAM	PAO	PAS	SAS
LPJmL	198	188	165	59	382	26	101	154	595	235
MAgPIE	52	105	125	21	150	9	71	48	251	110

function of MAgPIE. For instance, CO_2 emissions from land-use change can be reduced through avoided deforestation (carbon stock conservation). But unlike N_2O and CH_4 land-use emissions, CO_2 emission from the land-use system can become negative if photosynthetic carbon uptake from the atmosphere outweighs carbon release to the atmosphere from plant decomposition and land-use change (carbon stock increase). Therefore, land-based climate change mitigation via afforestation or bioenergy CCS is incentivized by the revenue from the GHG tax regime for carbon removal from the atmosphere. A detailed description of the underlying formulas is available in the supplementary data, available at stacks.iop.org/ERL/0/000000/mmedia. For the conversion of N_2O and CH_4 emissions into CO_{2eq} we use GWP100 (IPCC (2013)).

Bioenergy CCS

In MAgPIE, dedicated lignocellulosic biomass (rainfed only) can be converted to secondary energy via different conversion routes. The carbon released from biomass during combustion is captured and stored underground using CCS technology.

Herbaceous and woody bioenergy yields at grid cell level for the initialization of MAgPIE are derived from LPJmL (Beringer *et al* 2011). While LPJmL simulations supply data on potential yields, i.e. yields achieved under the best currently available management options, MAgPIE aims to represent actual yields. Therefore, LPJmL yields are reduced using information about observed land-use intensity (Dietrich *et al* 2012) and agricultural area (FAO 2013). For instance, in China (CPA) herbaceous bioenergy yields from LPJmL are reduced by about 45% to obtain actual yields for the initialization of MAgPIE (table 1). MAgPIE bioenergy yields can exceed LPJmL bioenergy yields over time as endogenous investments in R&D push the technology frontier. Higher bioenergy yields are associated with increased carbon uptake from the atmosphere per unit area.

Bioenergy CCS can provide energy and remove carbon from the atmosphere at the same time. Due to this versatility, bioenergy CCS is an attractive mitigation option in scenarios with ambitious climate targets. The largest share of profits in these scenarios comes from the carbon sequestration and not from the energy portion of the bioenergy CCS technology (Rose *et al* 2014). In this study, we focus on the carbon removal potential of bioenergy CCS and therefore disregard the value of the energy produced. We implemented three different conversion routes with CCS technology in the model: biomass to hydrogen (B2H2), biomass integrated gasification combined cycle and biomass to liquid (B2L) (Klein *et al* 2014). Levelized costs of energy (LCOEs) are calculated using the time horizon of 30 years and the discount rate of 7% (see MAgPIE description), which are both within the range of common assumptions for LCOE calculations (IEA and OECD NEA 2010, chap 2.3). The LCOEs include initial investment costs in infrastructure as well as operational and maintenance costs. The B2H2 technology in combination with CCS features a higher conversion efficiency (55%) and carbon capture rate (90%) at lower LCOEs (8 \$ GJ^{-1}) than the other available technologies (based on Klein *et al* 2014). Demand for bioenergy in this study does not rely on exogenous trajectories, but is derived endogenously as a response to the GHG tax, which rewards carbon removal due bioenergy CCS, while the value of energy produced due to bioenergy CCS is disregarded. The cost minimizing objective function of MAgPIE in combination with carbon removal being the only incentive in the model to employ bioenergy CCS renders the B2H2 technology superior to the other available conversion routes. The geological carbon storage capacity is constrained at the regional level (table S3), which adds up to 3960 $GtCO_2$ at the global level (Bradshaw *et al* 2007). We assume a lifetime of the CCS technology of 200 years (Szulczewski *et al* 2012) and therefore limit the annual geological injection of carbon to 0.5% yr^{-1} in terms of the geological carbon storage capacity, which results in an annual realizable geological injection rate of about 20 $GtCO_2$ yr^{-1} globally. As biomass can be traded, the location of geological carbon storage can differ from the location of biomass production. Levelized costs for transportation and injection of captured carbon are at 9 \$/tCO_2 (Klein *et al* 2014).

Afforestation

Compared to bioenergy CCS, afforestation can be considered as low-tech land-based mitigation strategy, since no technical infrastructure for processing is needed. In MAgPIE, afforestation is a managed regrowth of natural vegetation. The regrowth is managed in that way as endemic seed sources are put in place manually as part of the land conversion process. Regrowth of natural vegetation affects vegetation, litter and soil carbon stocks, which are calculated as the product of carbon density and afforestation area (see online supplementary data for details). Vegetation, litter and soil carbon density of potential natural vegetation in 1995 at grid cell level is derived from LPJmL (figure S4). Vegetation carbon density increases over time along S-shaped growth curves (figure 1). The vegetation carbon density growth curves are based on a Chapman–Richards volume growth model (Murray and von Gadow 1993, Gadow and Hui 2001), which is parameterized using vegetation carbon density of potential natural vegetation (figure S4(a)) and climate region specific

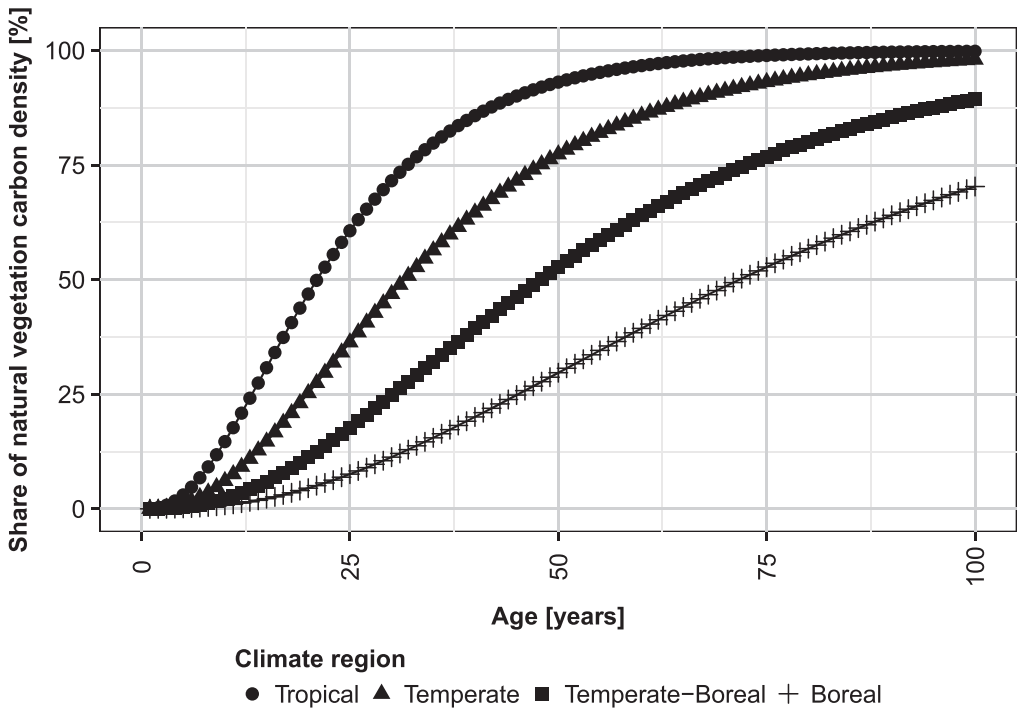

Figure 1. Climate region specific S-shaped vegetation carbon density growth curves for a period of 100 years. The vertical axis presents the share of grid-cell specific vegetation carbon density of potential natural vegetation in 1995 (figure S4a).

mean annual increment (MAI) and MAI culmination age (IPCC 2006). Litter and soil carbon density are assumed to increase linearly towards the value of potential natural vegetation (figures S4(b) and (c)) within a 20 years time frame (IPCC 2000). The initial value for vegetation and litter carbon density is assumed to be 0, while the initial grid-cell specific value for soil carbon density is the average between cropland and potential natural vegetation soil carbon density.

In MAgPIE, the decision to invest in afforestation depends on the benefit-cost ratio over the time horizon of 30 years, which is a common crediting period for afforestation projects (United Nations 2013). Firstly, cumulative carbon uptake over the time horizon is calculated as the product of new afforestation area and carbon density (vegetation, litter and soil) at age 30. Secondly, the benefit of an afforestation activity is calculated as the product of this cumulative carbon uptake 30 years ahead and the level of the current GHG tax (see online supplementary data for more details). Finally, for comparability with the annual bioenergy CCS activity, this future cash flow is annualized using the discount rate of 7% (see MAgPIE description). Land conversion and management costs are based on Sathaye *et al* (2005). Regional costs for the conversion of any land type into afforestation land range between 849 \$ ha^{-1} (SAS) and 2484 \$ ha^{-1} (NAM) (table S2) in the initial time step and are proportionally scaled with GDP for future time steps. Total land conversion costs are annualized. Annual management costs range between 2 \$ ha^{-1} yr^{-1} (FSU) and 127 \$ ha^{-1} yr^{-1} (AFR). Contrary to bioenergy CCS, technological change has no direct effect on the carbon removal potential of afforestation.

Scenarios

Our scenarios are based on the shared socio-economic pathways (SSPs) for climate change research (O'Neill *et al* 2014). It should be noted that the SSPs do not incorporate climate mitigation policies by definition. In this study, we choose SSP 2, a 'Middle of the Road' scenario with intermediate socio-economic challenges for adaptation and mitigation. Food, livestock and material demand (figure S2) is calculated using the methodology described in Bodirsky *et al* and the SSP 2 population and GDP projections (IIASA 2013).

We assume a GHG tax (Tax30) on Kyoto gases (CO_2, N_2O, CH_4) that increases nonlinearly at a rate of 5% yr^{-1} (Kriegler *et al* 2013). The GHG tax has a level of 30 \$/tCO_{2eq} in 2020 and starts in 2015 (figure S3). The resulting GHG tax with prices of 102 \$/tCO_{2eq} in 2045 and 1165 \$/tCO_{2eq} in 2095 is close to GHG price trajectories required to limit global average temperature increase to 2 °C above pre-industrial levels with a probability of 50% (Rogelj *et al* 2013b). Due to this ambitious climate change mitigation target, biophysical climate impacts on crop yields and carbon densities are assumed to be weak and are therefore not regarded in this study.

We investigate four scenarios, which cover two dimensions: GHG tax and availability of carbon removal options (table 2). In the business as usual scenario (BAU), no tax on GHG emissions is applied, i.e. there is no incentive to avoid GHG emissions. In the mitigation scenarios, the GHG tax penalizes all positive GHG emissions from the land-use system and rewards negative CO_2 emissions from afforestation in AFF, from bioenergy CCS in BECCS, and from both in AFF +BECCS.

Sensitivity analysis

To address the uncertainty in exogenous model parameters, we investigate the sensitivity of our simulations to changes in model parameterization. Crucial parameters in the context of this study are the geological carbon storage capacity (CCS capacity), the GHG tax, the time horizon for investment decisions, the discount rate, and assumptions about future bioenergy yields. Table 3 summarizes the parameter settings for the sensitivity analysis.

CCS capacity: Bradshaw *et al* (2007) estimates a range for geological carbon storage capacity at the global level of 100–200 000 $GtCO_2$. For DEFAULT, we use 3960 $GtCO_2$. For the sensitivity analysis we vary this value by factor 20 in both directions (198 $GtCO_2$ in LOW, 79 200 $GtCO_2$ in HIGH). Based on Szulczewski *et al* (2012), we assume a lifetime of CCS of 200 years. Therefore, we limit the annual injection of carbon to 0.5% yr^{-1} in terms of the total CCS capacity. For DEFAULT this results in 20 $GtCO_2$ of the total CCS capacity, for LOW in 1 $GtCO_2$ yr^{-1} and for HIGH in 396 $GtCO_2$ of the total CCS capacity globally.

GHG tax: in DEFAULT, the GHG tax has a level of 30 $/$tCO_{2eq}$ in 2020, starts in 2015 and increases by 5% yr^{-1}. For LOW and HIGH, the level of the GHG tax is 5 and 50 $/$tCO_{2eq}$ in 2020 respectively. The range for the sensitivity analysis is based on Kriegler *et al* (2013).

Time horizon: in DEFAULT, the time horizon for investments is 30 years, which is common in the energy sector as well as for afforestation projects. For LOW and HIGH we chose 10 and 50 years respectively (IEA and OECD NEA 2010, United Nations 2013).

Discount rate: in DEFAULT, the annual discount rate is 7%, which reflects the opportunity costs of capital at the global level. For the sensitivity analysis, we vary the discount rate by 3% points in both directions, based on a literature range of 4–12% (IPCC (2007), chap 2.4.2.1).

Bioenergy yields: in DEFAULT, bioenergy yields are variable and assumed to increase, along with food crop yields, due to endogenous technological change in the agricultural

Table 2. Scenario definitions; GHG tax: Tax30 has a level of 30 $/$tCO_{2eq}$ in 2020, starts in 2015 and increases by 5% yr^{-1}; carbon removal option(s): available option(s) for generating negative CO_2 emissions rewarded by the GHG tax.

	GHG tax	Carbon removal option(s)
BAU	—	—
AFF	Tax30	Afforestation
BECCS	Tax30	Bioenergy CCS
AFF +BECCS	Tax30	Afforestation and bioenergy CCS

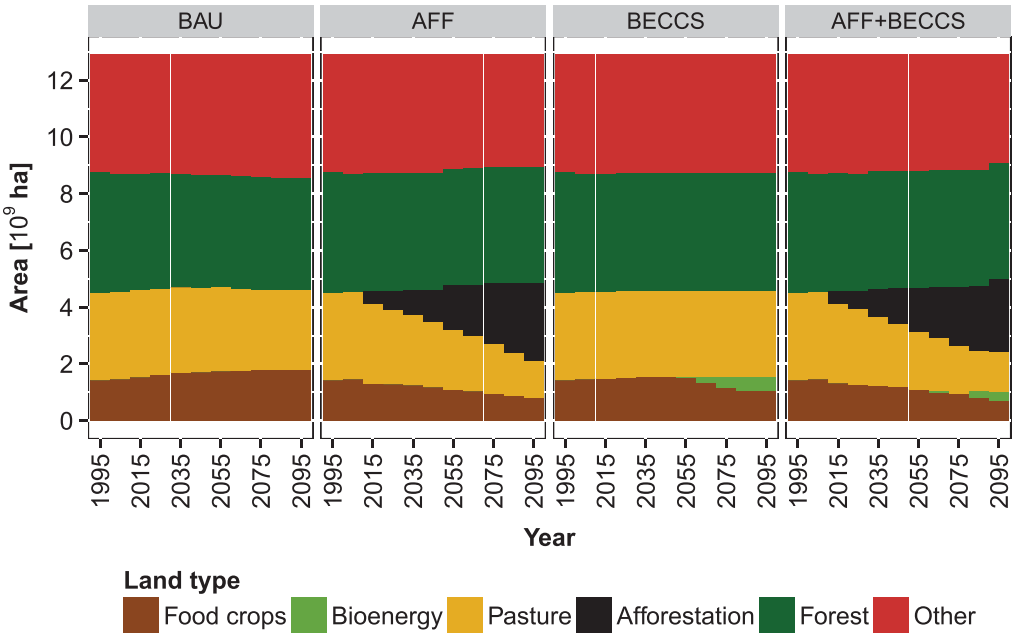

Figure 2. Time-series of global land-use pattern (10^9 ha) for BAU, AFF, BECCS and AFF+BECCS and six land types.

Table 3. Parameter settings for sensitivity analysis. 'DEFAULT' characterizes our default parameter settings used in this study. 'LOW'/ 'HIGH' characterize lower/higher parameter values compared to 'DEFAULT'.

	CCS capacity globally [Gt CO_2]	GHG tax in 2020 (2095) [US$/$tCO_{2eq}$]	Time horizon [Years]	Discount rate [% yr^{-1}]	Bioenergy yields [–]
LOW	198	5 (194)	10	4	Static
DEFAULT	3960	30 (1165)	30	7	Variable
HIGH	79200	50 (1942)	50	10	—

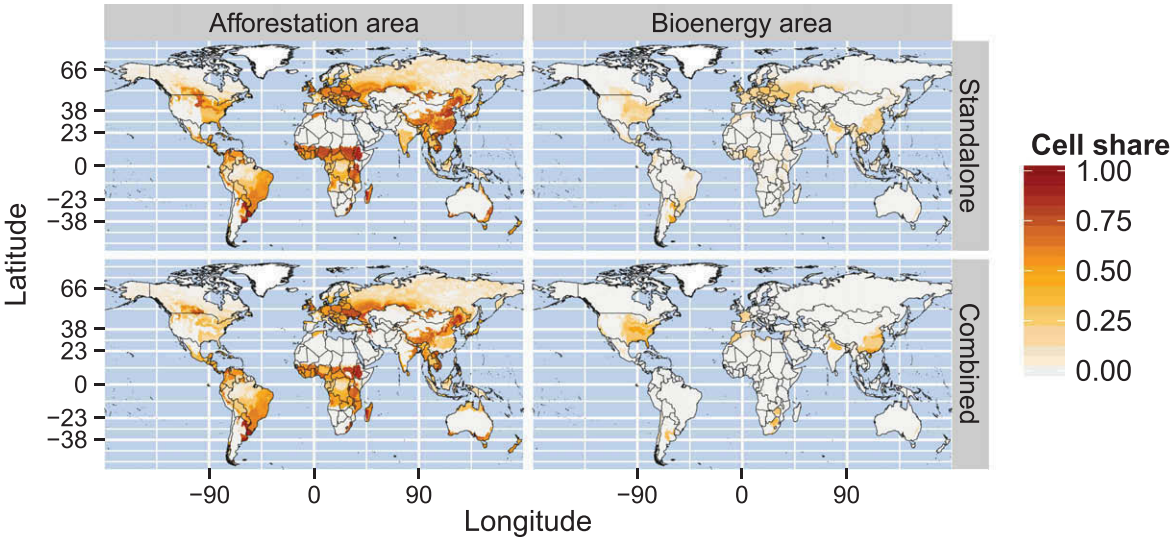

Figure 3. Grid-cell specific share of afforestation and bioenergy area in the standalone scenarios (top) and the combined setting (bottom) in 2095. Colors indicate the share of afforestation or bioenergy area in each cell. Grid-cell specific results are obtained by disaggregation of cluster level results (each grid cell is assigned the value of the cluster it belongs to).

sector. In LOW, technological change in the agricultural sector has no impact on bioenergy crop yields (i.e. bioenergy yields are fixed at the initial level), while food crop yields can still increase due to technological change.

To investigate the role of the algorithm used for clustering our high resolution input data, we perform sensitivity analysis with different numbers of spatial cluster units (100–500) in addition.

Results

Land-use dynamics

In 1995, total land cover (12 907 mio ha) consists of food crop (1425 mio ha), pasture (3073 mio ha), forest (4235 mio ha) and other land (4174 mio ha) (figure 2). In the BAU scenario (no GHG tax), food crop area increases by about 300 mio ha until 2095, mainly at the expense of forestland. In the second half of the century, pasture area decreases due to stabilizing livestock demand (figure S2) in combination with average yield increases of about 0.48% yr^{-1} (figure S10), leading to an increase of abandoned agricultural land. In the mitigation scenarios, the GHG tax on land-use change emissions keeps forestland almost constant over time. Afforestation emerges as cost-efficient mitigation strategy from 2015 (start of GHG tax at 24 \$/tCO_{2eq}) and increases, mainly at the expense of pasture and food crop area, to 2773 mio ha in AFF until 2095. Endogenous yield increases, accompanied by changes in spatial production patterns, compensate for the reduced agricultural area. In AFF, the cost–efficient level of average yield increases in the agricultural sector is 1.21% yr^{-1} throughout the 21st century (figure S10). Bioenergy CCS comes into play much later than afforestation, as it is cost-efficient first in 2065 (270 \$/tCO_{2eq}). Bioenergy area increases to 508 mio ha until 2095 in BECCS, mainly at the expense of food crop area. Total dedicated bioenergy production, mainly herbaceous crops, stabilizes at 237 EJ yr^{-1} until 2095 (figures S6, S7). In the combined setting, AFF+BECCS, afforestation area (2566 mio ha) is slightly smaller compared to AFF, while bioenergy area (300 mio ha) is almost halved compared to BECCS. Despite the smaller bioenergy area, bioenergy production remains at 237 EJ yr^{-1} in 2095 in AFF +BECCS, which is reflected in a higher level of average yield increases in AFF+BECCS (1.37% yr^{-1}) compared BECCS (1% yr^{-1}) (figure S10).

The maps in figure 3 illustrate the spatial distribution of afforestation and bioenergy area for the standalone scenarios (AFF/BECCS) and the combined setting (AFF+BECCS) in 2095. In the standalone scenarios, afforestation area is found in many world regions, predominantly in Sub-Sahara African, Latin America, China, Europe and the USA, while bioenergy area is mostly found in the northern hemisphere in the USA, China and Europe. In the combined scenario, afforestation area is similar to AFF. But due to competition for land between the two carbon removal options, afforestation area is reduced in favor of bioenergy area in the USA and China in AFF+BECCS. There are several reasons why the USA, China and Europe are the main bioenergy producers. We provide insight in subsection '*Bioenergy CCS and the role of yield increases*' along with figure 5.

Carbon dynamics

In the BAU scenario, CO_2 emissions from the land-use system accumulate to 177 $GtCO_2$ until 2095 (figure 4). The peak in mid-century is mainly caused by deforestation, while the following decline in CO_2 emissions is due to ecological succession on abandoned agricultural land. In the mitigation scenarios, the described land-use dynamics lead to net carbon removal from the atmosphere. More precisely, carbon is detracted from the atmosphere by photosynthesis and is either

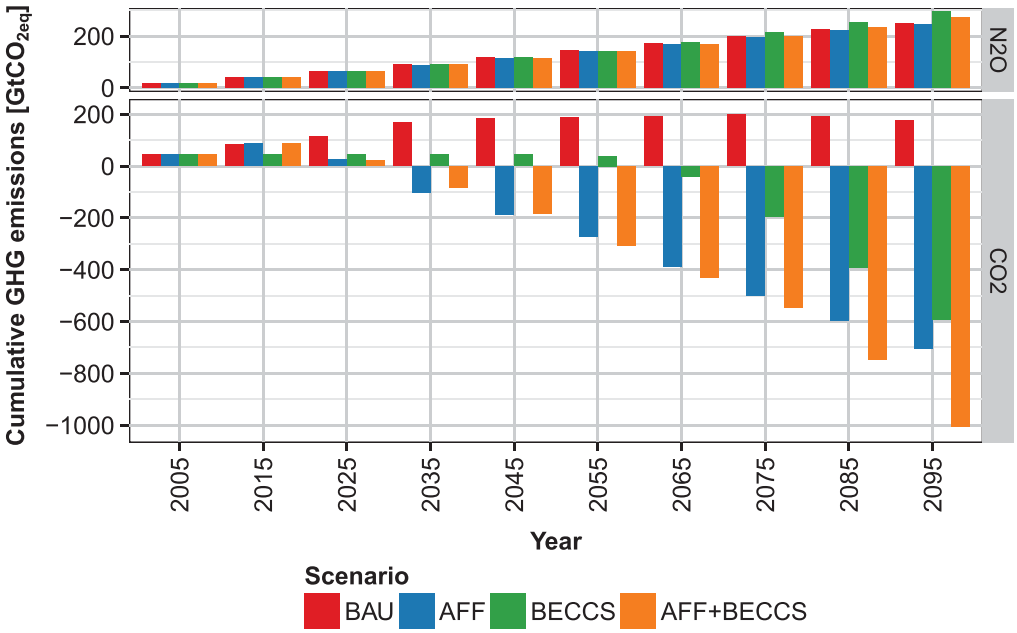

Figure 4. Time-series of global cumulative N_2O and CO_2 emissions ($GtCO_{2eq}$) from the land-use system for BAU, AFF, BECCS and AFF +BECCS.

biologically sequestered via afforestation or geologically sequestered via bioenergy CCS. In AFF, land conversion into afforestation area increases cumulative CO_2 emissions in 2015, followed by continuous carbon removal of about 10 $GtCO_2\,yr^{-1}$ throughout the 21st century. Until 2095, carbon removal in AFF accumulates to 703 $GtCO_2$. In BECCS, cumulative CO_2 emissions are almost constant until bioenergy CCS becomes cost-efficient as mitigation strategy in 2065 at GHG prices of 270 $\$/tCO_{2eq}$. From 2065, carbon removal in BECCS is about 20 $GtCO_2\,yr^{-1}$, which cumulates to 591 $GtCO_2$ until 2095. In AFF+BECCS, carbon dynamics are similar to AFF until bioenergy CCS becomes competitive as mitigation option in addition to afforestation in 2055. Carbon removal in AFF+BECCS is about 25 $GtCO_2\,yr^{-1}$ in from 2065 to 2095, which results in cumulative carbon removal of 1000 $GtCO_2$ until 2095. In 2095 in BECCS and AFF+BECCS, the constraint on the annual geological carbon injection rate is binding (20 $GtCO_2\,yr^{-1}$), while cumulative carbon storage capacity (3960 $GtCO_2$) would last for approximately another 150 years.

Bioenergy CCS and the role of yield increases

Contrary to afforestation, the carbon removal rates per unit area of bioenergy CCS can be enhanced through yield increases. Figure 5 illustrates the potential annual carbon removal rates during the optimization for 1995 and 2095 in the AFF+BECCS scenario, i.e. the annual carbon removal rates shown here represent realizable, but not necessarily realized, carbon removal rates (compare to figure 3). In the remainder of this subsection, we talk about annual realizable carbon removal rates. In 1995, afforestation shows higher carbon removal rates than bioenergy CCS in the majority of cells, with highest carbon removal rates in the tropics (about 20 $tC\,ha^{-1}\,yr^{-1}$). In 2095, the picture is fundamentally different due to yield-increasing technological change on bioenergy crops, which increases carbon removal per unit area. In AFF +BECCS, average yield-increase throughout the century is at 1.38% yr^{-1} (figure S10), which more than triples initial bioenergy yields until 2095 (figure S5). By end-of-century bioenergy CCS exceeds the carbon removal rates of afforestation in the tropics (about 25–30 $tC\,ha^{-1}\,yr^{-1}$). However, bioenergy production does not take place in the tropics but mainly in the USA, China and Europe (figure 3) for three reasons. First of all, the USA and China exhibit higher carbon removal rates in 2095 (about 30–40 $tC\,ha^{-1}\,yr^{-1}$) compared to the tropics. Second, the tropical regions are the most attractive places for afforestation. Third, bioenergy production relies on transport infrastructure, which is much more sophisticated in Europe, USA and China than in the tropics (Nelson 2008). Bioenergy yield gains go along with increased fertilizer use, which drives N_2O emissions. In 2095, cumulative N_2O emission in BECCS and AFF +BECCS are about 30–50 $GtCO_{2eq}$ higher compared to BAU or AFF (figure 4), although N_2O emissions are penalized by the GHG tax.

Sensitivity analysis

In order to test the stability of our results, we perform sensitivity analyses with crucial exogenous parameters (table 3). Figure 6 shows the results in terms of land and carbon dynamics at the global level. Regional results can be found in figure S8.

The constraint on the annual geological carbon injection rate is crucial for the scenarios with bioenergy CCS. With 1 $GtCO_2\,yr^{-1}$ and 20 $GtCO_2\,yr^{-1}$ the constraint is binding, which indicates that the mitigation potential of bioenergy

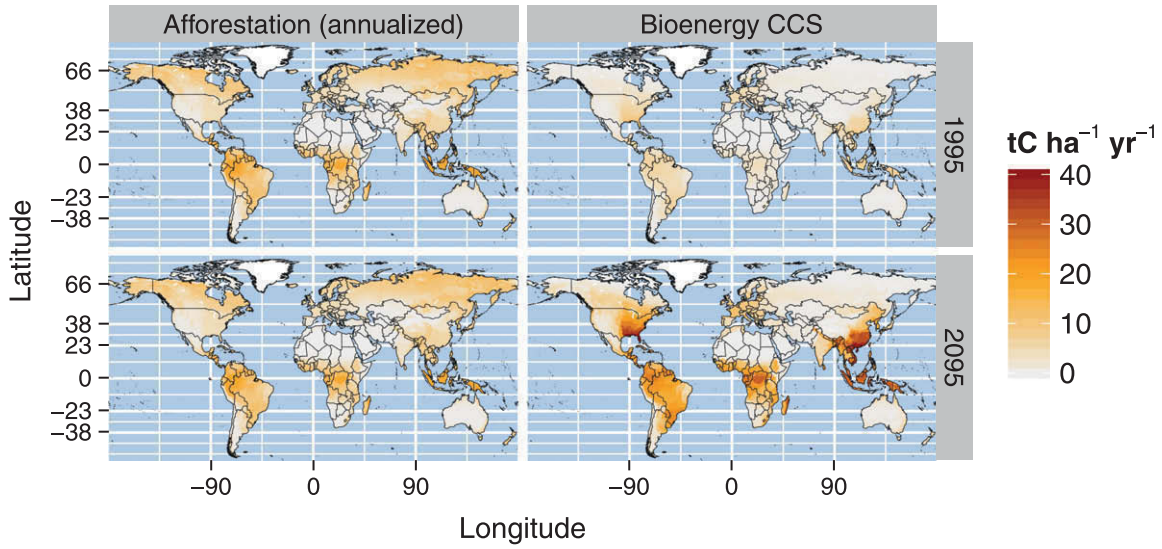

Figure 5. Grid-cell specific illustration of potential annual carbon removal rates from afforestation and bioenergy CCS for 1995 (top) and 2095 (bottom) in the AFF+BECCS scenario (tC ha^{-1} yr^{-1}). Annual carbon removal due to afforestation is calculated as average annual carbon increase in vegetation, litter and soil over a period of 30 years. Annual carbon removal due to bioenergy CCS is based on B2H2 conversion technology in combination with dedicated herbaceous bioenergy crops. Bioenergy yields are converted to carbon densities using a conversion factor of 0.45 t C/t DM. Grid-cell specific results are obtained by disaggregation of cluster level results (each grid cell is assigned the value of the cluster it belongs to).

CCS is mostly limited by the annual geological carbon injection rate. However, with a potential of 396 $GtCO_2$ yr^{-1} the constraint is not binding, which indicates that the potential of bioenergy CCS is also limited by other factors like land availability and costs associated with bioenergy production. Bioenergy production is 530 EJ yr^{-1} in HIGH, compared to 237 EJ yr^{-1} in DEFAULT and 4 EJ yr^{-1} in LOW (figure S7). In the combined setting, AFF+BECCS, land demand is similar for all parameter settings, while the difference in carbon removal is about 500 $GtCO_2$. This can be explained by considering that in the combined setting in HIGH average annual yield increases are at 1.5% yr^{-1} compared to 1.25% yr^{-1} in LOW (figure S10).

The carbon removal potential is highly sensitive to different levels of the GHG tax, which is the only driver for land-based mitigation in this study. In general, different GHG tax trajectories influence the point in time when bioenergy CCS and afforestation are cost-efficient, which translates into different mitigation potentials in 2095. While bioenergy CCS is cost-efficient starting from carbon prices of 165 \$/tCO_{2eq}, afforestation emerges as cost-efficient at prices of 6 \$/tCO_{2eq}. Therefore, the impact of changes in the GHG tax trajectory on the mitigation potential is higher in scenarios with bioenergy CCS. In AFF+BECCS, the range of sensitivity for the mitigation potential is about 900 $GtCO_2$. In general, the degree of sensitivity decreases with higher GHG tax levels, especially for afforestation.

Lower annual discount rates (4%) mostly affect the carbon removal potential of bioenergy CCS as lower discount rates facilitate long term investments in R&D translating into agricultural yield increases. On the contrary, higher discount rates (10%) increase the charges for credit, which is reflected in average annual technological change rates of 1.25% yr^{-1} in HIGH and 1.45% yr^{-1} in LOW (figure S10). The range of sensitivity for the mitigation potential is about 200 $GtCO_2$ for BECCS and 300 $GtCO_2$ for AFF +BECCS.

In terms of land, the time horizon for investment decisions mostly affects afforestation. With a time horizon of ten years, afforestation area accumulates to about 1500 mio ha, while with a time horizon of 30 or 50 years afforestation area is about 3000 mio ha, which translates into a difference in carbon removal of about 300 $GtCO_2$. The sensitivity of afforestation to the time horizon can be explained by recalling the shape of the forest growth curves in figure 3. The mitigation potential of bioenergy CCS is also affected as a shorter lifetime of investments in CCS infrastructure increases the costs associated with bioenergy CCS.

When bioenergy yields are fixed at their initial level, bioenergy CCS is less attractive as mitigation strategy. In BECCS, bioenergy production is reduced to 74 EJ yr^{-1} in LOW compared to 237 EJ yr^{-1} in DEFAULT, which results in a reduction of the mitigation potential of about 500 $GtCO_2$ until 2095. In the combined setting, AFF+BECCS, bioenergy CCS is no longer competitive with afforestation when bioenergy yield are not allowed to increase in the future, which reduces the mitigation potential in LOW compared to DEFAULT by about 300 $GtCO_2$.

The range of sensitivity across different numbers of spatial cluster units (100–500) during the optimization is small for AFF and BECCS (~50 $GtCO_2$), while it is more pronounced in the combined setting (~200 $GtCO_2$). In general, we observe a small trend towards less carbon removal with a higher number of cluster units (figure S12).

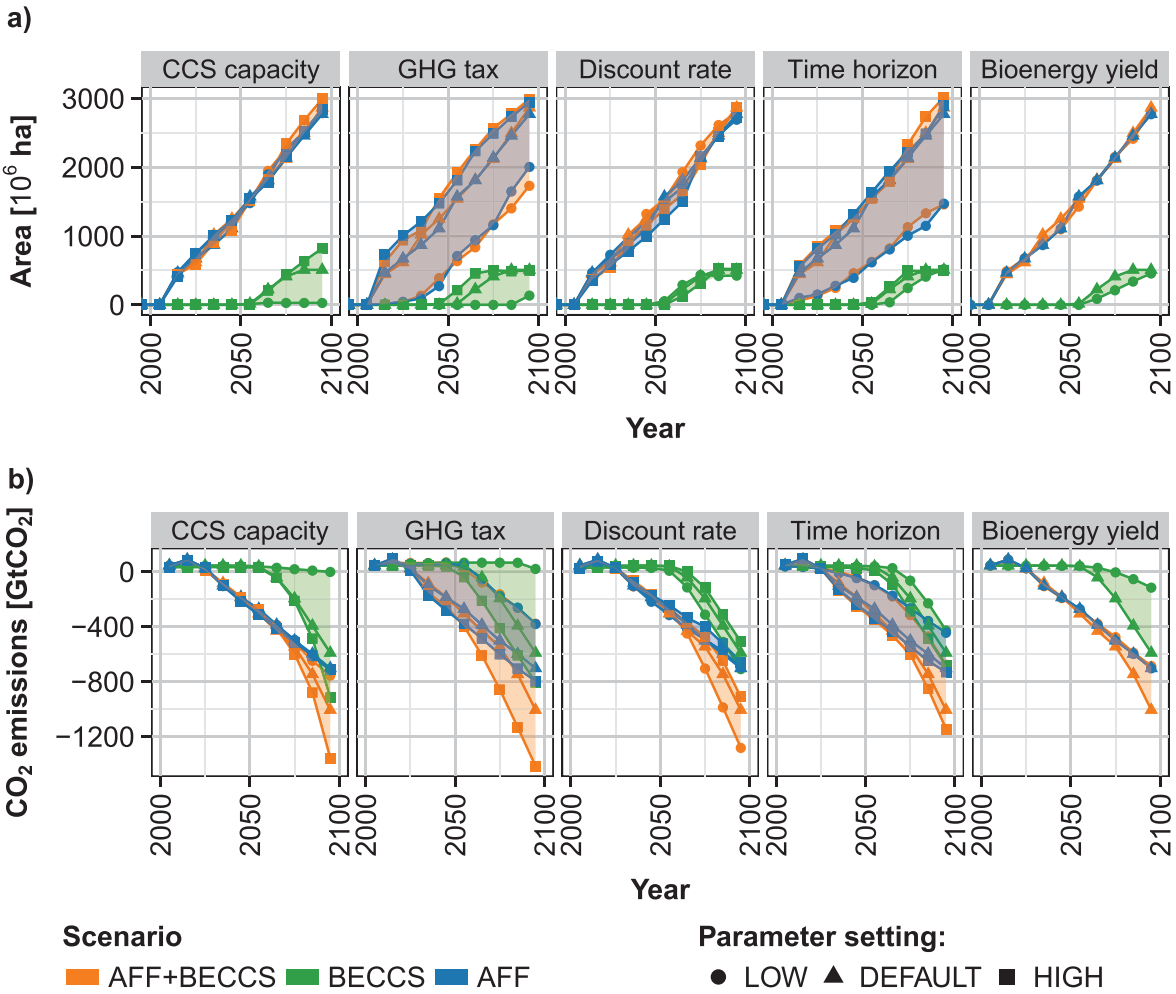

Figure 6. Time-series of sensitivity analysis for AFF, BECCS and AFF+BECCS at the global level. The settings (LOW, DEFAULT, HIGH) for the different parameters (CCS capacity, GHG tax, discount rate, time horizon, bioenergy yield) are described in table 3. The shaded areas span the whole range of sensitivity in the respective scenario in terms of (a) area in use for land-based mitigation (10^6 ha) and (b) cumulative CO_2 emissions ($GtCO_2$).

Discussion and conclusion

In this paper, we investigated the cumulative carbon removal potential in the land-use sector for climate change mitigation scenarios with different land-based mitigation strategies: afforestation, bioenergy CCS and the combination of both. In addition, we tested the sensitivity of our result to changes in crucial exogenous parameters.

As single mitigation strategy, afforestation is cost-efficient at relatively low carbon prices (6 $\$/tCO_{2eq}$), while bioenergy CCS only becomes competitive at higher carbon prices (165 $\$/tCO_{2eq}$). It should be noted that the value of energy produced via bioenergy CCS is disregarded in this study. Instead, the revenue from the GHG tax for carbon removal is considered as the only driver for bioenergy CCS and afforestation. By end-of-century, global area for land-based climate change mitigation is more than five times larger in case of afforestation (~2800 mio ha) compared to bioenergy CSS (~500 mio ha). For bioenergy CCS, our area estimates are comparable to recent IAM studies aiming at ambitious climate change mitigation (Popp *et al* 2014). In general, the limiting factor for land-based mitigation is the availability of land. Besides that, bioenergy production for use with CCS technology is capped by the annual realizable geological carbon injection. Therefore, bioenergy production stabilizes at 237 EJ yr^{-1} by end-of-century, which is within the range of estimated bioenergy deployment levels until 2050 (Chum *et al* 2011). Despite the dissimilarities in land demand, cumulative carbon removal by end-of-century is similar for afforestation (703 $GtCO_2$) and bioenergy CCS (591 $GtCO_2$). This can be explained by considering that, contrary to afforestation, yield-increasing technological change can enhance carbon removal per unit area of bioenergy CCS—at the expense of additional N_2O emission due to increased fertilizer use, which reduces the mitigation effect of bioenergy CCS throughout the century by about 30–50 $GtCO_{2eq}$. In addition, both options, afforestation and bioenergy CCS, benefit from area reductions in the agricultural sector due to yield increases. Based on several IAM studies, Tavoni and Socolow (2013) identify a range for cumulative

carbon removal until 2100 of 200–700 $GtCO_2$ for afforestation and of 460–910 $GtCO_2$ for bioenergy CCS. Our estimates for afforestation are at the upper end of this range, while for bioenergy CCS estimates are in the middle. The combination of afforestation and bioenergy CCS leads to higher cumulative carbon removal (1000 $GtCO_2$ in AFF+BECCS) compared to scenarios with single mitigation strategies. But carbon removal in the combined setting is less than the sum of carbon removal in the standalone settings, indicating that afforestation and bioenergy CCS compete for land. Although bioenergy area is halved compared to the standalone setting, biomass production and thereby carbon removal due to bioenergy CCS is maintained—at the cost of additional yield increases. The sensitivity analysis shows that land-based mitigation is very sensitive to different levels of GHG taxes. Different GHG tax trajectories influence the point in time when bioenergy CCS and afforestation are cost-efficient, which results in different mitigation potentials in 2095. Moreover, the mitigation potential of bioenergy CCS highly depends on the development of future bioenergy yields and the availability of geological carbon storage, while for afforestation projects the length of the crediting period is crucial.

Although in 2095 the constraint on annual geological carbon injection is binding (20 $GtCO_2\,yr^{-1}$), geological carbon injection could continue for approximately another 150 years at this rate after 2095 until the cumulative carbon storage capacity is exhausted. On the other hand, carbon removal rates due to afforestation can be expected to decline when no more land for afforestation is available and forests reach maturity. Therefore, in the longer run bioenergy CCS could probably remove more carbon from the atmosphere than afforestation. Experimental model runs until 2145 support this hypothesis (figure S11). Zeng (2008), Zeng *et al* (2013) suggest an alternative carbon sequestration strategy related to afforestation. Trees could be harvested regularly and buried underground in trenches, which would prevent the decomposition of the wood for long periods (100–1000 years). Using this approach, a piece of land could be used several times for afforestation, which would probably increase the competiveness of afforestation as mitigation strategy.

Competition for land mostly takes place in the USA, China and Europe, which are attractive for both mitigation strategies. By end-of-21st-century, afforestation area is found in many world regions, especially in the tropics, while bioenergy production concentrates in the USA, China and Europe. Large-scale land-based mitigation might change the albedo of land surfaces, leading to biophysical impacts on the climate system (Vuuren *et al* 2013). Specifically in snow-covered areas of the northern hemisphere, reduction of albedo due to afforestation might jeopardize the mitigation effect of carbon removal from the atmosphere, which could result in a net warming effect (Schaeffer *et al* 2006, Bala *et al* 2007, Jackson *et al* 2008, Jones *et al* 2013). In this study, we disregard such feedbacks on the climate system. According to our results, afforestation area is found in boreal regions that might be affected by the albedo effect. However, carbon removal rates due to afforestation in these regions are low compared to the tropics, where afforestation area is found in large part.

Our results indicate that land-based mitigation primarily expands at the expense of cropland and pastureland as the conversion of forestland or other carbon-rich natural vegetation is not attractive due to the GHG tax. Moreover, timberland is not available for conversion as it is reserved for wood production (about 1270 mio ha globally). When the revenue from carbon removal due to bioenergy CCS or afforestation exceeds the revenue from forestry products, timberland might become a source of feedstock for bioenergy or part of an afforestation project. Moreover, if price-induced changes in consumption would be taken into account, competition between food production and land-based mitigation is likely to reduce food demand due to increasing prices for food production, which would result in more area available for land-based mitigation. Therefore, the area available for land-based mitigation might be underestimated in this study to the extent forestry and agricultural demand could be reduced. However, for bioenergy CCS the constraint on geological carbon injection (20 $GtCO_2\,yr^{-1}$ globally) is binding at the end of the 21st century. Hence, more available land is rather to increase carbon removal due to afforestation than due to bioenergy CCS.

In order to maintain the provision of food and feed besides land-based mitigation, yield increases in the agricultural sector would be needed to compensate for the reduction in agricultural land. In addition, the mitigation potential of bioenergy CCS relies on future increases of bioenergy yields. According to our results, land-based mitigation measures would require average annual yield increases of 1–1.38% yr^{-1} globally throughout the 21st century, which is at the lower end of historic yield increases. In the last 40 years, corn and soybean yields grew at a rate of 1.4–1.8% yr^{-1} in the USA (Egli 2008). At the global level corn yields increased by a factor of 2.5 between 1961 and 2007 (Edgerton 2009), which translates into average annual yield increases of 2% yr^{-1}. However, it is unclear if these rates of yield increase can be maintained in the future and to which extent bioenergy yields will benefit from future yield-increases in the agricultural sector. On the one hand side, measures that increase the harvestable storage organ carbon pool are specifically designed for conventional crops, while the purpose of dedicated bioenergy crops is maximum carbon accumulation across all carbon pools including stems and leaves. On the other side, conventional crops are already at high breeding levels, while breeding in bioenergy crops just started (Głowacka 2011). For instance, the estimated potential yield increase of miscanthus by 2030 is 100% (Chum *et al* 2011, p 277). In addition, also the starting point for potential bioenergy yield increases is uncertain. The range of estimates for current lignocellulosic bioenergy yields is 120–280 GJ ha^{-1} in Europe and 150–415 GJ ha^{-1} in South America (Chum *et al* 2011, p 234). While initial bioenergy yields in MAgPIE are at the lower end of these estimates (125 GJ ha^{-1} in Europe, 150 GJ ha^{-1} in Latin America; see table 1), higher initial bioenergy yields would probably render bioenergy CCS cost-efficient at lower carbon prices.

Other studies investigating bioenergy CCS and afforestation as mitigation strategies (Wise *et al* 2009, Calvin *et al* 2014, Edmonds *et al* 2013) feature a detailed representation of the energy sector. In this study, we deliberately focus on the mitigation potential of bioenergy CCS and disregard the various usage options of energy within the energy sector. Using this simplified approach, we show that bioenergy CCS could contribute to climate change mitigation in a cost-efficient way even if only the carbon removal part is valued. Another important difference concerns the assumptions about future agricultural yield increases. In other studies, yield increases follow exogenous trajectories, while investment in yield-increasing technological change in MAgPIE is a variable. Therefore, the land-use system in MAgPIE can endogenously adapt to different situations, which is for instance reflected in the amount of land used for afforestation (~2800 mio ha) compared to other studies (~1000 mio ha) (Wise *et al* 2009, Calvin *et al* 2014).

The bioenergy CCS technology is still under development and currently not applied at large economic scale (Bennaceur *et al* 2008). Furthermore, the range of estimates for geological carbon storage capacities is huge (100–200 000 $GtCO_2$) (Bradshaw *et al* 2007). Therefore, the future economic and technical feasibility of bioenergy CCS is highly uncertain. Moreover, missing social acceptance of the bioenergy CCS technology can hinder political implementation (Johnsson *et al* 2010, Knopf *et al* 2010). On the contrary, afforestation as mitigation strategy for carbon removal can be applied immediately, as it is basically planting trees. Besides that, social acceptance of afforestation is unlikely to be problematic, since forests can provide a number of ecosystem services besides carbon sequestration (e.g. water purification, biodiversity conservation, recreation) (Barlow *et al* 2007, Onaindia *et al* 2013). Valuing these ecosystem system services in addition to carbon sequestration could increase incentives for afforestation. Nevertheless, monitoring carbon stock dynamics is critical for the implementation of afforestation as mitigation strategy (Calvin *et al* 2014).

We conclude that afforestation could turn the land-use sector from a net source into a net sink of carbon before mid-century. Moreover, our results indicate that early-century afforestation presumably will not negatively impact carbon removal due to bioenergy CCS in the second half of the century. Therefore, the near-term implementation of afforestation as climate change mitigation strategy could increase the likelihood of keeping global warming below two degree above pre-industrial levels (Meinshausen *et al* 2009), while bioenergy CCS could still contribute to climate change mitigation in the second half of the century if economically, institutionally and technically feasible.

Acknowledgments

The research leading to these results has received funding from the European Union's Seventh Framework Programme FP7/2011 under grant agreement n° 282846—LIMITS. The research leading to these results has received funding from the European Union's Seventh Framework Programme FP7/2007–2013 under grant agreement n° 603542—LUC4C. Funding from Deutsche Forschungsgemeinschaft (DFG) in the SPP ED 178/3-1 (CEMICS) is gratefully acknowledged.

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Supplementary Information

Investigating afforestation and bioenergy CCS as climate change mitigation strategies

Florian Humpenöder, Alexander Popp, Jan Philipp Dietrich, David Klein, Hermann Lotze-Campen, Markus Bonsch, Benjamin Leon Bodirsky, Isabelle Weindl, Miodrag Stevanovic, Christoph Müller

1. Additional model description

1.1. MAgPIE regions

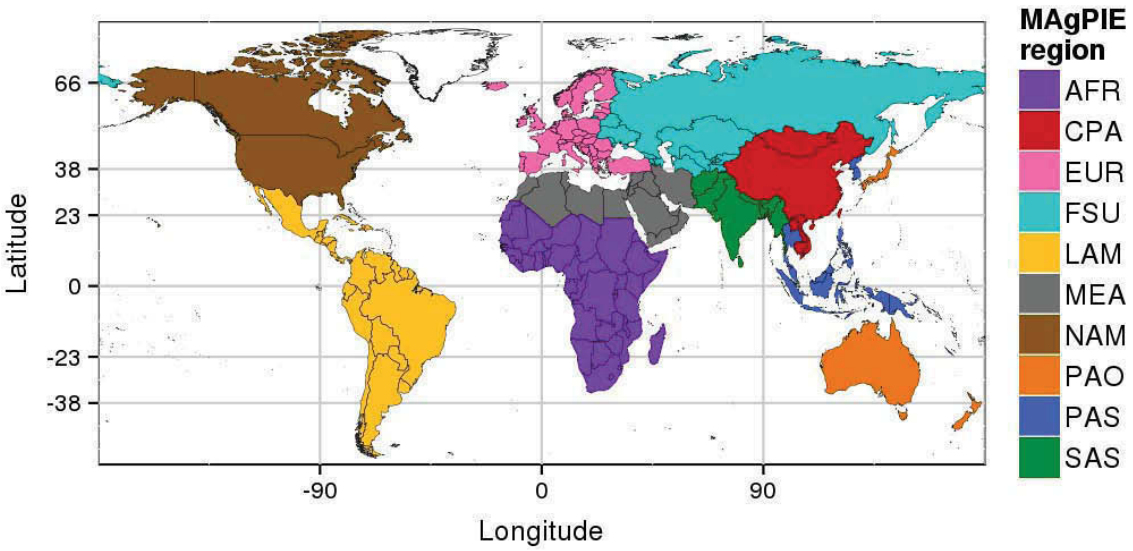

Figure S1. MAgPIE economic world regions.

MAgPIE	Region	SSP
AFR	Sub-Saharan Africa	MAF
CPA	Centrally planned Asia including China	ASIA
EUR	Europe including Turkey	OECD
FSU	States of the former Soviet Union	REF
LAM	Latin America	LAM
MEA	Middle East/North Africa	MAF
NAM	North America	OECD
PAO	Pacific OECD including Japan, Australia, New Zealand	OECD
PAS	Pacific (or Southeast) Asia	ASIA
SAS	South Asia including India	ASIA

Table S1. Abbreviations and names of the 10 economic world regions in MAgPIE, and mapping to the 5 SSP regions used in figure S8.

1.2. Demand

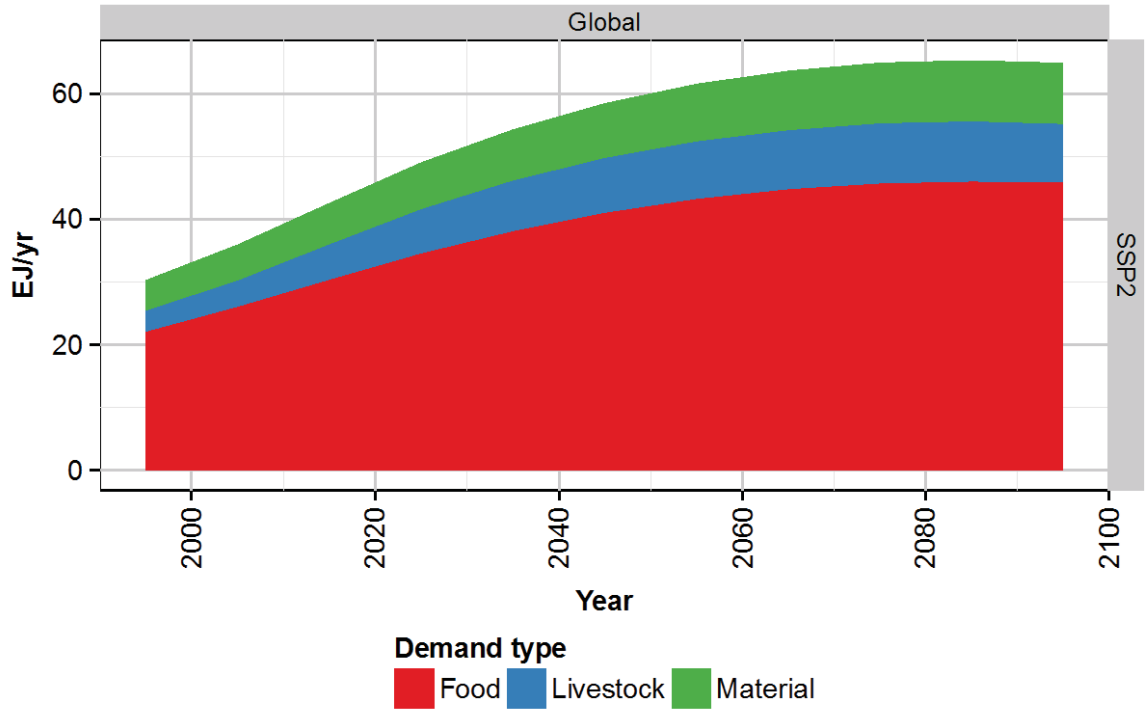

Figure S2. Time-series of food, livestock and material demand based on SSP 2 population and GDP projections (IIASA 2013).

1.3. GHG tax

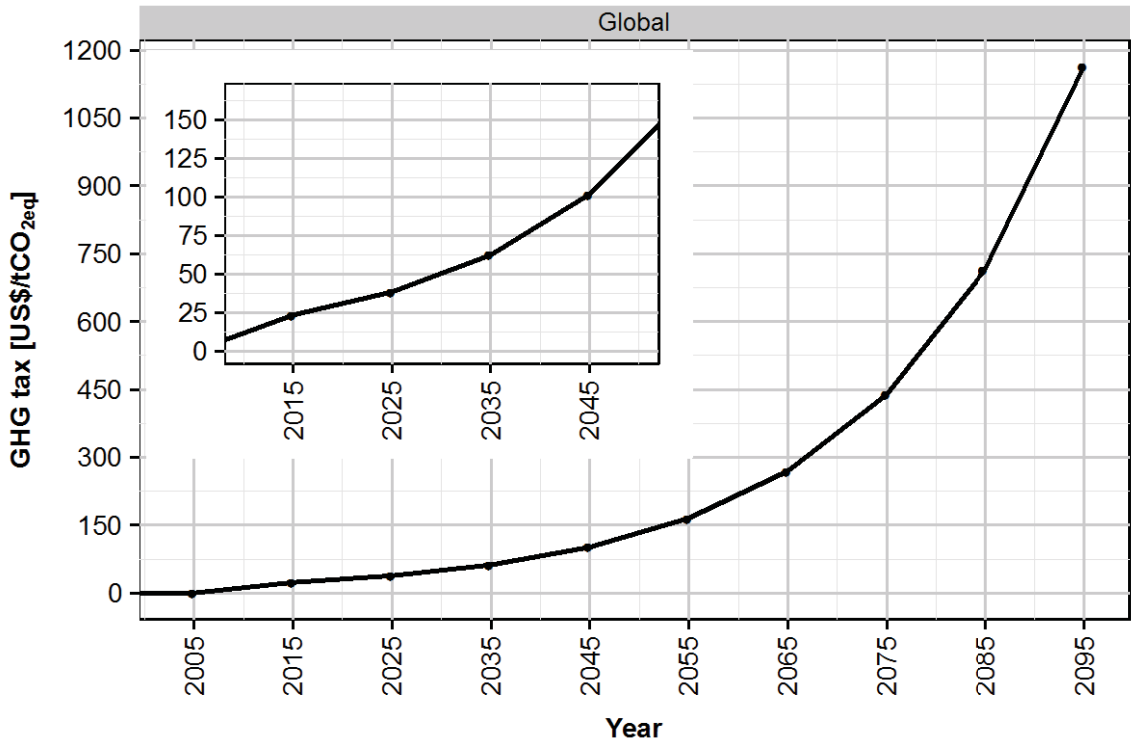

Figure S3. Time-series of assumed global tax on GHG emissions (tax30). Tax30 has a level of 30 $\$/tCO_{2eq}$ in 2020, starts in 2015 and increases by 5% per year.

1.4. Afforestation costs

	AFR	CPA	EUR	FSU	LAM	MEA	NAM	PAO	PAS	SAS
Land conversion	863	868	1849	943	1036	987	2484	2392	968	849
Management	127	6	101	2	52	127	46	13	73	23

Table S2. Regional land conversion costs (US$/ha) and annual management costs (US$/ha/yr) in 1995, based on (Sathaye *et al* 2005).

1.5. Geological carbon storage (CCS capacity)

	AFR	CPA	EUR	FSU	LAM	MEA	NAM	PAO	PAS	SAS	World
$GtCO_2$	229	367	178	917	550	458	458	367	183	183	3890
$GtCO_2$/yr	1.1	1.8	0.9	4.6	2.8	2.3	2.3	1.8	0.9	0.9	19.5

Table S3. Total regional geological carbon storage capacity ($GtCO_2$) and annual realizable injection rate ($GtCO_2$/yr). Based on Szulczewski *et al* (2012) we assume a lifetime of CCS of 200 years. Therefore, we limit the annual injection of carbon to 0.5 %/yr in terms of the total capacity. The total geological carbon storage capacity is based on Bradshaw *et al* (2007).

1.6. Regrowth of natural vegetation

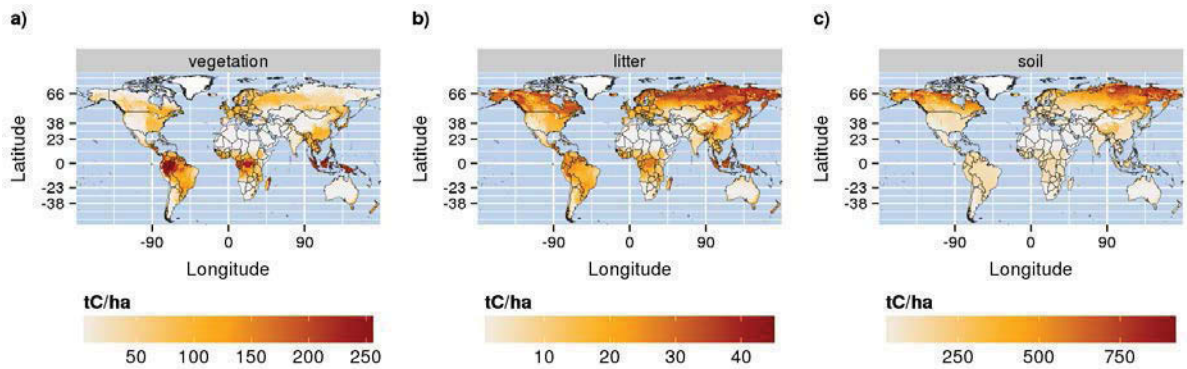

Figure S4. Grid-cell specific carbon density of potential natural vegetation in 1995 (tC/ha) derived from LPJmL for the carbon pools vegetation (a), litter (b) and soil (c). Grid-cell specific results are obtained by disaggregation of cluster level results (each grid cell is assigned the value of the cluster it belongs to).

Parameters

vegc	Age-class dependent vegetation carbon density (tC/ha)
vegc_max	Maximum vegetation carbon density (tC/ha) (Figure S4a)
k	0.075 for tropical, 0.050 for temperate, 0.033 for temperate-boreal and 0.022 for boreal climate regions. Calculated using the methodology described in Gadow and Hui (2001) and climate region specific data on Mean Annual Increment (MAI) and MAI culmination age (IPCC 2006)
m	m=3 (Murray and von Gadow 1993)
litc	Age-class dependent litter carbon density (tC/ha)
litc_max	Maximum litter carbon density (tC/ha) (Figure S4b)
soilc	Age-class dependent soil carbon density (tC/ha)
soilc_max	Maximum soil carbon density (tC/ha) (Figure S4c)
soilc_start	Soil carbon density of former land-use (tC/ha) (*soilc_start* < *soilc_max*)
c_density_ac	Age-class dependent vegetation, litter and soil carbon density (tC/ha)

Indices

j	Cluster (1-500)
ac	Age-classes (1-300)

Vegetation carbon density based on Chapman-Richards volume growth model (Murray and von Gadow 1993, Gadow and Hui 2001) (Figure 1 in manuscript):

$$vegc_{j,ac} = vegc\_max_j \times (1 - exp(-k_j \times ac))^m$$

Litter carbon density based on IPCC (2000)

$$if\ ac \leq 20: litc_{j,ac} = litc\_max_j \times 1/20 \times ac$$
$$if\ ac > 20: litc_{j,ac} = litc\_max_j$$

Soil carbon density based on IPCC (2000)

$$if\ ac \leq 20: soilc_{j,ac} = (soilc\_max_j - soilc\_start_j) \times 1/20 \times ac$$
$$if\ ac > 20: soilc_{j,ac} = soilc\_max_j - soilc\_start_j$$

Carbon density of all carbon pools

$$c\_density\_ac_{j,ac} = vegc_{j,ac} + litc_{j,ac} + soilc_{j,ac}$$

1.7. Economic incentive for land-based climate change mitigation in MAgPIE

The MAgPIE model structure is described in several publications (Lotze-Campen *et al* 2008, Popp *et al* 2010, Dietrich *et al* 2014). This section shows how the objective function of MAgPIE is modified to create an incentive for land-based climate change mitigation. The objective function has been extended for GHG emission costs, which can become negative (i.e. a cost reduction) if negative carbon emissions from afforestation or bioenergy CCS are rewarded by the GHG tax.

Variables

If not indicated otherwise, variables are defined for the range $[0, \infty]$. Variables might be subject to constraints (e.g. land).

X $[-\infty, \infty]$	Variable of the objective function (mio $/yr)
lu_costs	Total costs of land-use related activities (mio $/yr); includes all costs for the production of food, material, livestock and biomass. Considered costs types: land conversion costs, factor requirement costs (capital, labour, fertilizer), transportation costs and cost for yield-increasing technological change.
emis_costs $[-\infty, \infty]$	Total GHG emission costs (mio $/yr)
aff_costs	Total afforestation costs (mio $/yr)
beccs_costs	Total bioenergy CCS technology costs (mio $/yr)
n2o_emis	Total n2o emission (Mt N2O/yr)
ch4_emis	Total ch4 emission (Mt CH4/yr)
co2_emis $[-\infty, \infty]$	Total co2 emission (Mt CO2/yr)
c_emis_land $[-\infty, \infty]$	Total carbon emissions from the land-use system (Mt C/t)
c_emis_aff_exp $[-\infty, 0]$	Total expected carbon emissions from afforestation (Mt C/t)
c_emis_beccs $[-\infty, 0]$	Total carbon emissions from bioenergy CCS (Mt C/yr)
c_stock	Carbon stock (Mt C)
land	Total land for different land types (mio ha)
land_aff	Afforestation area (mio ha)
yield_bio	Bioenergy yield (tDM/ha/yr); can be increased due to endogenous technological change
c_removal_beccs	Carbon removal through bioenergy CCS (Mt C/yr)
bio_prod_reg	Regional biomass production (mio tDM/yr)
bio_use_reg	Regional biomass use (mio tDM/yr)
energy_costs	Total energy system costs (mio $/yr)
ccs_costs	Total CCS costs (mio $/yr)

Parameters

r	Time preference rate (7%/yr)
n	Time horizon for annualisation (30 yrs)
c_density	Carbon density for different land types (tC/ha) (based on LPJmL)
c_density_ac	Age-class dependent vegetation, litter and soil carbon density (tC/ha) for regrowth of natural vegetation (see section 1.6)
ghg_tax	Tax on GHG emissions ($/t$CO_{2eq}$) (see Figure S3)
annuity	Factor for annualisation of future cash flows
lndc	Afforestation land conversion costs ($/ha) (see section 1.4)
manc	Afforestation management costs ($/ha/yr) (see section 1.4)
tDMtoC	Conversion factor from t DM to t C (0.45 t C/t DM)
ccs_storage_potential	Regional CCS storage potential (Mt C) (see section 1.5)
injection_rate	CCS injection rate 0.5%/yr (see section 1.5)
injection_costs	CCS injection and transportation costs (33 $/tC)
lcoe	Levelized cost of energy for B2H2 (8 $/GJ)
tDMtoGJ	Conversion factor from t DM to GJ (18 GJ/t DM)
conv_eff	B2H2 conversion efficiency (55%)
cap_rate	B2H2 CCS capture rate (90%)

Indices

t	Time steps (11)
i	MAgPIE world regions (10)
j	Cluster (500)
l	Land types (food, bio, past, forest, aff, other)
ac	Age-classes (300)

Sets

cell(i,j)	Mapping of regions to cluster

$$annuity = \frac{1-(1+r)^{-n}}{\frac{r}{1+r}}$$

Objective function of MAgPIE

Minimization of X_t for each t (recursive dynamic optimization)

$$X_t = lu\_costs_t + ghg\_emis\_costs_t + aff\_costs_t + beccs\_costs_t$$

Conversion to CO_{2eq} is based on GWP100 (IPCC 2013)

$$ghg\_emis\_costs_t = (n2o\_emis_t \times 265 + ch4\_emis_t \times 28 + co2\_emis_t \times 1) \times ghg\_tax_t$$

c_emis_land and c_emis_aff _exp represent carbon emissions for the whole time step length and are therefore annuitized. Conversion factor from C to CO2: 44/12.

$$co2\_emis_t = ((c\_emis\_land_t + c\_emis\_aff\_exp_t) \times \frac{1}{annuity}) + c\_emis\_beccs) \times 44/12$$

Carbon emissions are calculated as the difference of carbon stocks

$$c\_emis\_land_t = c\_stock_{t-1} - c\_stock_t$$

Carbon stocks are calculated as the product of land and carbon density

$$c\_stock_t = \sum_j \sum_l land_{t,j,l} \times c\_density_{t,j,l}$$

Calculation of carbon density for afforestation land

Constraint for afforestation land

$$land_{t,j,l="aff"} = \sum_{ac} land\_aff_{t,j,ac}$$

Simulation of forest growth by shifting age-classes according to time step length (10 yrs) between time steps (after the end of t-1 optimization and before start of t)

$$land\_aff_{t,j,ac} = land\_aff_{t-1,j,ac-10}$$

Estimate of carbon density for afforestation land pool after shifting age-classes (weighted mean)

$$c\_density_{t,j,l="aff"} = \frac{\sum_{ac} c\_density\_ac_{j,ac} \times land\_aff_{t,j,ac}}{\sum_{ac} land\_aff_{t,j,ac}}$$

Calculation of afforestation expectations and costs

Expected cumulative carbon emissions (negative) beyond the current time step (time step length = 10 yrs) but within the time horizon (n = 30 yrs), due to new afforestation area serve as incentive for the model to invest in afforestation. Emissions due to new afforestation area within the current time step are included in $c\_emis\_land_t$.

$$c\_emis\_aff\_exp_t = \sum_j land\_aff_{t,j,ac=0} \times (c\_density\_ac_{j,ac=10} - c\_density\_ac_{j,ac=n})$$

Regional afforestation area

$$land\_aff_{t,i,ac} = \sum_{cell(i,j)} land\_aff_{t,j,ac}$$

Afforestation costs: land conversion costs for new afforestation area, management costs for total afforestation area

$$aff\_costs_t = \sum_i land\_aff_{t,i,ac=0} \times lndc_i \times \frac{1}{annuity} + \sum_i \sum_{ac} land\_aff_{t,i,ac} \times manc_i$$

Calculation of bioenergy CCS carbon removal (negative emissions) and costs

$$c\_emis\_beccs_t = -\sum_i c\_removal\_beccs_{t,i}$$

Regional biomass production

$$bio\_prod\_reg_{t,i} = \sum_{cell(i,j)} land_{t,j,l="bio"} \times yield\_bio_{t,j}$$

The location of biomass use and geological carbon storage can differ from the location of biomass production

$$\sum_i bio\_use\_reg_{t,i} = \sum_i bio\_prod\_reg_{t,i}$$

Carbon removal through CCS

$$c\_removal\_beccs_{t,i} = bio\_use\_reg_{t,i} \times tDMtoC \times cap\_rate$$

CCS constraint

$$c\_removal\_beccs_{t,i} \leq ccs\_storage\_potential_i \times injection\_rate$$

Costs of bioenergy CCS

$$beccs\_costs_t = energy\_costs_t + ccs\_costs_t$$

$$energy\_costs_t = \sum_i lcoe \times bio\_use\_reg_{t,i} \times tDMtoGJ \times conv\_eff$$

$$ccs\_costs_t = \sum_i c\_removal\_beccs_{t,i} \times injection\_costs$$

2. Detailed results

2.1. Herbaceous bioenergy yields

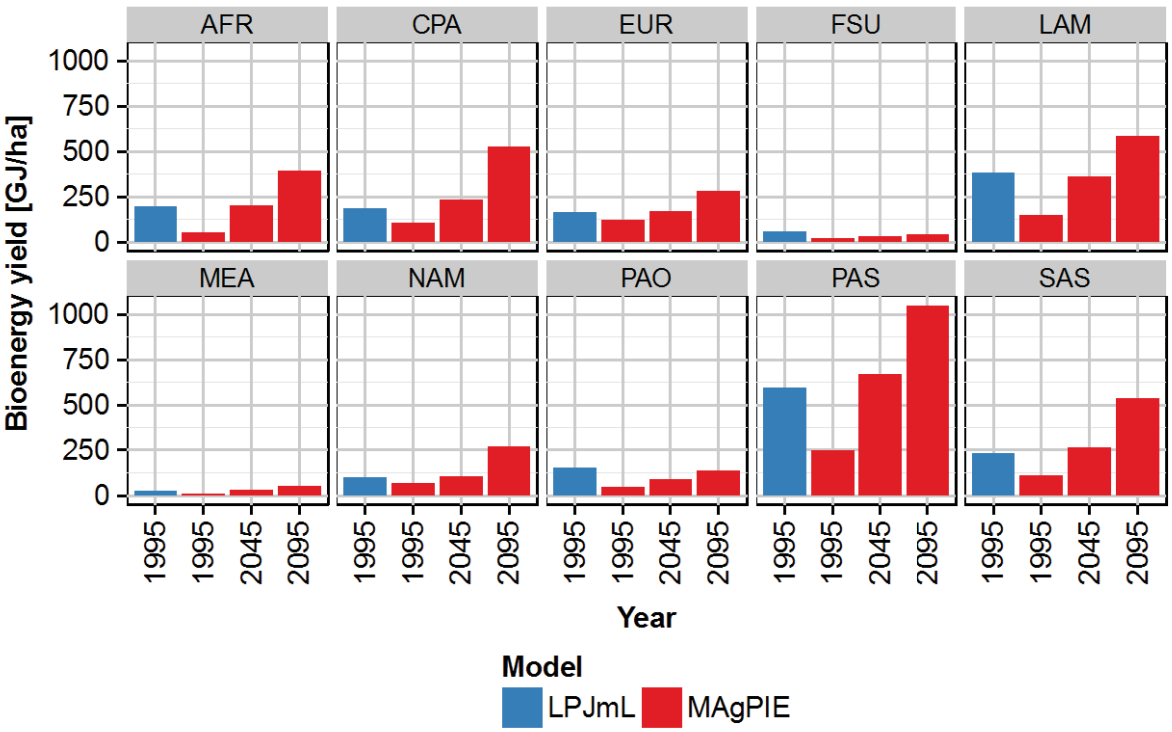

Figure S5. Regional herbaceous bioenergy yields (rain fed only) for LPJmL (1995) and MAgPIE (1995, 2045, 2095) from the AFF+BECCS scenario. LPJmL (Beringer *et al* 2011) represents potential yields, while MAgPIE aims to represent actual yields. Therefore, LPJmL yields are reduced using information about observed land-use intensity (Dietrich *et al* 2012) and agricultural area (FAO 2013). It is assumed that LPJmL bioenergy yields represent yields achieved under highest currently observed land use intensity, which is observed in EUR. Therefore, LPJmL bioenergy yields for all other regions than EUR are reduced proportional to the land use intensity in the given region. In addition, yields are calibrated at the regional level to meet FAO agricultural area in 1995, resulting in a further reduction of yields in all regions. MAgPIE bioenergy yields can exceed LPJmL bioenergy yields over time as endogenous investments in R&D push the technology frontier.

2.2. *Bioenergy production*

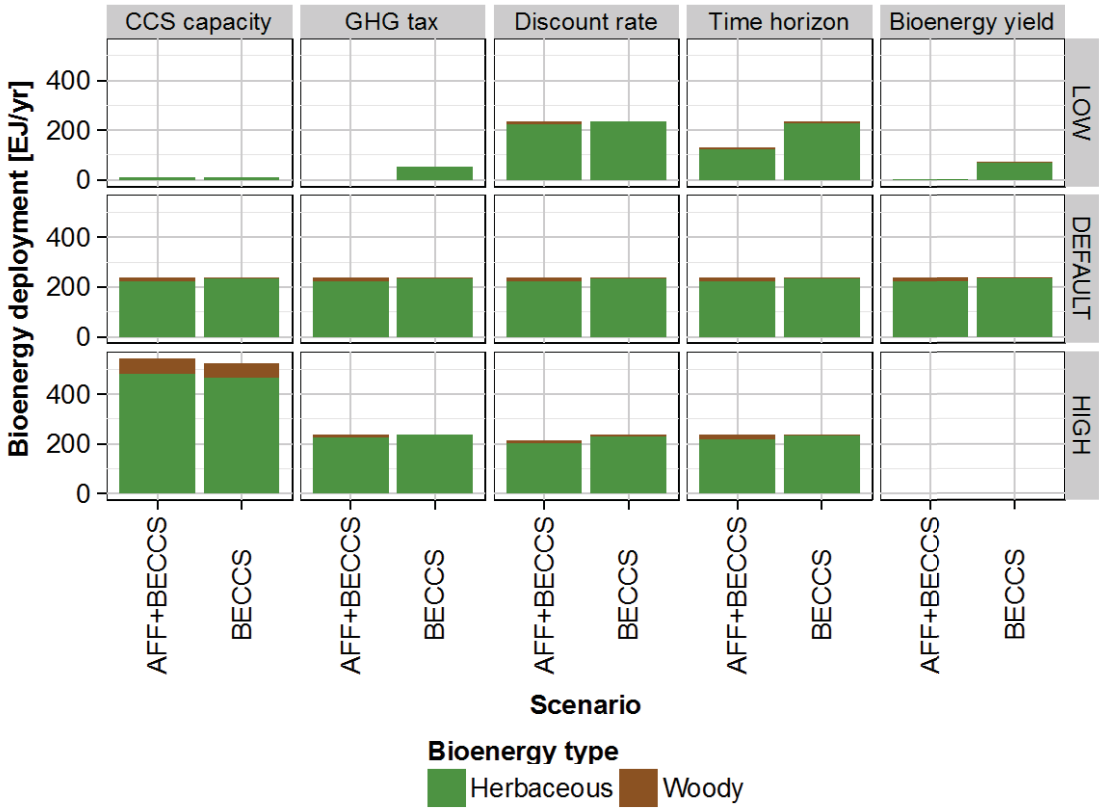

Figure S6. Sensitivity analysis of global dedicated herbaceous and woody bioenergy production (EJ/yr) for BECCS and AFF+BECCS in 2095. The settings (LOW, DEFAULT, HIGH) for the different parameters (CCS capacity, GHG tax, Discount rate, Time horizon, bioenergy yield) are described in table 3 of the main paper.

Figure 7. Time-series of sensitivity analysis of global bioenergy production (EJ/yr) for BECCS and AFF+BECCS. The settings (LOW, DEFAULT, HIGH) for the different parameters (CCS capacity, GHG tax, Discount rate, Time horizon, bioenergy yield) are described in table 3 of the main paper.

2.3. *Area in use for land-based mitigation*

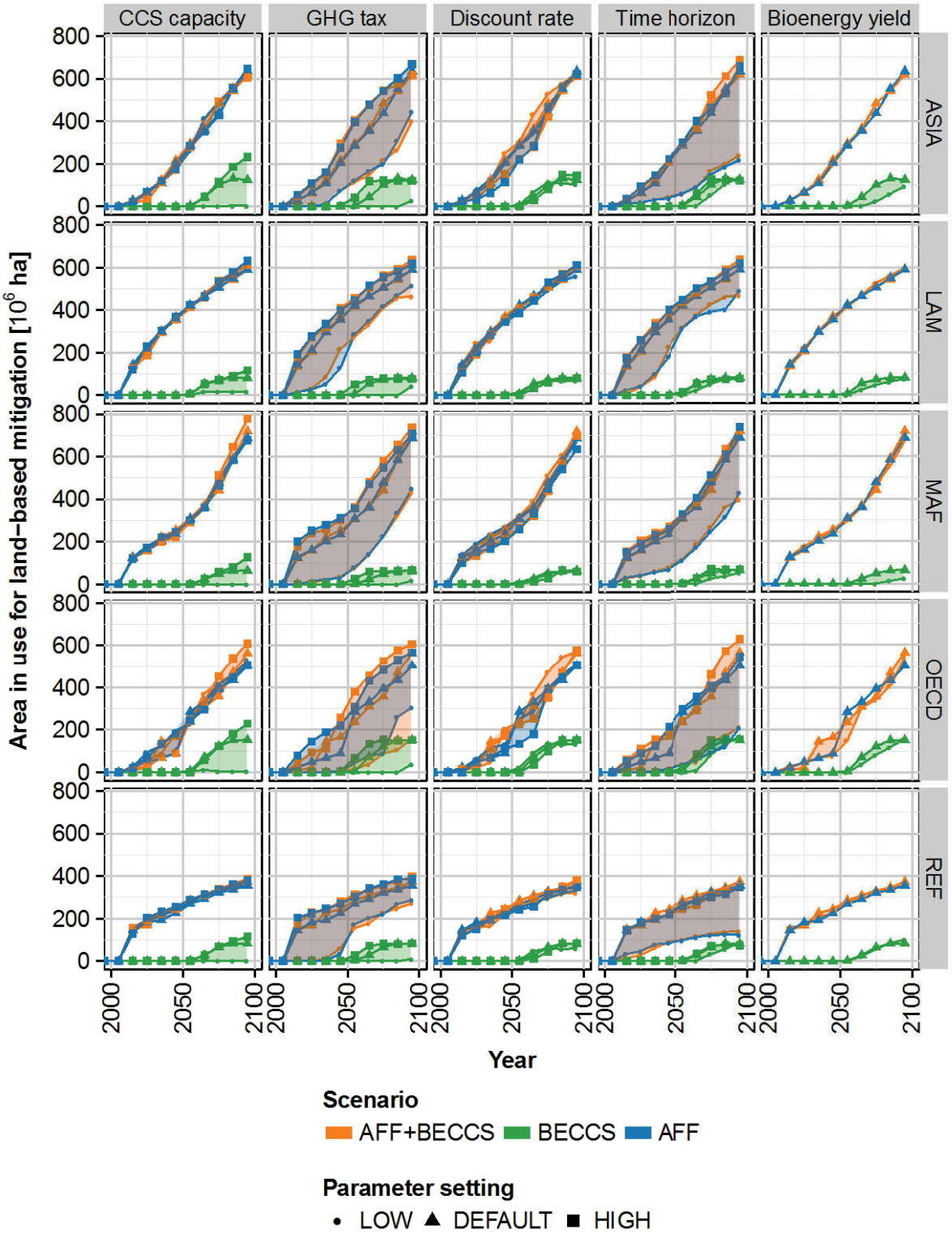

Figure S8. Time-series of sensitivity analysis of bioenergy and afforestation area for 3 scenarios at the regional level. See table S1 for mapping of regions. The settings (LOW, DEFAULT, HIGH) for the different parameters (CCS capacity, GHG tax, Discount rate, Time horizon, bioenergy yield) are described in table 3 of the main paper.

2.4. *Cumulative non-CO_2 GHG emissions from the land-use system (N2O, CH4)*

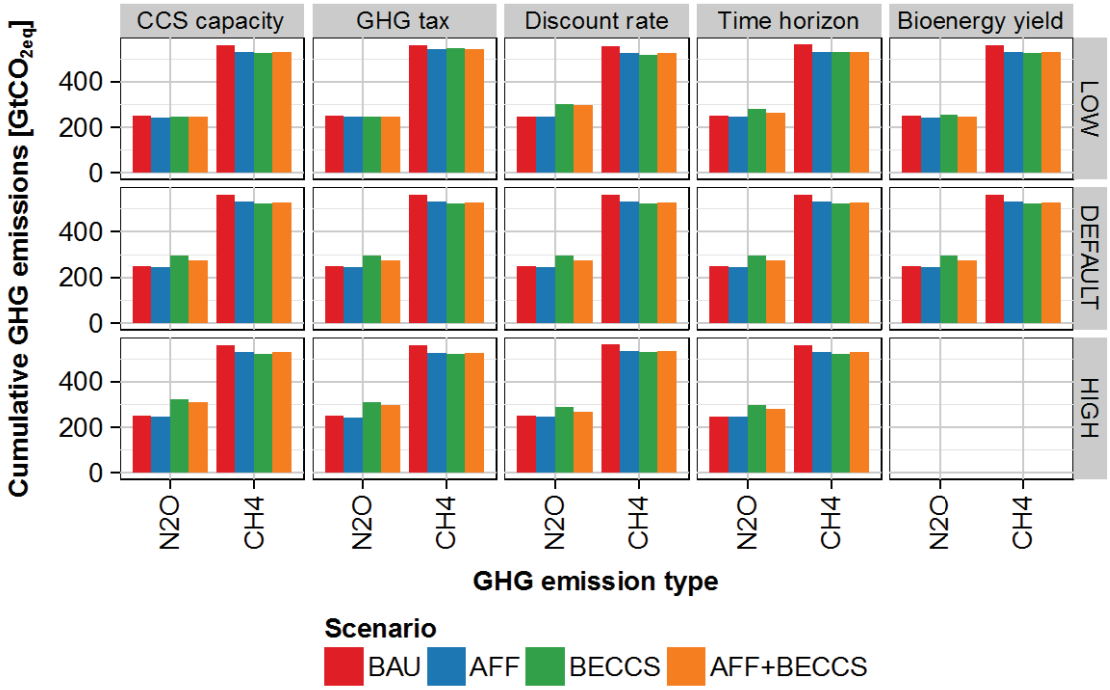

Figure S9. Sensitivity analysis of global cumulative N_2O emissions and CH_4 emissions ($GtCO_{2eq}$) in 2095 from the land-use system for 4 scenarios. The settings (LOW, DEFAULT, HIGH) for the different parameters (CCS capacity, GHG tax, Discount rate, Time horizon, bioenergy yield) are described in table 3 of the main paper.

2.5. *Average annual yield-increasing technological change (TC)*

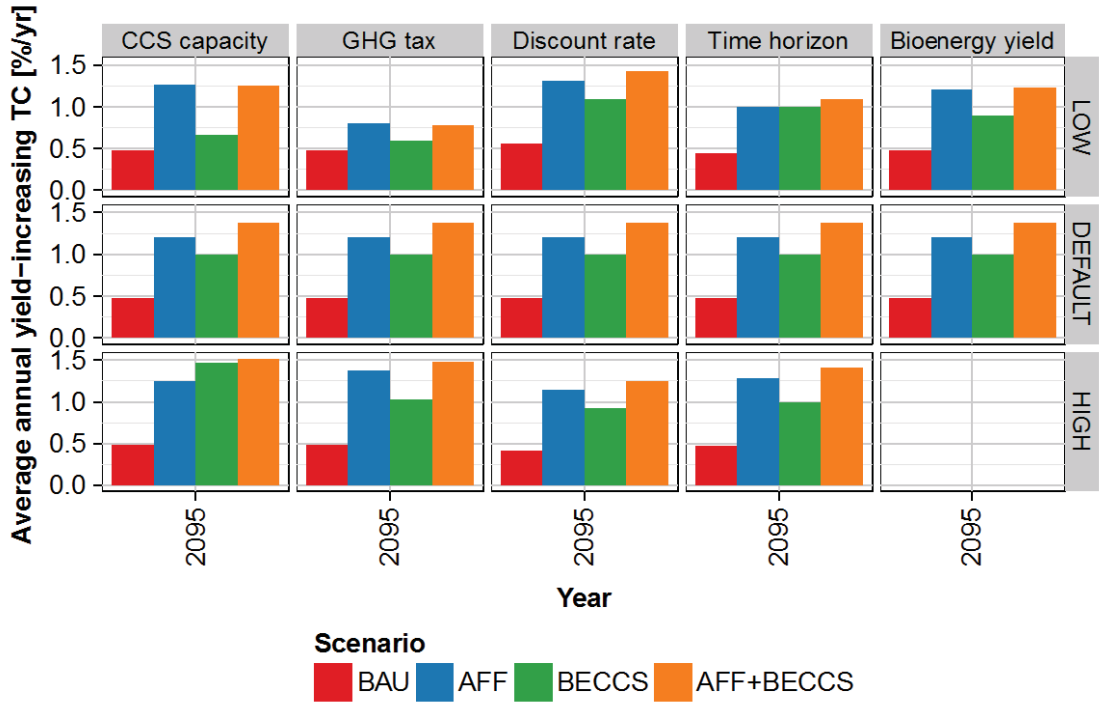

Figure S10. Sensitivity analysis of global average annual yield-increasing technological change (TC) until 2095 for 4 scenarios. For instance, in BECCS (DEFAULT) technological change increases food and bioenergy crop yields on average by 1 %/yr, which almost triples yields within 100 years. The settings (LOW, DEFAULT, HIGH) for the different parameters (CCS capacity, GHG tax, Discount rate, Time horizon, bioenergy yield) are described in table 3 of the main paper.

2.6. *Experimental model results until 2145*

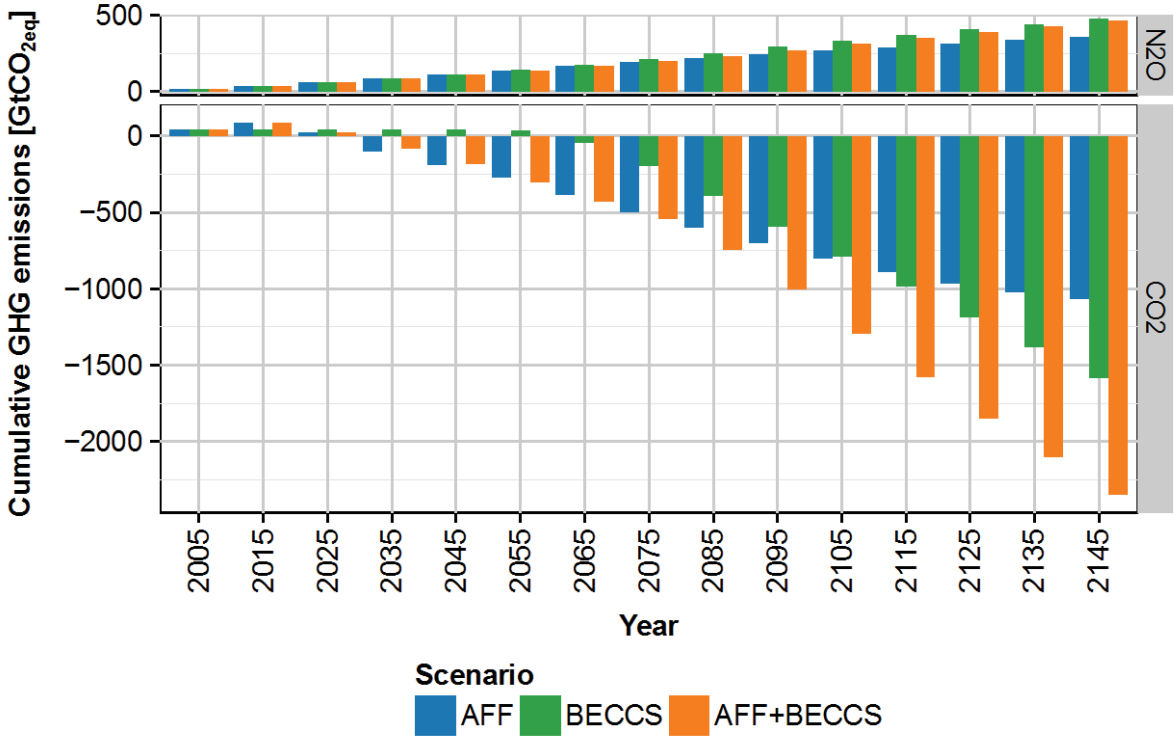

Figure S11. Time-series of global cumulative N_2O and CO_2 emissions ($GtCO_{2eq}$) from the land-use system for AFF, BECCS and AFF+BECCS until 2145. MAgPIE is parameterized to run until 2095. After 2095 we assume that food, material and livestock demand is constant on 2095 levels for this experimental model run.

2.7. Sensitivity analysis with number of cluster units

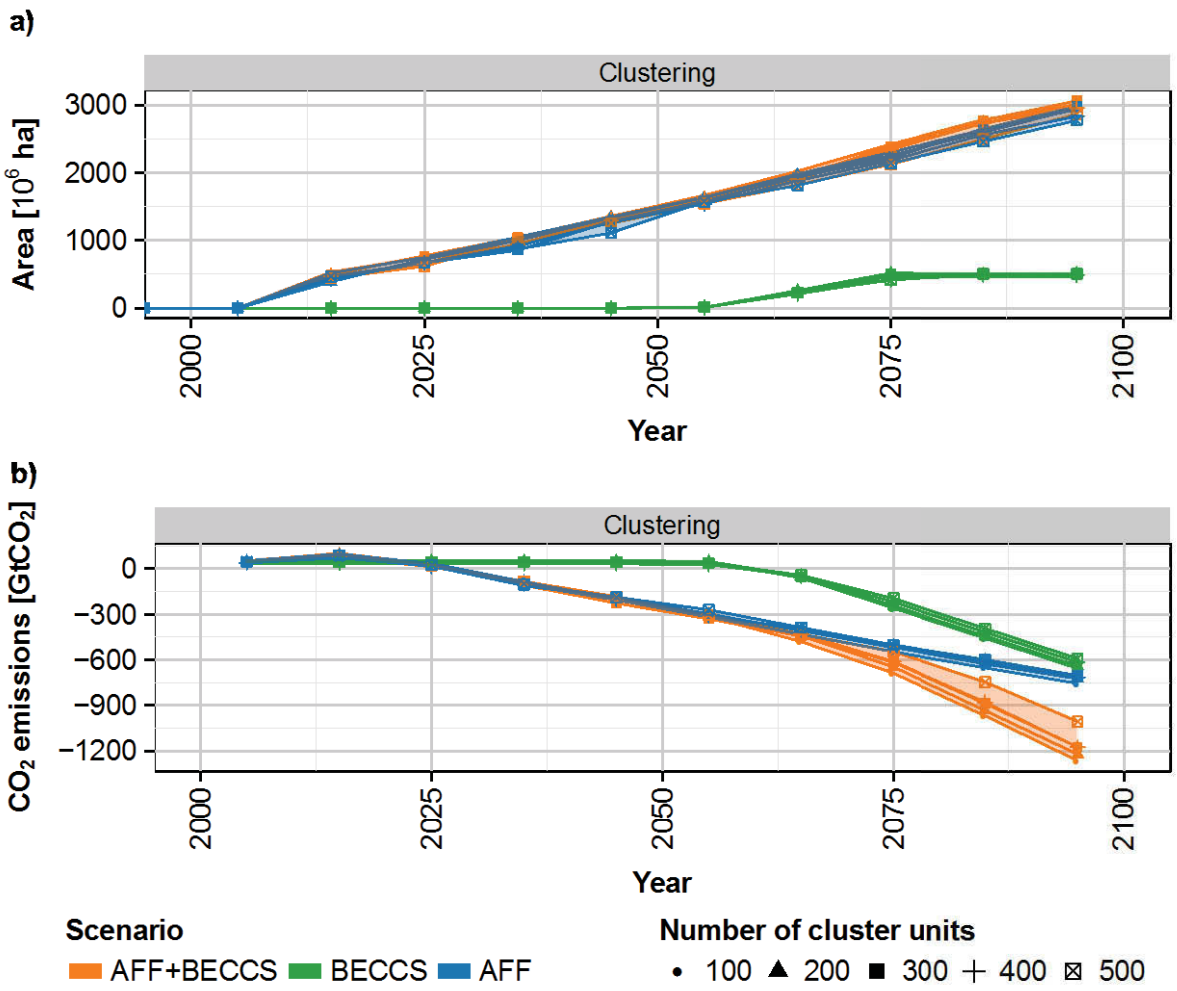

Figure S12. Time-series of sensitivity analysis for AFF, BECCS and AFF+BECCS at the global level for different numbers of cluster units. The shaded areas span the whole range of sensitivity in the respective scenario in terms of a) area in use for land-based mitigation (10^6 ha) and b) cumulative CO_2 emissions ($GtCO_2$).

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Chapter IV

Environmental flow provision: Implications for agricultural water and land-use at the global scale *

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* published in Global Environmental Change Volume 30, Pages 113-132, January 2015, DOI: 10.1016/j.gloenvcha.2014.10.015

Content of Chapter IV: Environmental flow provision: Implications for agricultural water and land-use at the global scale

Global Environmental Change 30 (2015) 113–132

Contents lists available at ScienceDirect

Global Environmental Change

journal homepage: www.elsevier.com/locate/gloenvcha

Environmental flow provision: Implications for agricultural water and land-use at the global scale

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ARTICLE INFO

Article history:
Received 7 April 2014
Received in revised form 8 September 2014
Accepted 21 October 2014
Available online 18 December 2014

Keywords:
Environmental flows
Water-use
Land-use
Land–water-nexus
Global
Model

ABSTRACT

Human activity has led to freshwater ecosystem degradation in the past and is likely to continue doing so if no appropriate protection mechanisms are implemented. One potential protection measure is the reallocation of water from human use to environmental purposes – also called environmental flows. Such reallocation may decrease the availability of irrigation water with possible adverse effects on agricultural production. In this analysis, we provide an initial quantitative estimate of how the allocation of annual volumes of water for environmental flow protection (EFP) might influence the food production system on a global scale. The application of a spatially explicit global land and water-use allocation model (MAgPIE) allows us to explore the effect of EFP on agricultural water withdrawals. We will also examine associated reactions in terms of land-use changes and agricultural intensification. Our results suggest that the implications of conserving annual volumes of water for EFP on the land-use system are moderate on an aggregate global level. Cropland expansion into unmanaged land arising from increased food demand up to 2045 is higher by a factor 5–9 than cropland expansion induced by EFP. Global forest losses associated with EFP remain below 1% of current forest area. Production reallocation and associated land-use change hotspots suggest that local effects are of more concern than aggregate cropland expansion and deforestation.

1. Introduction

Freshwater is indispensable for sustaining life on Earth and adequate freshwater availability is essential for human well-being (Oki, 2006). Direct human water abstraction includes water extracted for irrigated crop production, energy production, industrial fabrication, and domestic purposes (Flörke et al., 2013). Additionally, humanity benefits from a variety of services provided by freshwater ecosystems. These include the provision of fish, other food and fiber, water purification, erosion protection, biodiversity conservation and cultural and recreational benefits (Poff et al., 2010; UNESCO, 2009). The third World Water Development Report by UNESCO (2009) highlights that the economic value of freshwater ecosystem services exceeds $600 million in Uganda alone and that 10% of undernourished people depend on freshwater ecosystem services across developing countries.

At the same time, it is widely recognized that human disturbance leads to degradation of freshwater ecosystems (Grafton et al., 2012; Hoff, 2009; Poff et al., 2010). It has been estimated that freshwater vertebrate populations have declined by 54% globally and that 32% of the world's amphibian species are threatened with extinction (Dudgeon et al., 2006). Increasing human water demand due to population growth and economic development is likely to put additional pressure on freshwater resources in the future (Vorosmarty et al., 2010). This has led to the call for water management strategies that explicitly recognize environmental water requirements (Falkenmark and Molden, 2008). Sustaining river ecosystems and associated livelihoods

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http://dx.doi.org/10.1016/j.gloenvcha.2014.10.015
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requires the provision of environmental flows (EFs) that ensure sufficient quantity, quality and timing of stream-flow (Poff et al., 2010). However, focusing on a single conservation goal can lead to negative feedbacks on other ecosystems (Bennett et al., 2009; Seppelt et al., 2013). Therefore, environmental flow protection (EFP) cannot be pursued in isolation but has to be put into a broader context of sustainable use of natural resources (Gerten et al., 2013).

Freshwater use for irrigation accounts for around 70% of human water withdrawals (Rost et al., 2008) and plays an important role in global food production (Naylor, 1996). Since the allocation of water resources for environmental purposes reduces available water for other uses (Falkenmark and Molden, 2008), EFP is likely to reduce water available for irrigation (Rosegrant et al., 2009; Strzepek and Boehlert, 2010). A reduction of irrigation potential may have adverse effects on food prices (De Fraiture and Wichelns, 2010) and the agricultural production required to provide sufficient food for a growing world population (Falkenmark et al., 2009; Rosegrant et al., 2009). Moreover, decreased land productivity due to irrigation water shortages may trigger cropland expansion (Lambin and Meyfroidt, 2011) at the expense of natural forests and other unmanaged land (Foley et al., 2011). Such land cover is important for a variety of ecosystem services including biodiversity conservation (Barlow et al., 2007) and carbon sequestration (Onaindia et al., 2013).

It is therefore important to assess the feedback from EFP on land-use change in order to provide the knowledge needed to design sustainable water management strategies (Pahl-Wostl et al., 2013c). While the design and implementation of EFP policies need to be tailored to local conditions (Arthington et al., 2006), it is also necessary to adopt a global point of view in order to account for non-local drivers of water and land-use such as agricultural trade, population growth and economic development (Hoekstra, 2010; Hoff, 2009). Global investigations of the effects of EFP on agriculture to date are however lacking. Strzepek and Boehlert (2010) have studied the effects of providing annual volumes of water for EFP on current agricultural water availability. However, they do not investigate, which adaptation mechanisms in the agricultural system are to be expected and what the consequences would be for global land-use change.

This article aims to provide an initial quantitative estimate of the effect on global land-use change of a reduction in water available for agriculture which would result from the allocation of water for EFP. We compare scenarios including the reservation of the annual water quantity needed for EFP to a business as usual scenario without EFP. Global land-use patterns, yield increases and agricultural water withdrawals under biophysical and socioeconomic constraints are simulated by the spatially explicit global agricultural land and water-use model MAgPIE (Model of Agricultural Production and its Impacts on the Environment) (Lotze-Campen et al., 2008; Popp et al., 2010). This setup allows us to quantify the effect of conserving annual volumes of water for EFP on agricultural water withdrawals as well as associated reactions in terms of land-use change and agricultural intensification under different future socioeconomic conditions.

2. Methods

2.1. Model description MAgPIE

2.1.1. General model description

MAgPIE is a spatially explicit global land and water-use allocation model that simulates in 10-year time steps from 1995 to 2045 using recursive dynamic optimization (Lotze-Campen et al., 2008; Popp et al., 2010). The objective function of the model is to fulfill demand for food, livestock and materials at minimum costs under biophysical and socio-economic constraints. Demand and socio-economic constraints are defined at the regional level (10 world regions, Fig. A1). Biophysical constraints enter the model at grid cell level ($0.5^\circ \times 0.5^\circ$ longitude/latitude; 59,199 grid cells). Due to computational constraints, grid-cells are aggregated to 1000 simulation units globally (Fig. A2) based on similarity of biophysical conditions (Dietrich et al., 2013). Demand scenarios are based on exogenous population, income and diet projections (Section 2.2.2). The production function takes land, water, yield information and monetary costs as inputs. MAgPIE endogenously determines cropland (rainfed and irrigated), forest and other natural vegetation patterns. Urban areas and pasture areas are assumed to be static over time and forest land is partly reserved for forestry activities and nature conservation (Section 2.2.2). Each land pool is initialized with historical patterns in 1995 (Krause et al., 2013). Crop yields (rainfed and irrigated) for 17 cropping activities, crop water demand, and water availability are provided by the global vegetation and hydrology model LPJmL (Bondeau et al., 2007; Müller and Robertson, 2014). Yields are calibrated at a regional level to meet historical FAO cropland (FAOSTAT, 2013) in 1995.

Technological improvements have helped to increase yields in the past (Fischer and Edmeades, 2010). In MAgPIE, future yield increases from technological change (TC) are modeled endogenously. Land-use intensities (τ) for each region in 1995 are derived from historical data (Dietrich et al., 2012). Yields in MAgPIE scale linearly with τ, and the model can invest in TC in order to increase τ (Dietrich et al., 2014). The investment-yield ratio (IY; investments per unit of yield growth) is determined from historical data on total agricultural research and development spending (Pardey et al., 2006), agricultural infrastructure investments (transport, energy and water distribution, telecommunication and financial services; Narayanan and Walmsley, 2008) and yield data (FAOSTAT, 2013). A regression analysis is used to estimate the elasticity of IY with respect to τ (Eq. (1)).

$$\mathrm{IY}(\tau) = (1900 \pm 400) * \tau^{2.4 \pm 0.9} \quad (1)$$

Thus, yield improvements due to TC are more costly in intensive systems with high τ values where low-cost intensification options are already implemented. Factor requirement costs (capital, labor and chemicals, e.g. fertilizer) per hectare scale linearly with τ (Dietrich et al., 2014).

Agricultural production costs comprise factor requirement costs, land conversion costs, transportation costs to the closest market, investment costs for irrigation infrastructure and investment costs for technological change. The cost minimization problem is solved through endogenous variation of rainfed and irrigated production patterns for 17 cropping activities (subject to regional trade constraints; Schmitz et al., 2012), land conversion (all at simulation unit level) and technological change (at regional level) (Lotze-Campen et al., 2010).

2.1.2. Water and irrigation representation

Water-use categories in MAgPIE include irrigation and livestock production, non-agricultural human water demand (domestic use, industrial use and use for electricity production) and environmental water requirements. Non-agricultural human water demand and environmental water requirements enter the model as exogenous scenarios (Section 2.2.2). Non-agricultural water demand is always fulfilled before agricultural water demand and thus effectively limits water availability for agriculture (similar to Elliott et al. (2013)). Livestock water demand per unit produced is derived from FAO (2005). Rainfed crop production is based on green water only (precipitation infiltrated into the soil; Rost et al. (2008)). The model can endogenously decide to apply

additional irrigation water from blue water resources (rivers, lakes, aquifers) in order to increase yields. The amount of irrigation water per hectare that has to be applied to a field is simulated by LPJmL as the soil water deficit below optimal plant growth (Rost et al., 2008) and corrected for 36% losses from source to field based on Rohwer et al. (2007). Thus, MAgPIE endogenously determines the extent of irrigated areas but does not accommodate the switching to supplementary irrigation if water becomes scarce. Yield increases, through technological change, are assumed to leave crop water demand per hectare unchanged, thus enhancing water productivity per ton (irrigation efficiency). This is in line with findings that yield increases and improved agronomic practices are essential for improving water productivity (Kijne et al., 2004; Molden et al., 2010; Rosegrant et al., 2009). Increased water productivity can be accomplished by, for example, minimizing losses in the water distribution system, increasing the ratio of transpiration to evaporation on the field, increasing the ratio of harvested plant biomass to total biomass production, and increasing plant water-use efficiency by breeding and improved management of all inputs.

Water-use in MAgPIE is constrained by available blue water at simulation unit level. Available water is calculated on a 0.5 arc-degree grid using monthly hydrological inputs from LPJmL and subsequently aggregated to simulation unit level. For each river basin, total annual runoff (precipitation that enters rivers, lakes and aquifers) in the basin constitutes the amount of water available in one year. The allocation of available water at basin level to the grid cells is based on the grid cells' monthly river discharge, i.e. the runoff accumulated along the river network. The fraction of total basin water available for each grid cell is calculated as the ratio of the grid cell's monthly discharge to the sum of the monthly discharge for all cells in the basin (Appendix, Eq. (A4)). This procedure ensures that water usage does not exceed available water at basin level and has also been used by Schewe et al. (2013).

It has however been highlighted that due to seasonal distribution, not all blue water is accessible to humans (Postel et al., 1996). Quantifying this effect is difficult (Gerten et al., 2013) and annual water availability is often estimated in a rule of thumb manner by assuming a constant global availability fraction (Elliott et al., 2013) that differs across studies (Gerten et al., 2013). In this study, we use a process based estimate of the reduction of annual available water for irrigation due to seasonal variation. We assume that irrigation water can only be used by plants during their growing period. Therefore, we calculate the mean growing period over all crops for each grid cell and restrict water available for irrigation to the water available in this period. The following assumptions have been made for the mean growing period calculation:

- winter crops in the northern hemisphere (sowing date later than June 30th and harvest date later than December 31st) are excluded assuming that no irrigation takes place during winter time in this region;
- crops with yields below 10% of the world average yield for each grid cell are excluded. Such low yields indicate that the crop is not suitable for the location.

In grid cells where water storage facilities are provided by dams according to Biemans et al. (2011), total annual blue water resources are available for irrigation, resulting in a global water availability of 27,000 km^3 per year.

2.1.3. Irrigation costs

Irrigated crop production is not only constrained by water availability but also requires irrigation infrastructure for water distribution and application. The initial pattern of area equipped for irrigation is taken from the AQUASTAT database (Siebert et al., 2007). MAgPIE can endogenously deploy additional irrigation infrastructure. Regional costs per hectare of expanding irrigation infrastructure in 1995 are derived from Worldbank data (Jones, 1995) and range from 1900 to 37,200 US$/ha (Appendix, Table A1). Regional heterogeneity is largely driven by implementation difficulties such as funding shortages, procurement problems and construction quality (Jones, 1995). This is reflected by the comparatively higher costs in Africa and Latin America compared to Europe or Pacific Asia. Irrigation infrastructure costs are furthermore influenced by labor costs and the choice of irrigation technology (e.g. surface vs sprinkler irrigation; Rohwer et al., 2007). We assume that world regions will converge in the future with regard to economic, institutional and technological standards. Investment costs for irrigation infrastructure therefore converge linearly toward the European level of 5700 US$/ha up to 2050.

In order to conduct our analysis, we improved the model by including annual costs for irrigation (e.g. for water, fuel, labor and the maintenance of irrigation infrastructure). Unfortunately, however, there is no available global dataset on irrigation costs. Calzadilla et al. (2011) have proposed an approach to extract the rent associated with the application of irrigation water (i.e. factor payments for irrigation water) from the GTAP land rent (i.e. payments for the factor land) (Narayanan and Walmsley, 2008). We largely follow this approach to determine the factor requirement costs for irrigation from the GTAP land rent. A detailed description of the calculations can be found in Appendix A.2. The resulting annual irrigation costs (Appendix, Table A2) range from 10 to 2000 US$/ha for different crops and regions. 90% of the calculated costs are between 10 and 404 US$/ha and only two values exceed 1000 US$/ha (potato in the NAM and PAO). Case studies from the FAO report annual irrigation costs of 9.5–400 US$/ha for different crops in African countries (Palanisami, 1997). Average annual irrigation costs for a selection of farms in the US range between 167 and 392 US$/ha (Schaible and Aillery, 2013). Thus, our approximate calculation generally yields results that are comparable to observations from case studies.

2.2. Scenarios

We simulate global agricultural land and water use from 1995 to 2045 considering a total of eight scenarios that differ along two dimensions: environmental flow protection (four alternative futures) and socioeconomic development (two alternative futures). In order to limit the number of scenarios and to isolate the effects of EFP policies, climate change is not considered. All biophysical inputs are average LPJmL results for 1991–1999 throughout the simulation period and LPJmL is driven by climate data from the CRU 3.0 dataset (Mitchell and Jones, 2005). We also test the sensitivity of our results with respect crucial model parameters.

2.2.1. Environmental flow protection scenarios

We compare scenarios where annual volumes of water are secured for environmental flow protection (EFP) from 2015 onwards, to a baseline scenario without water allocation for EFP throughout the simulation period. The baseline assumption is based on findings that EF violation is a widespread global phenomenon. Hoekstra et al. (2012) have found that in 223 of 405 large river basins, EFs are violated at least one month a year. Furthermore it has been highlighted that current real-world water management rarely considers environmental water requirements (Arthington et al., 2010; Falkenmark and Molden, 2008).

For single river basins, EFs can be estimated with high confidence by applying holistic methods that combine hydrological, hydraulic and habitat simulation approaches (Pastor et al., 2013; Smakhtin and Eriyagama, 2008). Those cannot, however, be

easily scaled up since they rely on detailed site-specific data that is lacking at the global scale (Pastor et al., 2013). Therefore, global assessments like ours have to rely on approximate hydrological EF estimates based on modeled hydrological data. Over 200 EF estimation methods have been recorded (Arthington et al., 2006) and it has been highlighted that especially approximate hydrological methods are prone to large uncertainties (Poff et al., 2010; Smakhtin and Eriyagama, 2008). In order to account for this uncertainty, we consider three EFP scenarios with differing spatial EF patterns. For each EFP scenario, environmental water requirements are calculated from monthly hydrological data simulated by LPJmL on a 0.5 arc-degree grid. Subsequent aggregation to the 1000 simulation units used in this analysis, and a summation of all months, results in annual volumes for EFP.

The first EFP scenario (Tennant) is based on an early study by Tennant (1976) who proposes the reservation of 30% of available water for the environment in order to secure habitat for fishes, invertebrates and other wildlife and protect riparian vegetation from water stress. This threshold has subsequently been supported by a number of further studies (Hanasaki et al., 2008).

The second EFP scenario (Smakhtin) relies on research by Smakhtin et al. (2004). They propose a combination of low-flow requirements (LFRs) and high-flow requirements (HFRs) to sustain river ecosystems in a "fair" condition. The conservation goal is to limit species loss to very sensitive species and to limit intrusion by alien species. LFRs correspond to the 90% quantile of annual flow (Q90), i.e. to the discharge that is exceeded in nine out of ten months. Variable rivers are characterized by low Q90 values. In such cases, high-flow events are important for river channel maintenance, wetland flooding, and riparian vegetation. HFRs of 20% of available water are therefore assigned to rivers with a low fraction of Q90 in total discharge. Rivers with a more stable flow regime receive a lower HFR. For calculation details see Appendix A.3.

The third EFP scenario (Hanasaki) is constructed according to a study by Hanasaki et al. (2008). The method is based on case studies in semi-arid to arid regions. It considers deviations from natural hydrological conditions and estimates base flow and perturbation requirements. Technically, the method classifies rivers into four categories based on monthly discharge (Table 1). EFs between 0% and 40% of available water are then assigned based on monthly discharge values and category specific thresholds (Table 1; Appendix A.3).

The three EF calculation methods result in significantly different EF patterns (Fig. 1). The Tennant method is globally uniform. The Smakhtin method shows considerable spatial variation, estimating rather low EFs (20–30% of available water) in most regions of the world and high values of up to 50% in North America, Europe, the Amazon region, Central Africa and Indonesia. The Hanasaki method shows lower EFs in North America, Europe and North Africa and higher values throughout the tropics and large parts of Asia. Despite spatial variations, the global average fraction of EFs over available water is similar for all three methods (Tennant: 30%, Smakhtin: 30%, Hanasaki: 29%).

2.2.2. Socioeconomic scenarios

Since socioeconomic processes such as population growth, economic development and dietary preferences are key drivers of future land-use change (Popp et al., 2010), we conduct our analysis for two different socioeconomic scenarios. In order to span the plausible range of future conditions, we use two extreme cases from the SRES scenario family (Intergovernmental Panel on Climate Change and Working Group III, 2000). The A2 scenario is a high pressure scenario characterized by high population growth in developing regions (10.8 billion globally in 2045), self-reliance and regionally oriented economic development. In contrast, the B1 scenario depicts a world where global population growth slows down over the next decades reaching 8.6 billion in 2045. The emphasis of B1 is on global solutions to economic, social, and environmental problems. Within MAgPIE, these socioeconomic storylines are translated into scenarios for non-agricultural water demand (on a simulation unit level), demand for agricultural products (at a regional level), trade liberalization, and forest protection, as described below.

Spatially explicit non-agricultural human water demand for the domestic and industrial sectors is obtained from the WaterGAP model (Alcamo et al., 2003; Flörke et al., 2013). Non-agricultural water demand is subject to structural and technological change and scales with population and economic activity according to the SRES storylines. Under the A2 scenario, global non-agricultural human water withdrawals more than triple up to 2045 while they are almost constant under the B1 scenario (Fig. 2).

Global demand for agricultural products depends on population and food consumption per capita. Changes in regional food composition (e.g. livestock share) and per capita demand are driven by changes in per capita gross domestic product (Bodirsky et al., 2012). Food demand increases almost linearly from 2168 Mt dry matter in 1995 to 4745 Mt in 2045 under the A2 scenario (Appendix, Fig. A2). Under the B1 scenario, the increase is less pronounced and flattens out toward the middle of the century reaching 3892 Mt in 2045.

Agricultural trade between world regions in MAgPIE reproduces historical trade patterns in 1995 and liberalizes over time (Schmitz et al., 2012). The A2 scenario features little trade liberalization; in 2045, only 12% of total production can be allocated freely between the world regions based on competitive advantages. The remainder has to be produced regionally based on historical self-sufficiency rates and export shares. The B1 scenario allows for more interregional trade with 35% free trade in 2045.

Table 1
Environmental flow requirements for the Hanasaki scenario. Table adopted from Hanasaki et al. (2008) The discharge unit (mm/month) corresponds to monthly discharge divided by the catchment area (see Appendix A.3 for details).

River classification			Monthly environmental flow requirement	
Description	Minimum monthly streamflow q_{min} (mm/month)	Maximum monthly streamflow q_{max} (mm/month)	Condition for monthly discharge q (mm/month)	Monthly flow requirement q_{env} (mm/month)
Dry (dry throughout a year)	$q_{min} < 1$	$q_{max} < 10$	$0 \le q < 1$ $1 \le q$	$q_{env} = 0$ $q_{env} = 0.1q$
Wet (wet throughout a year)	$10 \le q_{min}$	$100 \le q_{max}$		$q_{env} = 0.4q$
Stable (stable throughout a year)	$1 \le q_{min}$	$q_{max} < 100$		$q_{env} = 0.1q$
Variable (dramatic difference between rainy and dry season)	Other than above		$0 \le q < 1$ $1 \le q < 10$ $10 \le q$	$q_{env} = 0$ $q_{env} = 0.1q$ $q_{env} = 0.4q$

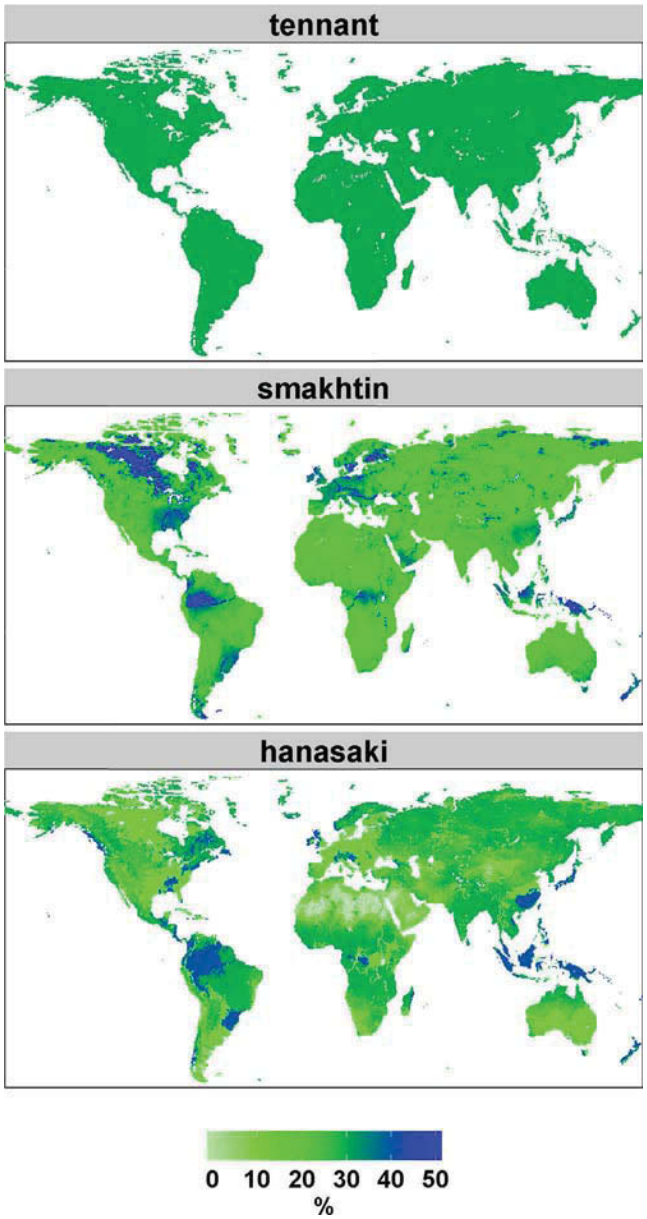

Fig. 1. Environmental flow requirements over available water in percent for the three EFP scenarios. Tennant method (top), Smakhtin method (middle) and Hanasaki method (bottom).

Forest protection covers areas needed for wood production and natural conservation. Initial protection areas are derived from the FAO forest resource assessment (FAO, 2010) at a regional level and amount to ~1800 Mha globally. In A2, this area stays constant over time while in B1, forest protection areas increase linearly reaching ~2450 Mha globally in 2045.

2.2.3. Sensitivity analysis

In order to test the stability of our results with respect to economic assumptions, we conduct a sensitivity analysis with respect to three cost parameters that are crucial for irrigation patterns and land-use dynamics.

Technological change costs: We vary the parameters of the cost function for technological change (Eq. (1)) from their default values, to the low and the high end of their uncertainty range.

Annual irrigation costs: We vary the annual costs for operation and maintenance of irrigation systems by ±30% around their default value.

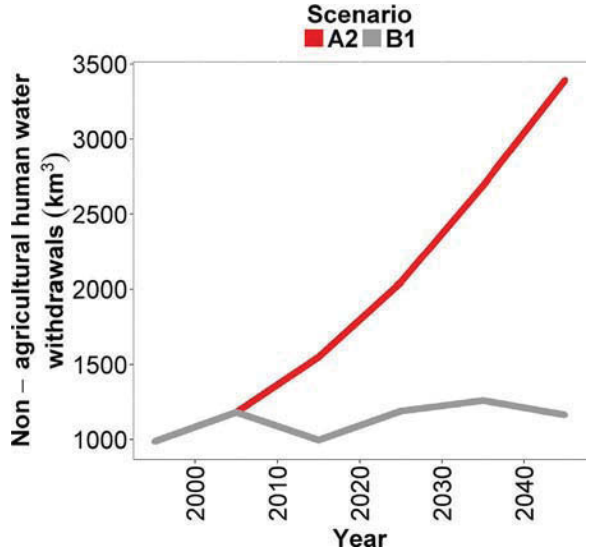

Fig. 2. Global non-agricultural human water withdrawals including industrial production, domestic use and electricity production for the two socioeconomic scenarios.
Data from WaterGAP (Alcamo et al., 2003; Flörke et al., 2013).

Investment costs of irrigation infrastructure: In addition to the default setting of regionally converging costs, we consider a scenario with globally uniform investment costs (at the European level of 5700 US$/ha) and a scenario without convergence of regional costs to the European level.

3. Results

3.1. Development under business as usual scenarios

Before investigating the effects of environmental flow protection (EFP), this subsection highlights the water situation under the business as usual scenarios without EFP. We concentrate on water scarcity projections and environmental flow (EF) violations. More detailed results regarding the general land- and water-use dynamics in the baseline scenarios can be found in Appendix B. These include comparisons with historical data for validation purposes.

3.1.1. Human water-use

In order to investigate human pressure on global renewable freshwater resources, we calculate the annual water withdrawal to availability ratio (WTA). For each socioeconomic scenario, total annual human water withdrawals in 2045 are divided by available water for the 1000 spatial simulation units (Fig. 3). Many regions have a high WTA under both scenarios, especially in the USA, Mexico, the Middle East, parts of Africa, Southern Europe, India and Northern China. Under the A2 scenario, the WTA is higher than under B1, especially in the USA, Eastern Europe and the Middle East.

3.1.2. Environmental flow violations

Where WTA values are high, annual human water withdrawals in the baseline scenario may tap into the annual volume of water required for EFP. We determine where the sum of baseline human water withdrawals in 2045 together with estimated annual EF volumes (Section 2.2.1) exceeds available water for the three EF estimates considered in this analysis (Fig. 4).

Around 8% of global land surface (excluding Antarctica and Greenland) exhibits EF violation for both socioeconomic scenarios

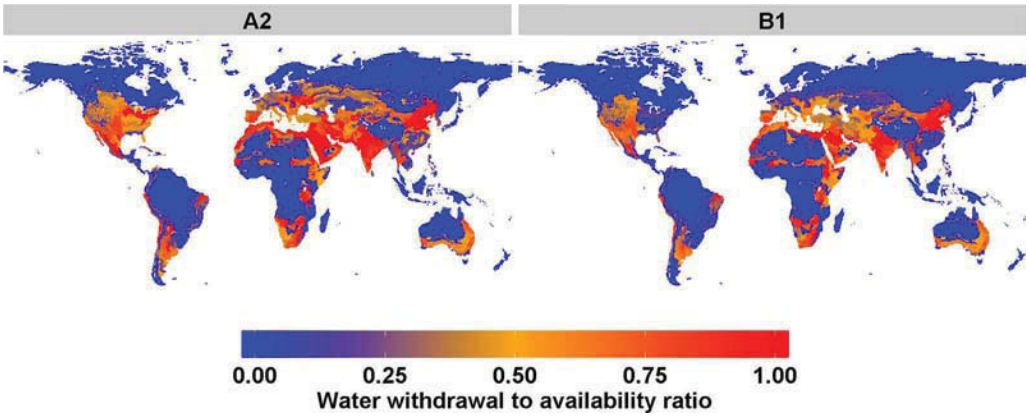

Fig. 3. Human water withdrawal to water availability ratio under both socioeconomic scenarios without EFP in the year 2045. The index is zero if no water is used by humans and one if all available water is extracted.

and all three EF estimates. These areas are responsible for ~10% of global agricultural production. The largest regions in this category are India, Northern China and the Middle East. Areas where at least one EF estimate under at least one socioeconomic scenario suggests EF violation cover 19% of the global land surface and account for ~30% of global agricultural production. In A2, EF violation under all three EF estimates occurs on 12% of global land area while in B1 this is only the case for 8% of land area. Regions where both socioeconomic scenarios exhibit EF violation under one or two EF estimates cover around 4% of global land area.

3.2. Effects of environmental flow protection

This section concentrates on the differences between the EFP scenarios and the baseline scenarios.

3.2.1. Irrigated agriculture

All three EFP scenarios substantially reduce agricultural water withdrawals (AWW) under both socioeconomic scenarios (Fig. 5, left). With increasing food demand and non-agricultural human water demand toward the middle of the century, the impact becomes more pronounced. By the middle of the century, AWW are reduced by 580–770 km^3 in the B1 scenario and 560–820 km^3 in the A2 scenario. This represents 20–27% and 20–29% of the simulated AWW for 1995 (2840 km^3), respectively. Regional results show that under the A2 scenario, reductions of agricultural water withdrawals due to EFP are most pronounced in the Middle East and North Africa (Appendix, Fig. C1).

These changes in AWW are driven by a reduction of irrigated areas (Fig. 5, right). The pattern of changes across the scenarios is similar to AWW changes. Reductions of irrigated area in 2045 due to EFP amount to 48–61 Mha in B1 and 54–82 Mha in A2. This corresponds to 15–20% (B1) and 17–26% (A2) of modeled irrigated area in 1995 (312 Mha).

3.2.2. Intensification versus cropland expansion into unmanaged land

Irrigated production accounts for around 40% of global agricultural production in our baseline scenarios. Since we assume that there is no demand-side reaction to the decrease in irrigated production capacity, the production losses due to EFP have to be compensated for by the supply side. The two main mechanisms for increasing agricultural production, apart from increasing irrigation water inputs, are cropland expansion and intensification, i.e. yield increases by better management and technological innovation (Lambin and Meyfroidt, 2011).

The reaction to EFP in terms of cropland changes shows strong variations, and heavily depends on the socioeconomic as well as on the EFP scenario (Fig. 6, left).

In the B1 scenario, cropland with EFP is higher than baseline cropland in all time steps under all three EFP scenarios. The

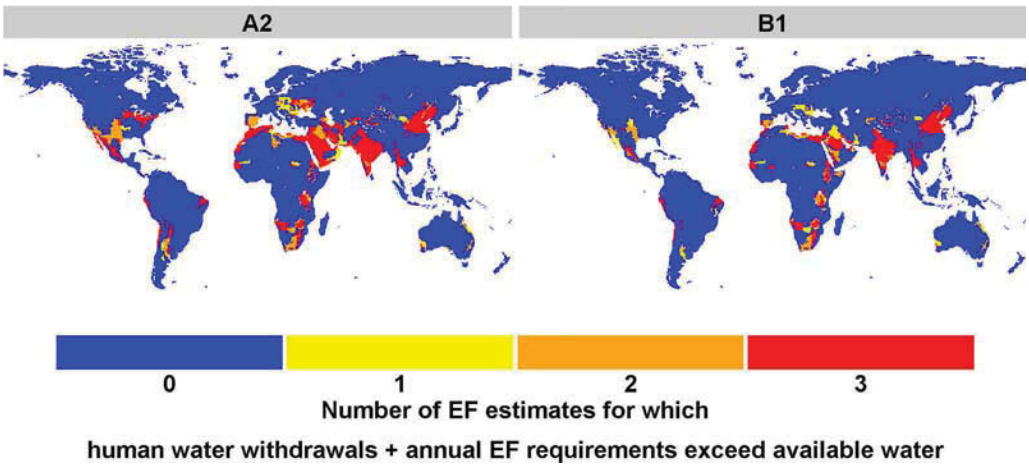

Fig. 4. Violation of annual environmental flow requirements for the two socioeconomic scenarios in 2045 under baseline conditions. Colors indicate the number of EF estimates for which human water withdrawals in the baseline + annual EF requirements exceed available water. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

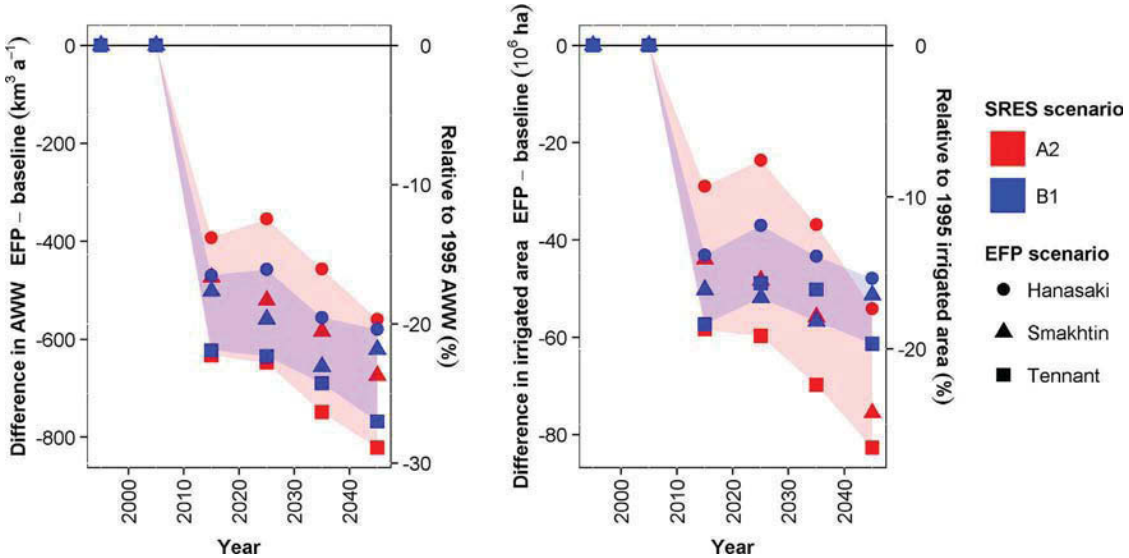

Fig. 5. Changes in global agricultural water withdrawals (AWW, left) and global irrigated area (right) in the EFP scenarios with respect to the baseline for both socioeconomic scenarios. Different point shapes represent the three EFP scenarios, while color distinguishes socioeconomic scenarios. Shaded areas span the full range of EFP impacts under the respective socioeconomic scenario. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

maximum cropland increase with respect to the baseline is observed right after the implementation of the EFP policy in 2015 and amounts to 73 Mha. In 2045, the three EFP scenarios require 34–39 Mha additional cropland compared to the baseline. This corresponds to around 2–3% of simulated cropland in 1995 (1420 Mha).

In the A2 scenario, the picture is more diverse with strong variations between EFP scenarios especially in 2025. Between 2015 and 2025, cropland expands by 150 Mha in the baseline scenario. In the Hanasaki scenario, cropland only expands by 50 Mha. The scarcity of land and water resources, as expressed by shadow prices for land (Appendix, Fig. C2) and water (Appendix, Fig. C3), is higher for the Hanasaki scenario compared to the baseline, especially in Latin America, North America, Pacific Asia, and South Asia. This indicates that the water constraints in the Hanasaki scenario limit the availability of attractive production sites. Consequently, the cost-optimal solution is less cropland expansion and stronger intensification for the Hanasaki scenario compared to the baseline without EFP (Fig. 6). In 2045, projected cropland expansion due to EFP amounts to 2–45 Mha (0–3% of simulated 1995 cropland). Cropland expansion especially in the Middle East and North Africa is restricted by land availability (Appendix, Fig. C4).

Agricultural intensification, i.e. an increase of agricultural yields by technological change and better management as a reaction to EFP, can be observed in all scenarios (Fig. 6, right). Under the A2 scenario, global yield increases from technological change of around 4–6% with respect to the baseline, are required in 2045. Under the B1 scenario, required yield increases are lower, peaking at 4% in 2035 and reaching around 2–3% in 2045. Land-use intensities under baseline conditions are shown in the Appendix (Fig. B4). Furthermore, EFP leads to the reallocation of production

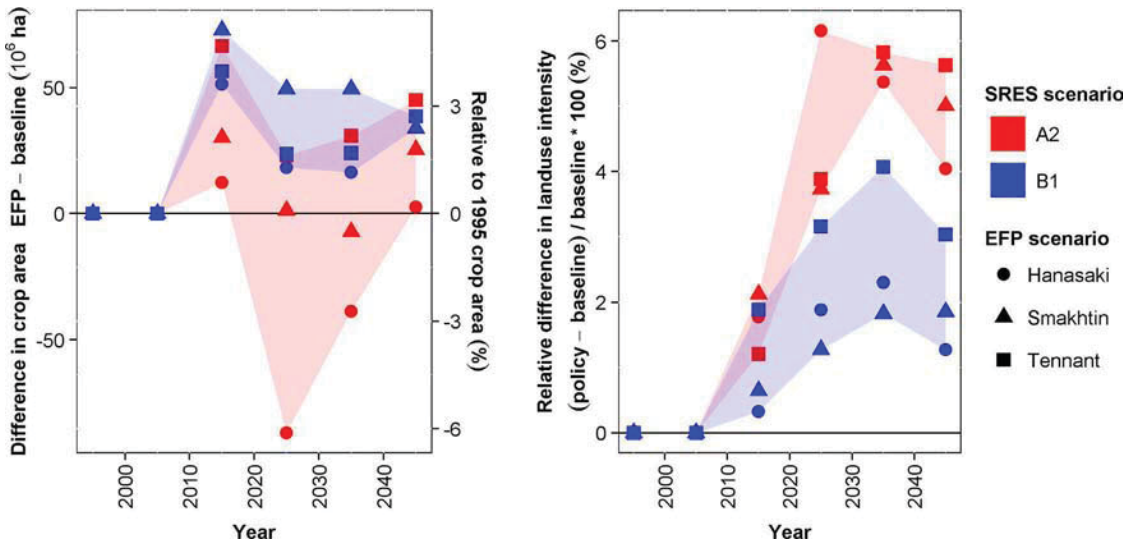

Fig. 6. Changes in global crop area (left) and relative changes in land-use intensity (right) in the EFP scenarios with respect to the baseline for both socioeconomic scenarios. Different point shapes represent the three EFP scenarios, while color distinguishes socioeconomic scenarios. Shaded areas span the full range of EFP impacts under the respective socioeconomic scenario. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

between regions, especially in the B1 scenario with lower trade barriers (Appendix, Fig. C5).

Within MAgPIE, cropland expansion due to EFP can take place at the cost of unmanaged forests and other natural vegetation. In order to isolate the effects of EFP on natural forests, we determine the change in global forest area due to EFP (Fig. 7). The general pattern of changes in forest area due to EFP is similar to the changes in crop area (Fig. 6, left) because cropland expansion is the driver of deforestation.

Under the B1 scenario, losses of natural forests occur throughout the simulation period and under all EFP scenarios. The highest spread between different EFP scenarios can be observed in 2035 with deforestation of 9–39 Mha. In 2045, the model results for different EFP scenarios converge to ~37 Mha of lost forest. For the A2 scenario, differences between EFP scenarios are most pronounced in 2025 with changes in forest area of −17 to 51 Mha. Positive values (more forest area with EFP) are a result of strong baseline deforestation between 2015 and 2025 (100 Mha). In the Hanasaki scenario, only 40 Mha of forest are lost during that period. By the middle of the century, changes in forest area are 7 to −37 Mha. In the Hanasaki scenario, EFP even leads to less deforestation than under baseline conditions.

In order to investigate where there is high pressure on land ecosystems due to EFP, we determine where cropland area changes due to EFP under the three different EFP scenarios. A spatially explicit index ranging from 0 to 3 is derived, indicating how many EFP scenarios lead to cropland expansion into unmanaged land (Fig. 8, top) and to cropland contraction (Fig. 8, bottom).

Under the A2 scenario, differences between EFP scenarios are high, indicated by the large areas where only one or two, out of three EFP scenarios, lead to cropland changes (Indonesia, USA, Africa and Brazil). Hotspots, where cropland expansion is likely to occur, are located in Argentina, Chile, the United States, Canada, Western and Central Africa, Northern China, Japan, Turkey, India and Afghanistan. Under the B1 scenario, land-use change hotspots are located in the USA, Western and Central Africa, the Middle East and South Asia.

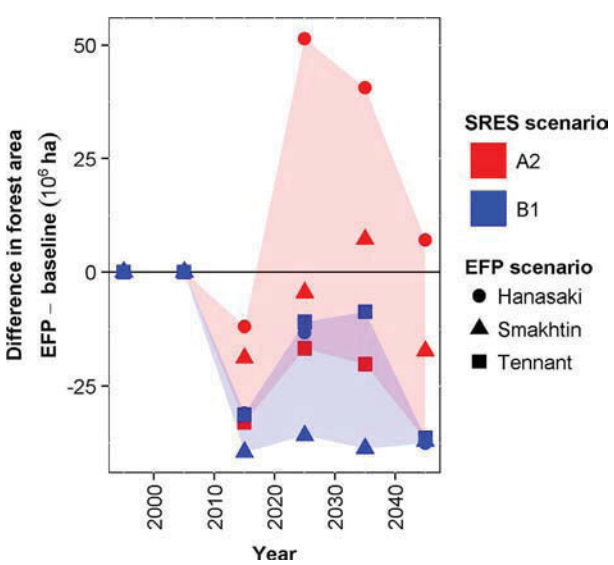

Fig. 7. Changes in global forest area with EFP compared to the baseline for both socioeconomic scenarios. Different point shapes represent the three EFP scenarios, while color distinguishes socioeconomic scenarios. Shaded areas span the full range of EFP impacts under the respective socioeconomic scenario. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

3.2.3. Sensitivity analysis

Under the B1 scenario, the implications of EFP for global AWW and crop area in 2045 are generally robust against variations of technological change (TC) costs (Appendix, Fig. D1), annual costs for operation and maintenance of irrigation systems (Appendix, Fig. D2), and investment costs for irrigation infrastructure (Appendix, Fig. D3). Under the A2 scenario, high TC costs can lead to more cropland expansion due to EFP (up to 80 Mha in 2045) while low TC costs limit crop area expansion due to EFP in 2045 to 30 Mha. High annual irrigation costs increase cropland expansion due to EFP in A2 (up to 107 Mha in 2045) and decrease the implications of EFP for crop area in B1 (below 30 Mha in 2045). Changes in irrigation infrastructure investment costs can lead to reduced crop area requirements with EFP when compared to the baseline (up to 90 Mha in 2045).

4. Discussion

This study investigates how the conservation of annual volumes of water for different EFP regimes may affect global agricultural water-use and land-use dynamics under two different socioeconomic scenarios. Our modeling approach allows us to simulate spatially explicit global reactions of the land-use system to EFP without exogenously prescribing production patterns, irrigation patterns, trade flows or yield increases. This endogenous treatment of many key agricultural variables in a cost optimization framework is important in order to capture the linkages and cross dependencies in the land and water-use sector. Projected global AWW, irrigated area, cropland and land-use intensity under baseline conditions are consistent with historical data around the year 2000 (Appendix B).

4.1. Development under business as usual scenarios

Human pressure on water resources is projected to be strong under both socioeconomic scenarios, indicated by high WTA values. Under the pessimistic A2 scenario featuring high population growth and little global cooperation, the fraction of global land area (excluding Antarctica and Greenland) with a WTA > 0.4 is 28%. This is in line with results from Alcamo et al. (2007) who estimate that in the 2050s, 26–28% of total land area show a WTA higher than 0.4 for A2 depending on future climatic conditions. Our spatial patterns of WTA are similar to this in Latin America, Africa, the Middle East and Europe, but we estimate higher WTAs in the east of the USA, Australia and China. One methodological difference is that Alcamo et al. assume constant irrigated area while we allow for expansion of irrigated agriculture. Therefore, agricultural water withdrawals are significantly higher in our analysis (3815 km^3 in 2045 for A2) compared to their projection of 2282 km^3 in 2055, leading to higher WTA values. These higher WTA values are mostly concentrated in areas where Alcamo et al. (2007) also estimate WTAs >0.4 except for the differences mentioned above. An analysis by Shen et al. (2008) projects agricultural water withdrawals of 4691 km^3 in 2055 under the A2 scenario which is higher than our estimate because the study assumes constant irrigation per capita and neglects water availability constraints. In the B1 scenario, with low population growth and low trade restrictions, 22% of the total land area has a WTA higher than 0.4. These are lower than those found in the A2 scenario, mainly due to lower non-agricultural human water withdrawals in B1, since AWW are similar for the two scenarios at the global level. Whether human water withdrawals are expected to tap into the water volumes needed for EFP does not only depend on the amount of water withdrawn but also on the underlying EF estimate. We estimate that EF volume violations are equally dependent on socioeconomic drivers and EF estimates.

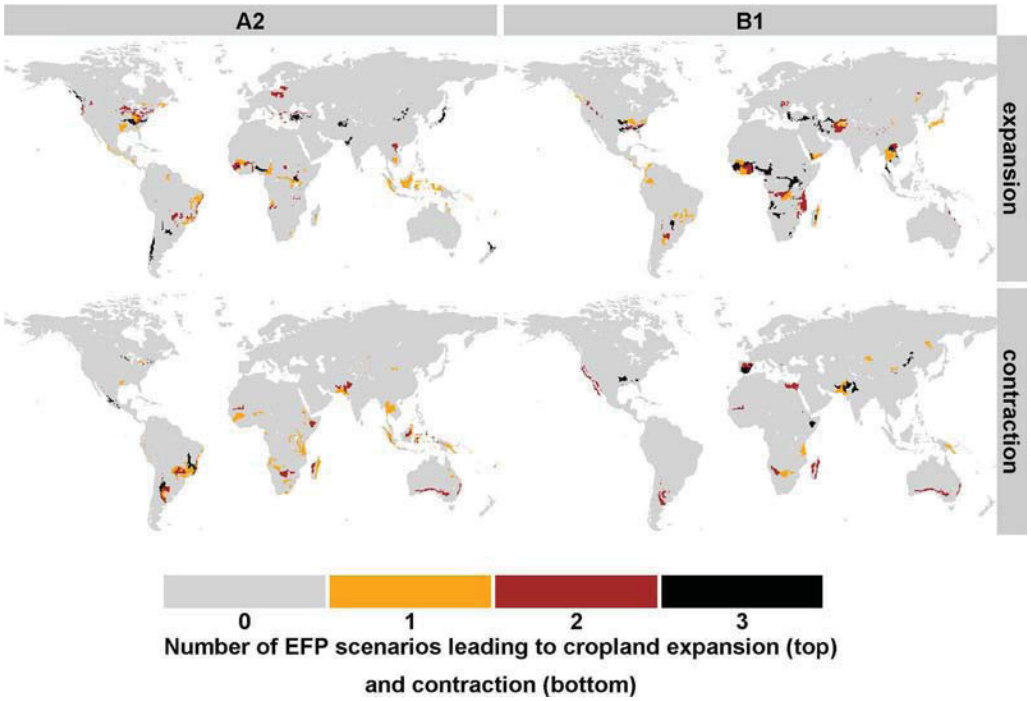

Fig. 8. Locations where conserving annual volumes of water for EFP leads to cropland expansion into previously unmanaged land (top) and cropland contraction (bottom) for the two socioeconomic scenarios in 2045. Colors indicate the number of EFP scenarios that lead to cropland expansion/contraction at a given location. If changes in cropland extent stay below 1% of total land area, the location is marked as no change (0). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

4.2. Effects of environmental flow protection

4.2.1. Irrigated agriculture

Our results suggest that conserving annual volumes of water for EFP may substantially reduce the potential for irrigated agriculture to contribute to global food production by reducing AWW and irrigated area. This is in line with findings that environmental water requirements are a bigger threat to AWW than non-agricultural human water-use and climate change effects (Strzepek and Boehlert, 2010). In our results, reductions of AWW and irrigated area are sensitive to the EFP scenario but are similar for the socioeconomic scenarios. The spread between EFP scenarios is lower in B1, where more liberalized agricultural trade compensates for spatial differences in EF patterns between EFP scenarios.

4.2.2. Intensification versus cropland expansion

In our scenarios, the agricultural system reacts to EFP through intensification and cropland expansion. Reductions of agricultural water withdrawals due to EFP are most pronounced in the Middle East and North Africa, a region where no suitable land for cropland expansion is available. Latin America on the other hand, being the region with the largest available land pool, is not strongly affected by EFP. As a consequence, intensification is the main reaction to EFP under the A2 scenario which features high population growth and strong trade restrictions. Under the B1 scenario, more liberalized trade encourages reallocation of production to regions with abundant land resources (Africa and North America). This means that intensification due to EFP is less pronounced and there is a clearer trend toward cropland expansion for all EFP scenarios. Moreover, production reallocation under the B1 scenario reduces the spread of cropland expansion between EFP scenarios compared to A2. A rough calculation reveals the reason for the moderate feedback of the conservation of annual water volumes for EFP on crop area: irrigated agriculture accounts for around 40% of total production in our scenarios, which compares to the 30–40% reported by Rost et al. (2008). Reductions in irrigated area of ~20% therefore lead to global production losses of ~8% assuming homogenous yields. A major part of these production losses is then compensated for by production reallocation and yield increases on current cropland.

4.2.3. Implications for forests and other unmanaged land

In our scenarios, cropland expansion due to allocation of annual water volumes for EFP can reduce global unmanaged natural land by up to 45 Mha by 2045. Around 80% of the loss of unmanaged land can be attributed to natural forests that mostly belong – according to our definition – to large intact forest landscapes (Krause et al., 2013) and represent valuable undisturbed ecosystems. The decline of land ecosystems could lead to biodiversity loss, carbon emissions, and could adversely affect water regulation (Onaindia et al., 2013).

How do the benefits associated with conserving annual volumes of water for EFP (including the provision of food and fiber, water purification, biodiversity conservation, and erosion protection; Poff et al., 2010) compare to the detriments of losing 45 Mha of natural land ecosystems? Answering this question would require a detailed analysis of the quality and quantity of the affected ecosystem services, would inevitably involve value judgments (Ford et al., 2009), and is therefore beyond the scope of this analysis. We can however analyze the additional challenges for sustainable land management strategies due to annual water allocation for EFP. To this end, we set the consequences of our EFP scenarios into the context of the general pressure on land ecosystems due to population growth and economic development under baseline conditions.

In our scenarios, global cropland expansion due to EFP by 2045 stays below 3% of the 1995 value. Expansion due to population and income growth under baseline conditions up to 2045 is higher by a factor 5–9 (229 Mha in B1 and 413 Mha in A2). A recent agricultural model intercomparison used an ensemble of

models to project cropland development under an SSP2 (IIASA, 2013) middle of the road scenario (Schmitz et al., 2014). Projected cropland expansion until 2050 for the ensemble mean, without climate change, is around 200 Mha, a factor of 4 higher than our estimated impact of securing annual volumes of water for EFP. In our scenarios, long term deforestation associated with EFP corresponds to approximately 1% of current global forest area (FAOSTAT, 2013). This is twice the annual amount of lost forest of around 19 Mha during the period 2000–2012 (Hansen et al., 2013). The Hanasaki EFP scenario even leads to reduced deforestation compared to the baseline in A2.

Even though there is no major feedback from securing annual volumes of water for EFP on land-use change at an aggregate global level, local land-use change can impair the livelihood and wellbeing of people. Indigenous people and the rural poor are especially dependent on natural and wild resources (Naughton-Treves et al., 2005) that are threatened by land-use change. Environmental damages of local land-use change include biodiversity loss, soil erosion and changes in the water cycle (Klink and Machado, 2005). Due to the rather coarse resolution of 1000 spatial units, our modeling approach is not suited to investigate detailed local effects. However, important insights can be gained from the patterns of cropland expansion hotspots arising from EFP. Cropland expansion happens on 10% of global land area under at least one EFP scenario and at least one socioeconomic scenario. Less than 1% of global land area is affected by cropland expansion under all considered scenarios. This indicates that the pattern of land-use changes associated with EFP is highly dependent on the EFP scenario. Comparing the patterns of cropland expansion in the EFP scenarios to EF violation patterns under baseline conditions shows that there is little overlap between regions where EF violations occur and where EFP leads to land-use changes. For both socioeconomic scenarios, around 90% of the area affected by cropland expansion under at least one EFP scenario is located in regions where no EF violations are observed in the baseline.

4.2.4. Implications of different environmental flow volume targets

Our three EFP scenarios differ in their impact on land-use change and deforestation because the spatial distribution of the EF volume targets is different. The globally uniform Tennant scenario has the strongest impact on global irrigated agriculture, land-use change, agricultural intensification, and deforestation. This indicates that spatial variation of EFs in the other two scenarios can be exploited by reallocating production in order to tap new water resources. This is also reflected in the strong dependence of spatially explicit land-use change patterns on the EFP scenario. The Hanasaki scenario has the least impact on land-use change and forest area at the global scale. Thus, from a global land-management perspective, the Hanasaki EF volume estimate is the best choice of the three considered methods.

The question of which of the three methods is most suitable concerning freshwater ecosystem protection, is more complex. There is a general agreement that freshwater ecosystem protection requires environmental flows that mimic natural conditions including the magnitude, timing, and frequency of flow events (Arthington et al., 2006). In this analysis, we concentrate on annual volumes of water for EFP and do not consider the timing within a year (see Section 4.3). However, the considered methods of estimating annual EF volumes differ in the way they account for flow variability and river-specific characteristics: the Tennant method does not consider intra-annual flow distribution; the Smakhtin method takes into account intra-annual flow variation to estimate low and high flow components of EFs; the Hanasaki method uses the most detailed rules (4 river categories, EF volume targets depending on monthly hydrological conditions) to estimate annual EF volumes. This suggests that the annual EF volumes of the Hansaki method are most appropriate for freshwater ecosystem protection, followed by the Smakhtin estimate and the Tennant estimate. However, a recent review of five global EF estimates involving a comparison against more detailed holistic EF estimates for 11 local case studies (Pastor et al., 2013) found that the Smakhtin and the Tennant method are comparable ($R^2_{adj} = 0.86$ and 0.88 respectively). The Hanasaki method was not tested. Thus, different detail of allocation mechanisms can lead to comparable results and it is likely that all three methods provide good estimates for some rivers and fail for others.

In this analysis, we investigate the implications of global EFP regimes. While there is an increasing intervention of international law in water issues, in reality water is managed at a local or country scale (Gupta et al., 2013). Our results indicate that securing annual volumes of water for EFP can lead to comparative disadvantages for food production. Local management authorities may therefore decide to implement EFs of low rigor or may not consider EFP at all. Fragmented EFP can lead to leakage of water stress due to production reallocation, i.e. to aggravation of water stress in regions without EFP. A fragmented EFP regime would likely put less pressure on global land-use and agriculture than a global EFP regime because more water would still be available for irrigation. Local production patterns might however change considerably.

4.3. Assumptions and limitations

Within our EFP scenarios, annual volumes of water are reserved for sustaining freshwater ecosystems. In reality, the functioning of such ecosystems depends on a variety of additional factors including water quality, the timing of stream flow within a year (Poff et al., 2010) and riparian land-use (Arthington et al., 2010). Thus, our EFP scenarios depict a world where a necessary step toward freshwater ecosystem protection is taken (managing water quantities) but where ecosystems may still face degradation due to factors that are not considered here. A comprehensive EFP policy that includes flow-timing and water quality aspects would likely increase the impact of EFP on agriculture.

EF requirements are highly dependent on the characteristics of individual river ecosystems and a thorough EF estimation requires site-specific data and a combination of hydrological, hydraulic and habitat simulation approaches (Pastor et al., 2013; Smakhtin and Eriyagama, 2008) that are not available at the global scale. The approximate global EF estimation methods that are applied here are therefore prone to large uncertainties (Poff et al., 2010; Smakhtin and Eriyagama, 2008). The three EFP scenarios in this analysis rely on different EF estimation methods in order to account for this uncertainty.

EFP can have many consequences across the water–energy–food nexus including increasing prices for food and energy that influence equity and social stability (Ringler et al., 2013). Economic and social consequences of EFP depend on a variety of factors that are not considered in our analysis, e.g. market access, income distribution, and social welfare systems. Our study is limited to the environmental tradeoff between water and land and within our modeling framework, non-agricultural water demand does not react to EFP so that the water requirements for EFP have to be met at the expense of AWW. The justification for this assumption is the comparatively higher value of water in domestic and industrial sectors (Strzepek and Boehlert, 2010) and the high proportion of AWW in total human water-use.

Not all withdrawn water is consumed. The majority of water withdrawn returns to the river system and is available for downstream use after proper wastewater treatment, particularly in non-agricultural sectors (Flörke et al., 2013). Taking such return flows into account would require tracking the water along the river network during the optimization, which is not feasible in our

modeling framework due to computational limitations. This is a common issue of global water assessments (e.g. Alcamo et al., 2007; Chaturvedi et al., 2013; Sauer et al., 2010; Smakhtin et al., 2004; Strzepek and Boehlert, 2010) and it remains outstanding to include a full river routing routine into global optimization approaches. We therefore cannot consider upstream-downstream interactions but we can ensure that water withdrawals at basin level do not exceed available water and that EF volumes are met in each simulation unit.

Our results are conditional on the underlying optimization structure and model parameterization. Validation against historical data demonstrates that the default parameterization of the MAgPIE model can reproduce key characteristics of the real system. Sensitivity with respect to future socioeconomic conditions is addressed by choosing two SRES scenarios. The sensitivity analysis with respect to technological change (TC) costs suggests that the potential for increasing yields and irrigation efficiencies via TC is important for mitigating impacts of EFP on land-use dynamics. Further sensitivity analysis shows that model results are more sensitive to annual costs of irrigation than to investment costs for irrigation infrastructure. The balance between cropland expansion and intensification depends on the availability of suitable land. We do not consider dynamics of urban areas since they only account for ~1% of global land-area (Erb et al., 2007). We assume pasture areas remain constant as the extent has only changed by around 27 Mha (<1%) in the period 1989–2009 (FAOSTAT, 2013) and future projections of pasture dynamics do not show a clear trend toward expansion or contraction (Schmitz et al., 2014; Smith et al., 2010). We do not explicitly model forest management in the current model version and keep forest areas needed for wood production and nature conservation static over time (around 43% of global forest area in 1995). It is not clear, how forest dynamics will affect future land availability since a wide range of plausible projections of future wood demand exist and future wood harvest yields are uncertain (Smeets and Faaij, 2007). Finally, it has been shown that dietary shifts and waste reductions can decrease the pressure on the agricultural system (Godfray et al., 2010; Popp et al., 2010; Smith et al., 2013). Allowing for such adaptation measures would likely decrease the impact of EFP on land-use change compared to our setup with exogenous food and material demand.

5. Conclusions

Today, we see an alarming decline of aquatic biodiversity (Pahl-Wostl et al., 2013b) and a lack of appropriate responses in terms of the implementation of sustainable water management policies (Pahl-Wostl et al., 2013a). This has led to the proposition of a scientific agenda concentrating on the generation of robust knowledge concerning cross-sectoral implications of water management strategies to support the transition toward a sustainable water future (Pahl-Wostl et al., 2013c). In this article, we estimate how global allocation of annual volumes of water for EFP may affect global agricultural water-use and land-use up to the middle of the century. While our study focuses exclusively on water quantity, comprehensive EFP policies should also target water quality and the intra-annual timing of flows. Further research is therefore needed in order to incorporate these aspects into global analyses of the effects of EFP on land-use and agricultural water-use.

It has been shown recently that the amount of water available for human use strongly depends on the definition of environmental water requirements (Gerten et al., 2013). Our results show that the feedback of EFP policies on the land-use system also depends on the underlying protection scenario in terms of absolute numbers, as well as with regard to the spatial distribution of impacts. We can, however, identify robust findings that prevail across different protection scenarios and socioeconomic conditions.

Our results suggest that conserving annual volumes of water for EFP can be achieved without major losses of forests and other unmanaged land at the global level. Socioeconomic changes have been shown to put considerably higher pressure on land resources than EFP. This indicates that EFP does not fundamentally oppose sustainable land management at the global scale. Limiting the negative feedback of EFP on land ecosystems will likely require agricultural intensification on existing cropland. Such intensification should target productivity of all production factors, including water and nutrients in order to avoid negative environmental impacts. In this context, it is promising that agricultural research and development is increasingly focused on reducing the environmental impacts of intensive agriculture (Alston et al., 2009).

From a local perspective, changes in comparative advantages, associated production reallocation and local hotspots of land-use change may have a strong influence on livelihoods and environmental sustainability. We estimate that securing annual volumes of water for EFP leads to cropland expansion into natural land ecosystems in locations where agricultural water-use is not directly affected by EFP. This indicates that a river basin perspective on water management, as advocated by Lawford et al. (2013), may miss land-use feedbacks that can put the sustainability of local water management strategies into question. Furthermore, recent distortions of the global food market have resulted in food self-sufficiency programs in several countries including India, Japan and Qatar (Fader et al., 2013). In this context, our result showing that EFP may lead the reallocation of production away from water-scarce regions gives rise to the concern that regional and local food self-sufficiency ambitions may hamper the implementation of EFP policies.

In summary, our results indicate that global reservation of annual volumes of water for EFP can be achieved with moderate consequences for land resources. In reality, water is, however, managed by local or national authorities. Concerns about production reallocation and local land-use implications may therefore prevent the implementation of EFP.

Acknowledgements

The authors would like to thank three anonymous reviewers for their valuable comments. The research leading to these results has received funding from European Community's Seventh Framework Program (FP7/2007–2013) under Grant Agreement Nos. 266992 (Global-IQ) and 308329 (ADVANCE). Additional funding was provided by the BMBF through the INNOVATE project (Grant Agreement 01LL0904D).

Appendix A. Additional methods and scenario descriptions

A.1. MAgPIE regions and simulation units

See Figs. A1 and A2.

A.2. Irrigation costs

We use the rent associated with irrigation water application (i.e. factor payments for irrigation water) as a proxy for the annual factor requirement costs of irrigation. In order to derive the rent of irrigation water application from the GTAP land rent (Narayanan

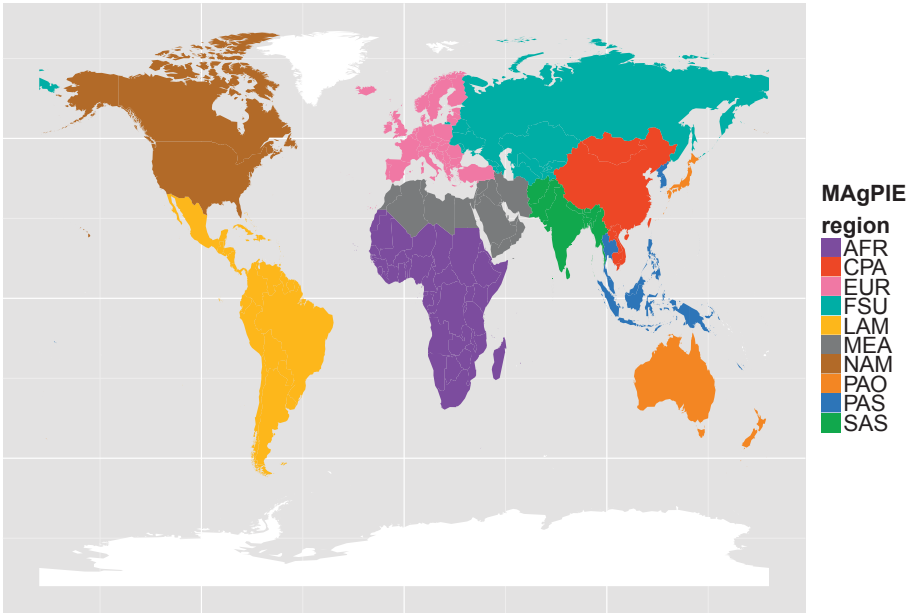

Fig. A1. MAgPIE world regions. AFR = Sub Saharan Africa, CPA = centrally planned Asia, EUR = Europe, FSU = former Soviet Union, LAM = Latin America, MEA = Middle East and North Africa, NAM = North America, PAO = Pacific OECD, PAS = Pacific Asia, SAS = South Asia. Greenland and Antarctica are not covered by MAgPIE.

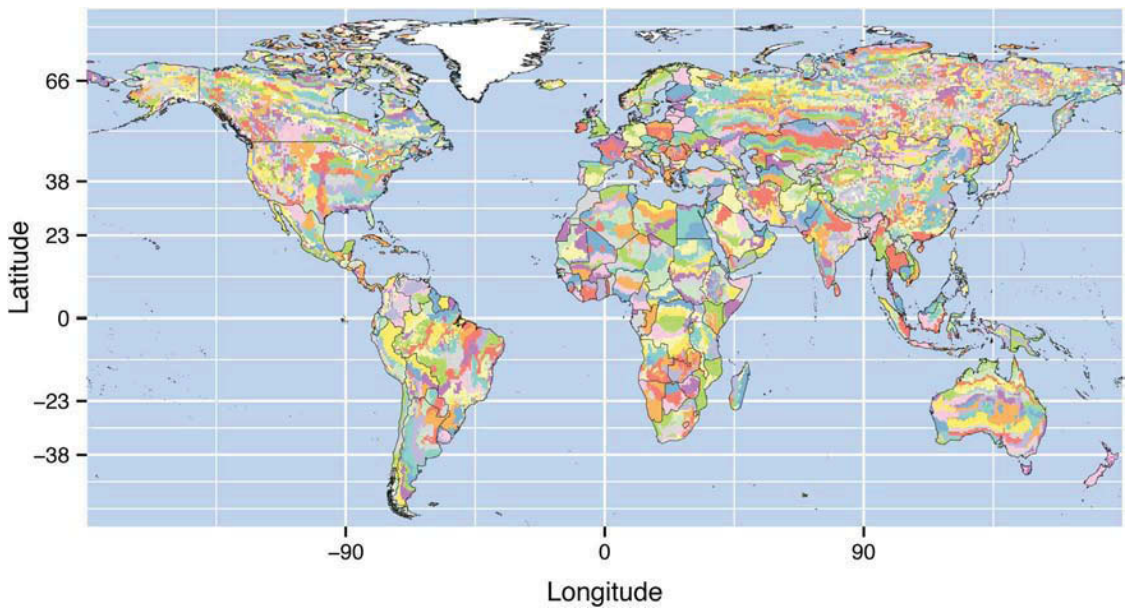

Fig. A2. MAgPIE simulation units. The 1000 simulation units are aggregates of 0.5 × 0.5 latitude/longitude grid cells based on similarity of biophysical conditions (Dietrich et al., 2013) and differ in size.

and Walmsley, 2008), we largely follow an approach by Calzadilla et al. (2011).

In a first step, the ratio of irrigated production ($prod_{ir}$) to total production ($prod_{tot}$) based on LPJmL yields and the crop pattern from the MIRCA dataset (Portmann et al., 2010) is calculated. This share is multiplied by the total land rent from GTAP (R_{tot}) to obtain the land rent of irrigated production (R_{ir}) (Eq. (A1)). The calculation is done for all GTAP crop categories (*crop*) and countries (*cntr*) separately.

$$R_{ir(cntr,crop)} = \frac{prod_{ir}(cntr, crop)}{prod_{tot}(cntr, crop)} * R_{tot}(cntr, crop) \quad \text{(A1)}$$

Using physical production as a weight for splitting the land rent assumes that the economic value is equal for all units produced.

This is usually not the case as prices for agricultural products vary strongly over the season. For instance, irrigation is often used to increase production of a commodity at a time when it is scarce and the price is high. This leads to a higher economic value of irrigation than physical production numbers would suggest. As a result, our approach may lead to a slight underestimation of the land rent associated with irrigated production.

In a second step, the land rent of irrigated production is further split into the value share of irrigable land (R_{ir}^{land}) and the value share of irrigation water (R_{ir}^{wat}). To calculate R_{ir}^{land}, we use the ratio

of rainfed and irrigated yields Y_{rf} and Y_{ir} and multiply it with the total rent of irrigated production (Eq. (A2)).

$$R_{ir}^{land}(cntr, cro\,p) = \frac{Y_{rf}(cntr, cro\,p)}{Y_{ir}(cntr, cro\,p)} R_{ir}(cntr, cro\,p) \quad (A2)$$

$$R_{ir}^{wat}(cntr, cro\,p) = R_{ir}(cntr, cro\,p) - R_{ir}^{land}(cntr, cro\,p) \quad (A3)$$

The rent associated with irrigable land (R_{ir}^{land}) can be interpreted as the rent that could have been obtained by doing rainfed production on the land actually used for irrigated production. The remainder (R_{ir}^{wat}) of the rent of irrigated production is consequently assigned to the actual application of irrigation water (Eq. (A3)). In reality, irrigation takes place in locations where the economic benefit is highest, i.e. where the ratio of rainfed to irrigated yields is low. Applying the mean country ratio of rainfed to irrigated yields therefore tends to overestimate the value of irrigable land R_{ir}^{land}. To avoid this bias, we refined the algorithm by using the yield ratio only on area equipped for irrigation (Siebert et al., 2007). Wherever there is no information available for a specific country – crop combination, the world average value of that crop is used for computation. Finally, the country values are aggregated to the MAgPIE regions using a production weighted mean and the GTAP crops are aggregated to the MAgPIE crop categories (Tables A1 and A2).

A.3. Environmental flow calculations

This is a detailed description of the calculations for available water and environmental water requirements. All hydrological data is obtained from LPJmL. The index j stands for the cell, b is for the river basin and m for the month. Hydrological data is provided by LPJmL.

The calculation of monthly available water, MAW(j,m), is done based on monthly discharge, $Q(j,m)$, and monthly runoff, $R(j,m)$, according to Eq. (A4).

$$\mathrm{MAW}(j,m) = \sum_{c \in b(j)} R(c,m) * \frac{Q(j,m)}{\sum_{c \in b(j)} Q(c,m)} \quad (A4)$$

Monthly environmental flow requirements, EFR(j,m), for the Tennant method (Tennant, 1976) are calculated according to Eq. (A5).

$$\mathrm{EFR}(j,m) = \mathrm{MAW}(j,m) * 0.3 \quad (A5)$$

Table A1
Investment costs for expanding irrigation infrastructure in US$ per hectare.

	1995	2005	2015	2025	2035	2045
AFR	37,173	31,444	25,715	19,986	14,257	8528
CPA	8781	8214	7647	7080	6514	5947
EUR	5663	5663	5663	5663	5663	5663
FSU	5663	5663	5663	5663	5663	5663
LAM	12,237	11,042	9846	8651	7456	6261
MEA	5933	5884	5835	5786	5737	5688
NAM	5663	5663	5663	5663	5663	5663
PAO	5663	5663	5663	5663	5663	5663
PAS	2078	2730	3382	4033	4685	5337
SAS	1899	2584	3268	3952	4637	5321

Monthly low-flow requirements for the Smakhtin (Smakhtin et al., 2004) method (LFR(j,m)) are calculated based on Q90(j), the 90% quantile of monthly discharge over one year (Eq. (A6)).

$$\mathrm{LFR}(j,m) = \mathrm{MAW}(j,m) * \frac{\min(Q(j,m), \mathrm{Q90}(j,m))}{Q(j,m)} \quad (A6)$$

Associated high-flow requirements, HFR(j,m), are subsequently derived using Eq. (A7).

$$\mathrm{HFR}(j,m) = \begin{cases} 0,2 * \mathrm{MAW}(j,m) & \text{if } \frac{\mathrm{Q90}(j,m)}{Q(j,m)} < 0.1 \\ 0,15 * \mathrm{MAW}(j,m) & \text{if } 0.1 \le \frac{\mathrm{Q90}(j,m)}{Q(j,m)} < 0.2 \\ 0,07 * \mathrm{MAW}(j,m) & \text{if } 0.2 \le \frac{\mathrm{Q90}(j,m)}{Q(j,m)} < 0.3 \end{cases} \quad (A7)$$

Total monthly EFR are the sum of low and high flow components: EFR(j,m) = LFR(j,m) + HFR(j,m).

For the Hanasaki method (Hanasaki et al., 2008), calculations are based on QC(j,m) (in mm per month), the monthly discharge divided by the catchment area, i.e. the area of all upstream cells (according to personal communication with authors). As shown in Table 1, cells are classified into four categories according to minimum and maximum annual discharge. A category and cell specific fraction of available water $F(j,m)$ is then used as the cellular environmental flow requirement (Eq. (A8)). The values for the protected fraction depend on monthly discharge and are shown in Table 1.

$$\mathrm{EFR}(j,m) = \mathrm{MAW}(j,m) * F(j,m) \quad (A8)$$

Table A2
Annual factor requirement costs for irrigation for all regions and crops in US$ per hectare and year.

	AFR	CPA	EUR	FSU	LAM	MEA	NAM	PAO	PAS	SAS
Temperate cereals	31	109	96	34	48	39	51	70	97	56
Tropical cereals	10	71	67	11	30	22	52	52	30	15
Maize	10	55	46	19	17	52	53	78	20	15
Rice	26	118	117	43	48	123	88	195	57	57
Others	83	441	295	187	178	273	482	965	138	203
Potato	332	695	805	389	396	764	1098	2011	609	696
Cassava	46	162	94	72	72	397	92	230	67	129
Pulses	59	271	280	177	81	125	220	248	153	104
Soybean	21	80	105	20	77	76	71	105	36	40
Rapeseed	222	400	597	92	304	325	246	439	728	229
Groundnut	36	188	119	87	70	193	120	175	66	66
Sunflower	105	398	243	141	286	140	199	229	196	125
Oil palm	75	381	193	193	266	193	193	193	278	193
Sugar beet	77	65	111	37	117	80	84	212	77	56
Sugar cane	218	401	321	319	287	883	301	765	318	361
Cotton	60	280	256	145	111	284	127	423	200	103

Finally, annual environmental flow requirements AEFR(j) are obtained by summing the monthly values and truncating at 50% of annually available water based on (Revenga et al., 2004) (Eq. (A9)).

$$\mathrm{AFER}(j) = \min\left(0.5 * \sum_m \mathrm{MAW}(j,m), \sum_m \mathrm{EFR}(j,m)\right) \quad \text{(A9)}$$

A.4. Food, livestock and material demand

See Fig. A3.

Appendix B. Additional model results for baseline runs

B.1. Agricultural water withdrawals

Without environmental flow protection (EFP), increasing food demand leads to an increase of agricultural water withdrawals (AWW) from 2842 km^3 in 1995 to 3815 km^3 (A2) and 4010 km^3 (B1) in 2045 (Fig. B1). Our base year estimate of agricultural water withdrawals is at the higher end of historical estimates. It is however well consistent with data provided by (Wisser et al., 2008) who used different irrigated area patterns and climate datasets to estimate AWW. The slope of our projection is similar to historical trends estimated by (Shiklomanov, 2000).

Even though food demand increases for the A2 scenario are stronger than for B1 (Fig. B1), agricultural water withdrawals are higher in B1 by the mid of the century. The reason behind this behavior is the stronger increase in competing water uses under

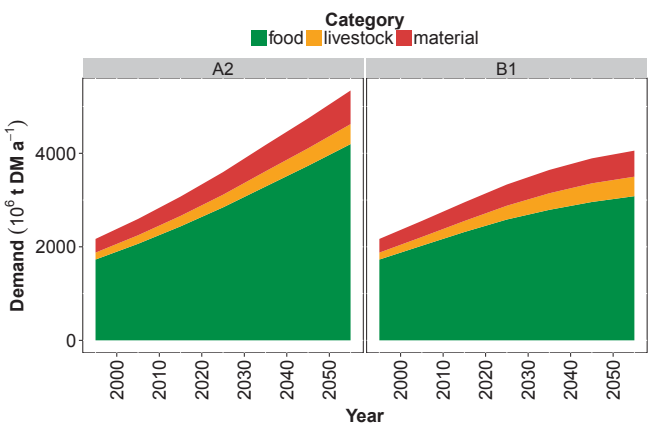

Fig. A3. Global demand for vegetal food, livestock products and material for the two socioeconomic scenarios in million tons dry matter per year (Bodirsky et al., 2012).

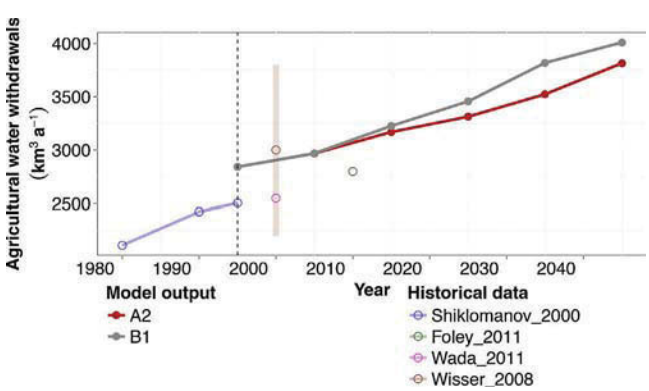

Fig. B1. Global agricultural water withdrawals without EFP under the two socioeconomic scenarios. Estimates of historical and current agricultural water withdrawals by Shiklomanov (2000) (Shiklomanov_2000), Foley et al. (2011) (Foley_2011), Wada et al. (2011) (Wada_2011) and Wisser et al. (2008) (Wisser_2008). The brown shaded bar corresponds to the uncertainty estimate provided by Wisser et al. A vertical dashed line marks the start of the simulation. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

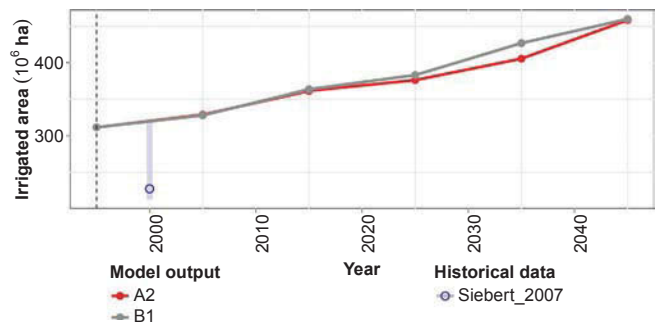

Fig. B2. Global irrigated area development without EFP under the two socioeconomic scenarios. The blue shaded bar corresponds to the range of estimated historical irrigated area as reported by Siebert and Döll (2007) (Siebert_2007). A vertical dashed line marks the start of the simulation. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

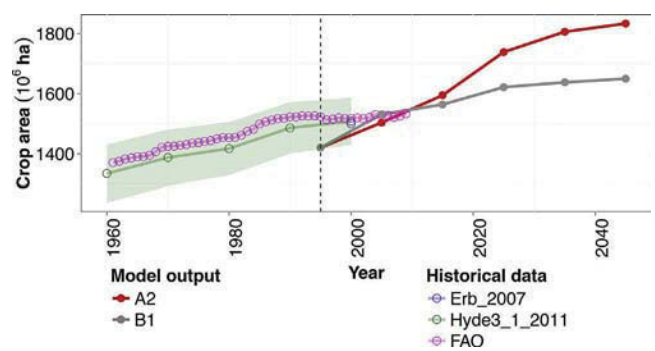

Fig. B3. Global cropland development without EFP under the two socioeconomic scenarios. Estimates of historical cropland by Erb et al. (2007) (Erb_2007, blue), Klein Goldewijk et al. (2011) (Hyde3_1_2011, green with uncertainty range) and FAOSTAT (2013) (FAO, purple) for comparison. A vertical dashed line marks the start of the simulation. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

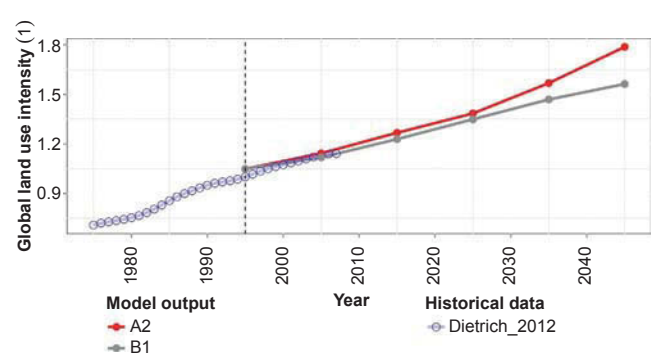

Fig. B4. Global land-use intensity for the baseline scenarios without EFP. Increases over the simulation period reflect investments into yield increasing technological change (TC). Historical data from Dietrich et al. (2014). A vertical dashed line marks the start of the simulation.

the A2 scenario (Fig. 2) that limits the water available for agriculture. This is in contrast to results by (Shen et al., 2008), who project higher AWW in 2055 under the A2 scenario (4691 km^3) than under the B1 scenario (3683 km^3). Their analysis assumes constant irrigated area per capita and neglects water availability constraints and competing water demand from other sectors.

B.2. Irrigated area

Irrigated area develops similar to agricultural water withdrawals, increasing from 312 Mha in 1995 to 460 Mha in 2045 under both socioeconomic scenarios (Fig. B2). Our initial irrigated area is at the higher end of the range of irrigated area estimates as reported by (Siebert and Döll, 2007). Irrigated area

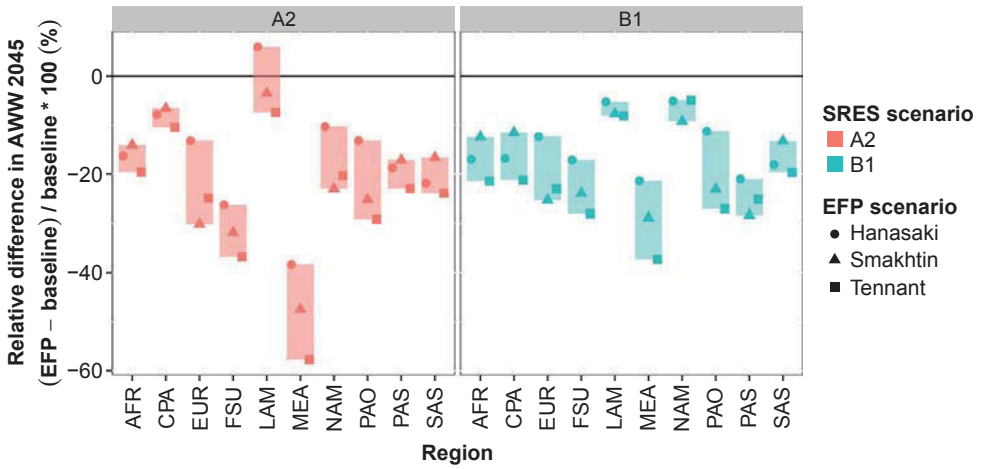

Fig. C1. Regional changes in agricultural water withdrawals (AWW) due to EFP in 2045 relative to baseline agricultural water withdrawals for the two socioeconomic scenarios. Points indicate the value for the individual EFP scenarios. Shaded areas correspond to the whole range of changes due to EFP under each socioeconomic scenario.

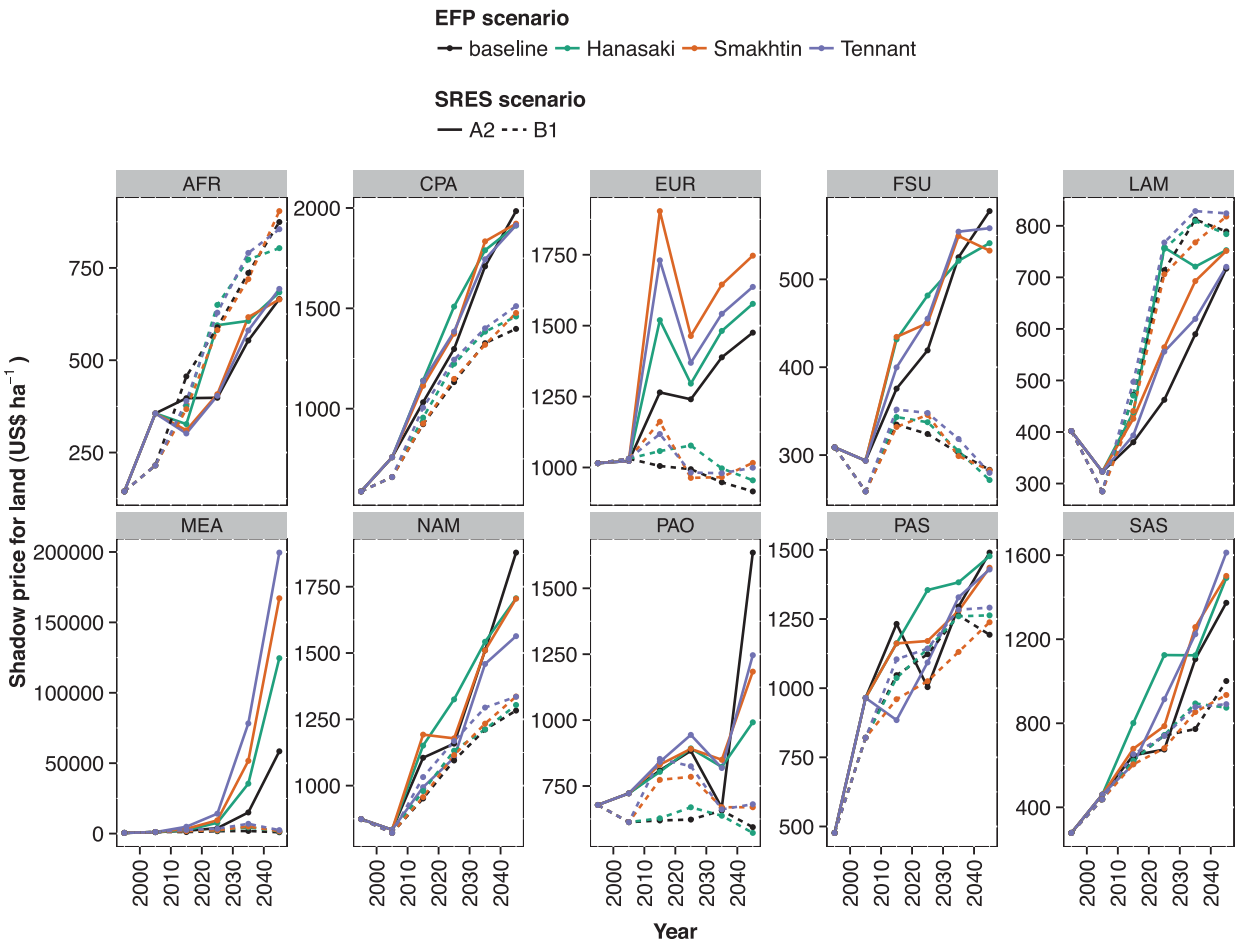

Fig. C2. Average regional shadow prices for relaxing the land constraint by one unit. Shadow prices are aggregated to regional level using a cropland weighted mean.

increases by ∼20% until 2025. This is slightly a stronger increase than found by (Sauer et al., 2010) who estimate irrigated area in 2030 to be 15% higher than in the year 2000. Even though agricultural water withdrawals differ between the two scenarios, crop- as well as location-specific irrigation water demand allows the model to optimize irrigation patterns so that total irrigated area is similar for both scenarios. Moreover, investments into technological change are higher in the A2 scenario (Fig. B4) resulting in improved water use efficiency.

B.3. Cropland

Global cropland projections for A2 are substantially higher than for B1 after 2015 (Fig. B3) because of the stronger increase in food demand (Fig. A2). Our base year cropland (1420 Mha) is calibrated to the average FAO cropland (category "Arable land and permanent crops") for the period 1990–1999 (FAOSTAT, 2013) at regional level. Global cropland area reaches 1833 Mha in 2045 under the A2 scenario and 1650 Mha under B1.

B.4. Land-use intensity

Global land-use intensity, i.e. global yield levels, increased by around 1.5% per year between 1970 and 2007. In our projections, increases continue until the mid of the century at a rate of ∼1% for A2 and 0.9% for B1. This is lower than the increases we have seen in the past.

Appendix C. Additional model results on the effects of environmental flow protection

C.1. Regional changes in Agricultural water withdrawals due to environmental flow protection

See Fig. C1.

C.2. Shadow prices for land

See Fig. C2.

C.3. Shadow prices for water

See Fig. C3.

C.4. Regional land available for cropland expansion

See Fig. C4.

C.5. Changes in regional production due to environmental flow protection

See Fig. C5.

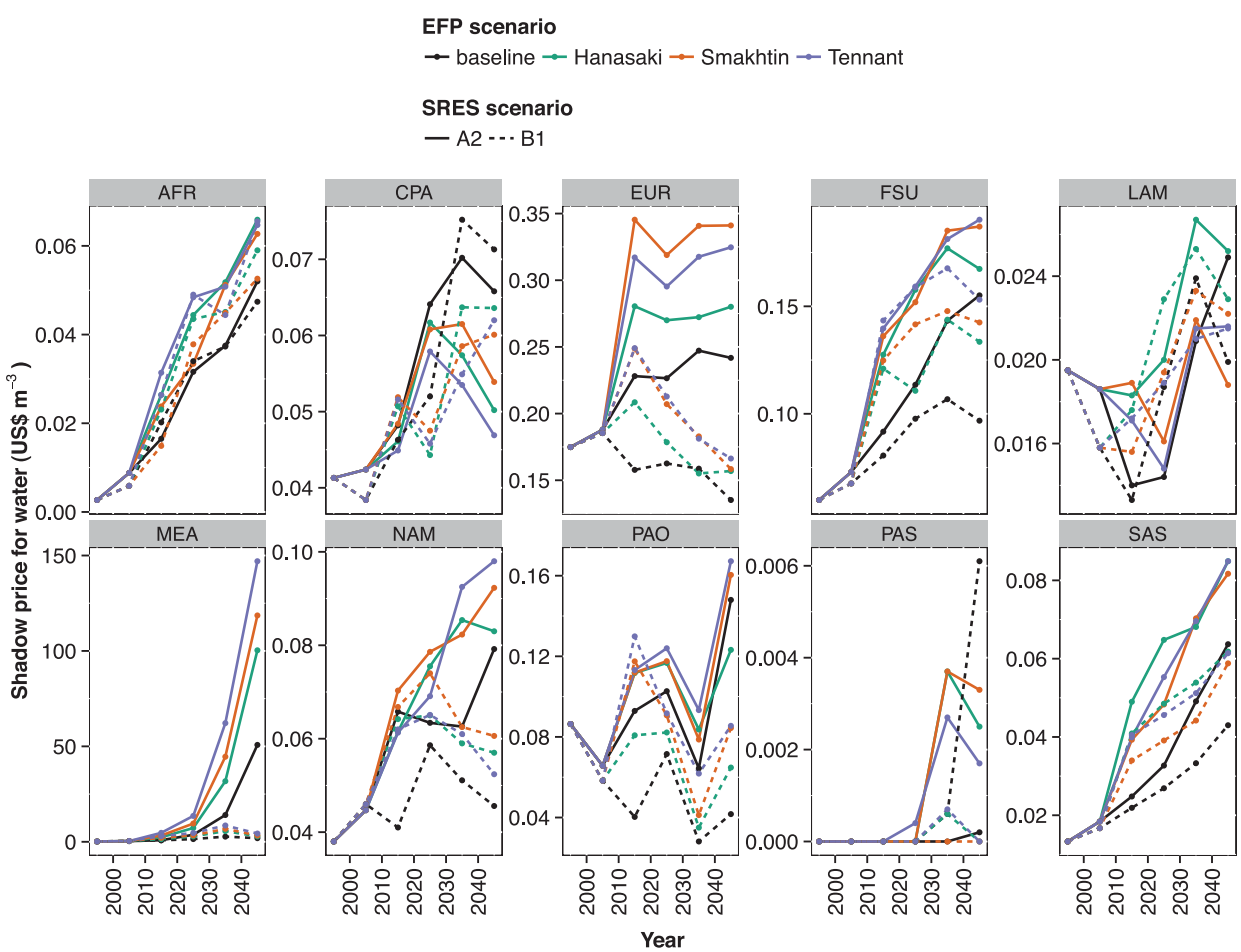

Fig. C3. Average regional shadow prices for relaxing the water constraint by one unit. Aggregation to regional level using an agricultural water withdrawal weighted mean.

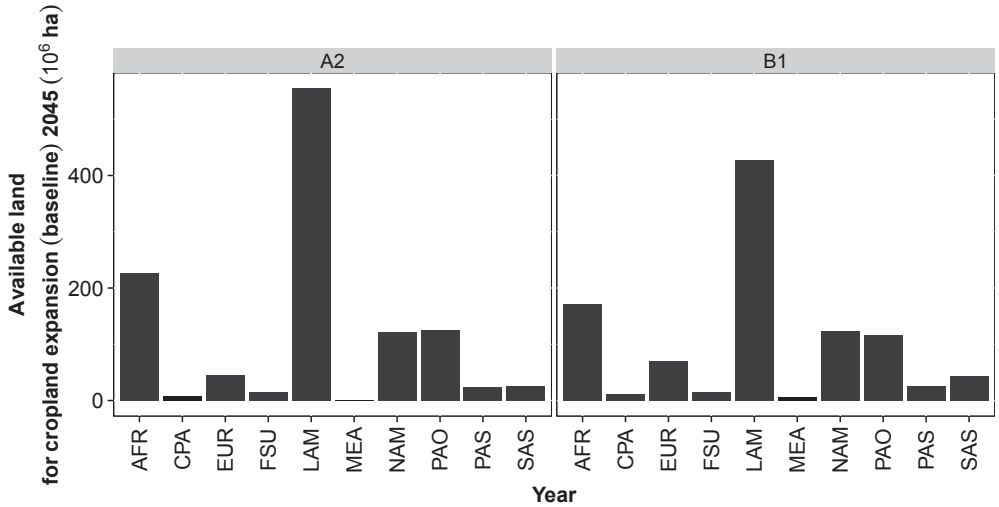

Fig. C4. Regional land available for cropland expansion in 2045 in the baseline scenarios without EFP. Available land comprises unprotected natural forests (Section 2.2.2) and other unmanaged land.

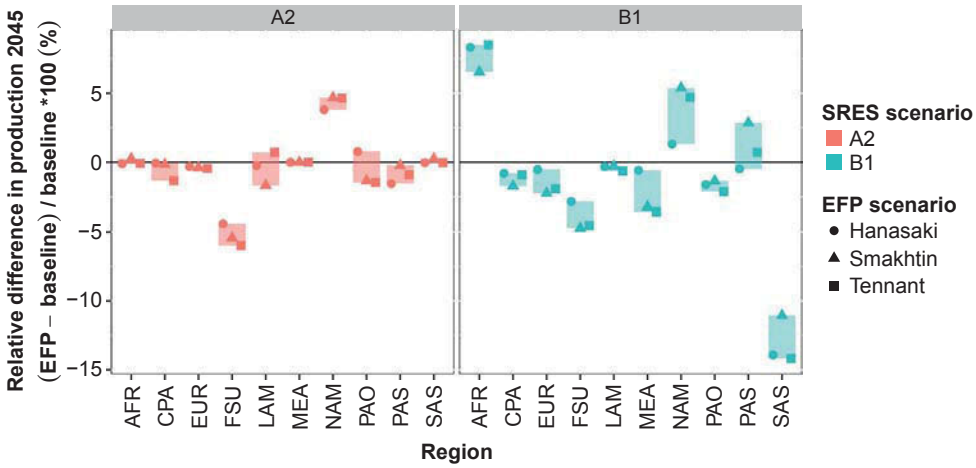

Fig. C5. Regional changes in production due to EFP in 2045 relative to baseline production for the two socioeconomic scenarios. Points indicate the value for the individual EFP scenarios. Shaded areas correspond to the whole range of changes due to EFP under the respective socioeconomic scenario.

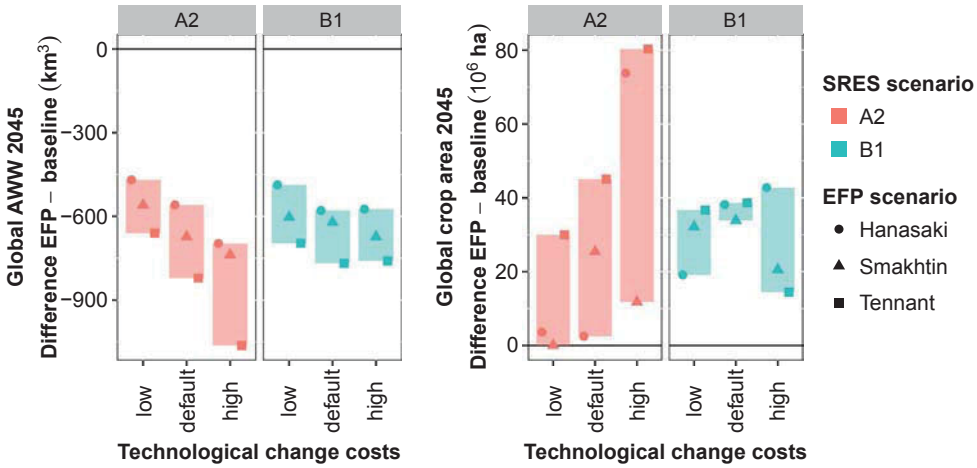

Fig. D1. Changes in global agricultural water withdrawals (AWW, left) and crop area (right) due to EFP in 2045. Three different assumptions on the costs of yield increasing technological change (low, default, high) are differentiated on the *x*-axis. Colors differentiate socioeconomic scenarios. EFP scenarios are distinguished by the shape of points. Shaded areas span the full range of EFP impacts under the respective socioeconomic scenario. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

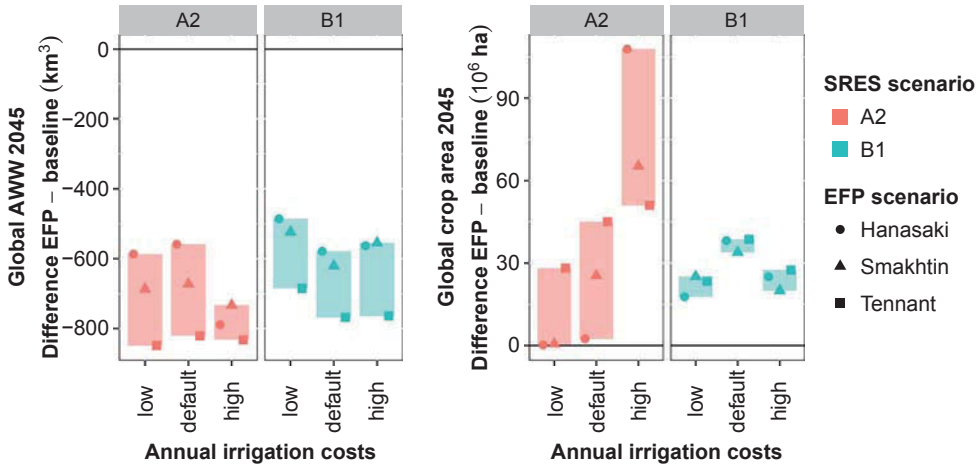

Fig. D2. Changes in global agricultural water withdrawals (AWW, left) and crop area (right) due to EFP in 2045. Three different assumptions on the annual operation and maintenance costs irrigation systems (low, default, high) are differentiated on the *x*-axis. Colors differentiate socioeconomic scenarios. EFP scenarios are distinguished by the shape of points. Shaded areas span the full range of EFP impacts under the respective socioeconomic scenario. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

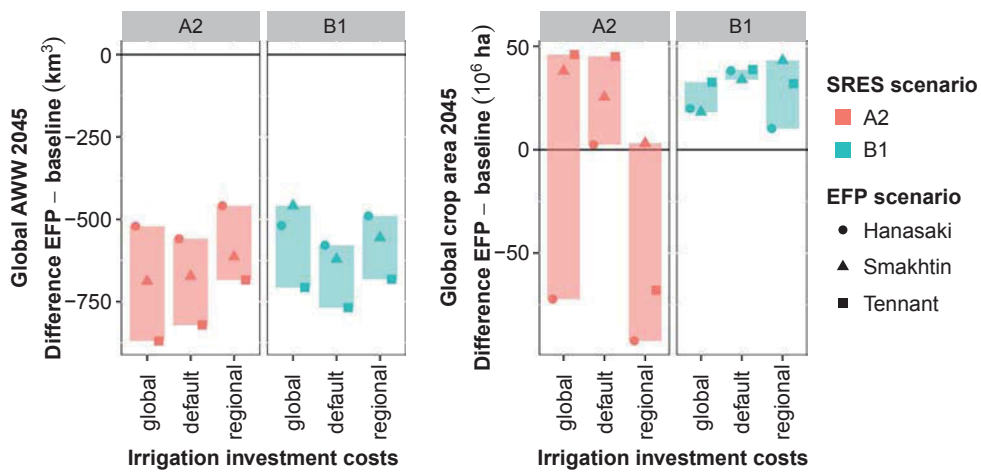

Fig. D3. Changes in global agricultural water withdrawals (AWW, left) and crop area (right) due to EFP in 2045. Three different assumptions on investment costs for installing new irrigation systems (regional, default, global) are differentiated on the *x*-axis. Colors differentiate socioeconomic scenarios. EFP scenarios are distinguished by the shape of points. Shaded areas span the full range of EFP impacts under the respective socioeconomic scenario. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

Appendix D. Sensitivity analysis

D.1. Technological change costs

Higher costs for yield increasing technological change (TC) amplify the effect of EFP on AWW (reductions due to EFP up to 1060 km^3; Fig. D1, left) and cropland expansion (expansion due to EFP up to 80 Mha; Fig. D1, right) under the A2 scenario. Cheaper TC leads to a reduction of EFP implications for AWW (reductions due to EFP up to 660 km^3) and cropland extent (expansion due to EFP up to 30 Mha). In B1, the difference between assumptions on TC costs is less pronounced.

D.2. Annual irrigation costs

We find that annual irrigation cost assumptions have little impact on the implications of EFP on global AWW (Fig. D2, left). Higher costs can however lead to increased cropland expansion due to EFP under the A2 scenario, especially for the Hanasaki EFP scenario (107 Mha). The picture is different under the B1 scenario, where higher costs as well as lower costs lead to decreased cropland expansion due to EFP (below 30 Mha in 2045).

D.3. Investment costs for irrigation infrastructure

The implications of EFP for AWW are not very sensitive to changes in irrigation investment costs (Fig. D3, left). Changes in cropland due to EFP show a stronger dependence on the EFP scenario for regional and global investment cost assumptions than for the default value, especially for the A2 scenario. With global irrigation investment costs, EFP can lead to a decrease in crop area by up to 90 Mha in 2045 for the Hanasaki scenario. The sensitivity of model results with respect to irrigation investment costs in B1 is less pronounced than in A2.

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Chapter V

Trade-offs between land and water requirements for large-scale bioenergy production *

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* published in Global Change Biology – Bioenergy, November 2014, DOI: 10.1111/gcbb.12226

Content of Chapter V: Trade-offs between land and water requirements for large-scale bioenergy production

GLOBAL CHANGE BIOLOGY
BIOENERGY

GCB Bioenergy (2014), doi: 10.1111/gcbb.12226

Trade-offs between land and water requirements for large-scale bioenergy production

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Abstract

Bioenergy is expected to play an important role in the future energy mix as it can substitute fossil fuels and contribute to climate change mitigation. However, large-scale bioenergy cultivation may put substantial pressure on land and water resources. While irrigated bioenergy production can reduce the pressure on land due to higher yields, associated irrigation water requirements may lead to degradation of freshwater ecosystems and to conflicts with other potential users. In this article, we investigate the trade-offs between land and water requirements of large-scale bioenergy production. To this end, we adopt an exogenous demand trajectory for bioenergy from dedicated energy crops, targeted at limiting greenhouse gas emissions in the energy sector to 1100 Gt carbon dioxide equivalent until 2095. We then use the spatially explicit global land- and water-use allocation model MAgPIE to project the implications of this bioenergy target for global land and water resources. We find that producing 300 EJ yr^{-1} of bioenergy in 2095 from dedicated bioenergy crops is likely to double agricultural water withdrawals if no explicit water protection policies are implemented. Since current human water withdrawals are dominated by agriculture and already lead to ecosystem degradation and biodiversity loss, such a doubling will pose a severe threat to freshwater ecosystems. If irrigated bioenergy production is prohibited to prevent negative impacts of bioenergy cultivation on water resources, bioenergy land requirements for meeting a 300 EJ yr^{-1} bioenergy target increase substantially (+ 41%) – mainly at the expense of pasture areas and tropical forests. Thus, avoiding negative environmental impacts of large-scale bioenergy production will require policies that balance associated water and land requirements.

Keywords: bioenergy, land, land-use model, projection, sustainability, water, water-land nexus

Received 6 June 2014 and accepted 29 August 2014

Introduction

A recent model intercomparison study projects bioenergy deployment between 70 and 230 Exajoules (EJ) per year in 2100 for scenarios without climate policy, with bioenergy primarily used to produce liquid fuels for the transport sector (Rose *et al.*, 2014). With climate policies aiming at ambitious mitigation targets, bioenergy demand in 2100 is projected to reach 200–320 EJ yr^{-1} (Rose *et al.*, 2014) since bioenergy in combination with carbon capture and storage can remove carbon dioxide from the atmosphere (Azar *et al.*, 2006).

This bioenergy demand is in the same order of magnitude as the gross energy value of all harvested biomass in the year 2000 of around 300 EJ (Haberl *et al.*, 2007). Therefore, concerns about negative environmental and societal implications of large-scale bioenergy production are discussed. It is expected that bioenergy production may require up to 550 Mha of additional cropland (Popp *et al.*, 2014), corresponding to around 35% of current total cropland (FAO, 2013). Such substantial land requirements may have negative impacts on greenhouse gas emissions (Searchinger *et al.*, 2008; Popp *et al.*, 2011a, 2012), food prices (Lotze-Campen *et al.*, 2014) and biodiversity (Smith *et al.*, 2013).

Land requirements for bioenergy production are highly dependent on the achievable yield. Cultivation of dedicated bioenergy crops is very water intensive (Berndes, 2002; Gerbens-Leenes *et al.*, 2009), so water limitations are a key constraint for achievable bioenergy yields in rainfed production systems. Reducing the water deficit by applying additional irrigation water plays a crucial role in achieving high yields (Smith *et al.*,

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2012). Beringer *et al.* (2011) estimate that irrigation has the potential to increase bioenergy yields by more than 100% compared to rainfed production systems in large parts of India, Africa, Latin America, North America, and Australia. Irrigation may therefore be an option to reduce the pressure of bioenergy production on land resources.

Additional irrigation water requirements can however fundamentally change the global water cycle and put additional pressure on water resources. Potential water requirements of large-scale irrigated bioenergy production may be in the same order of magnitude as current total agricultural water withdrawals (Berndes, 2002; Beringer *et al.*, 2011) or even up to twice as high (Chaturvedi *et al.*, 2013). This is critical as many regions already face water scarcity (Falkenmark & Molden, 2008; Arnell *et al.*, 2011) and freshwater ecosystems are degraded by human activity (Poff & Zimmerman, 2010; Grafton *et al.*, 2012). We may thus face a fundamental trade-off between global land and water requirements for bioenergy production. A quantitative assessment of this trade-off is however lacking to date.

We investigate the land- and water-use implications of bioenergy production under two different scenarios using the spatially explicit land- and water-use allocation model MAgPIE (Model of Agricultural Production and its Impacts on the Environment) (Lotze-Campen *et al.*, 2008; Popp *et al.*, 2010). In the first scenario, no restrictions on irrigated bioenergy production are imposed. The share and spatial allocation of irrigated bioenergy production is determined endogenously based on economic optimization. The second scenario includes a policy that prohibits irrigated bioenergy cultivation. This setup allows us to quantify the implications of water-saving bioenergy production strategies for global land-use dynamics and natural land ecosystems.

Materials and methods

MAgPIE model

General description. Model of Agricultural Production and its Impacts on the Environment is a spatially explicit, global land- and water-use allocation model and simulates land-use dynamics in 10-year time steps until 2095 using recursive dynamic optimization (Lotze-Campen *et al.*, 2008; Popp *et al.*, 2010). The objective function of MAgPIE is the fulfilment of food, feed and material demand at minimum costs under socio-economic and biophysical constraints. Demand trajectories are based on exogenous future population and income projections (see Section Scenarios). Major cost types in MAgPIE are: factor requirement costs (capital, labour and chemicals, e.g., fertilizer), land conversion costs, transportation costs to the closest market and investment costs for technological change. Socio-economic constraints like demand, factor requirement costs and investment costs are defined at the regional level (10 world regions) (Figure S1). Biophysical constraints such as crop yields, carbon density, and water availability – derived from the global hydrology and vegetation model LPJmL (Bondeau *et al.*, 2007; Rost *et al.*, 2008; Müller & Robertson, 2014) – as well as land availability (Krause *et al.*, 2013), are introduced at the grid cell level (0.5 degree longitude/latitude; 59 199 grid cells). Due to computational constraints, all model inputs at 0.5 degree resolution are aggregated to 1000 simulation units for the optimization process based on a k-means clustering algorithm (Dietrich *et al.*, 2013). The clustering algorithm combines grid cells to simulation units based on the similarity of biophysical conditions.

MAgPIE features land-use competition based on cost-effectiveness at simulation unit level among the land-use related activities crop, livestock, and bioenergy production. Available land types are cropland, pasture, forest, other land (including nonforest natural vegetation, abandoned agricultural land and desert) and settlements. Cropland (rainfed and irrigated), pasture, forest, and other land are endogenously determined, while settlement areas are assumed to be constant over time. The forestry sector, in contrast to the crop and livestock sectors, is currently not implemented dynamically in MAgPIE. Therefore, timberland needed for wood production – consisting of forest plantations and modified natural forest – is excluded from the optimization (about 30% of the initial global forest area of 4235 Mha). In addition, other parts of forestland, mainly undisturbed natural forest, are within protected forest areas, which cover about 12.5% of the initial global forest area (FAO, 2010). Altogether, about 86% of the world's land surface is freely available in the optimization of the initial time-step.

Crop yields depend not only on biophysical conditions but also on management practices that differ across world regions and can change over time (Dietrich *et al.*, 2012). Therefore, biophysical yield potentials from LPJmL are calibrated to FAO yields (FAO, 2013) in 1995 before they enter MAgPIE. Regional land-use intensities that reflect the status of agricultural management in 1995 are derived from historical data (Dietrich *et al.*, 2012). MAgPIE can endogenously decide to invest into technological change (TC) on a regional level in order to increase land-use intensities, thereby increasing agricultural yields (Dietrich *et al.*, 2014). The ratio of TC investments to yield improvements (investment-yield ratio) is determined from historical data on agricultural Research and Development spending (Pardey *et al.*, 2006), agricultural infrastructure investments (Narayanan & Walmsley, 2008), and yields (FAO, 2013). The investment-yield ratio increases with the land-use intensity (Dietrich *et al.*, 2014), reflecting the fact that low land-use intensities can be improved by closing yield gaps while yield increases in intensive systems require higher investments to push the technology frontier.

The cost minimization problem is solved through endogenous variation in spatial rainfed and irrigated production patterns (subject to regional trade constraints; Schmitz *et al.*, 2012), land conversion (all at simulation unit level) and technological change (at regional level) (Lotze-Campen *et al.*, 2010).

Bioenergy. Present day modern bioenergy for electricity and liquid fuel generation relies mainly on conventional food crops such as maize and sugarcane (first generation bioenergy) (Gerbens-Leenes *et al*., 2009). To avoid competition with food production, techniques are being developed to convert the lignocellulosic components of plant biomass to biofuels (Schmer *et al*., 2008; Gerbens-Leenes *et al*., 2009). This will allow the use of dedicated grassy and woody bioenergy crops (second generation bioenergy) and is expected to increase the energy yield per unit of crop significantly (Gerbens-Leenes *et al*., 2009).

In MAgPIE, bioenergy feedstock consists of dedicated herbaceous and woody lignocellulosic bioenergy crops. Bioenergy demand enters the model as an exogenous trajectory at the global level (see Section Scenarios). Spatial allocation of bioenergy production is an endogenous model decision resulting from the cost minimizing objective function, which takes into account land and water availability as well as bioenergy yields, production costs, and competing demand for food and material.

In MAgPIE, bioenergy crops can be grown in rainfed and irrigated production systems. Rainfed and irrigated bioenergy yields at simulation unit level for the initialization of MAgPIE are derived from LPJmL (Beringer *et al*., 2011). While LPJmL simulations supply data on potential yields, i.e., yields achievable under the best currently available management options, MAgPIE aims at representing actual yields, i.e., yields realizable under actual current management that differs regionally. Therefore, LPJmL bioenergy yields are calibrated on regional level based on FAO yield data (FAO, 2013) and the ratio of regional land-use intensities to the European intensity. This is done because LPJmL potential bioenergy yields are consistent with observations from well-managed test sites in Europe and the United States (Beringer *et al*., 2011) and management intensities in other world regions are generally lower (Dietrich *et al*., 2012). Low calibration factors for Sub-Saharan Africa (0.28) and Latin America (0.38) reflect large yield gaps with respect to best management practices (Table S1).

Highest rainfed herbaceous bioenergy yields occur in the South-Eastern US, China, Pacific Asia and Latin America (Figure S2, Table S1). Irrigation renders regions attractive for bioenergy production, where rainfed yields are strongly water limited (Beringer *et al*., 2011): India, Northern Africa, Southern US and Australia. Within MAgPIE, endogenous investments into yield increasing technological change (see General description) affect all crops equally, including bioenergy crops.

Water and irrigation

In MAgPIE, available water at simulation unit level for domestic, industrial, and agricultural use comprises renewable blue water resources only, i.e., precipitation that enters rivers, lakes, and aquifers (Rost *et al*., 2008). Input data for available water is obtained from LPJmL (details in the Supplementary Online Material). We assume that all renewable freshwater is available for human use, i.e., no water is reserved for environmental purposes. Domestic and industrial water demand enters the model as an exogenous scenario (Figure S3) based on WaterGAP simulations (Alcamo *et al*., 2003; Flörke *et al*., 2013). We assume that domestic and industrial water demand is fulfilled first, effectively limiting water availability for agricultural use (similar to Elliott *et al*., 2013). Within these limits of available water, agricultural water demand for irrigated food, feed, and bioenergy production as well as livestock feeding is determined endogenously based on cost-effectiveness. Spatially explicit per hectare irrigation water requirements for the 16 food crops and two bioenergy crops represented in MAgPIE are obtained from LPJmL (see Supplementary Online Material), while livestock water requirements are based on FAO data (FAOSTAT, 2005). Rainfed crop production relies on green water resources only, i.e., precipitation infiltrated into the soil, and does therefore not affect agricultural irrigation water demand.

Irrigated crop production is not only constrained by water availability but also requires irrigation infrastructure for water distribution and application. The initial pattern of area equipped for irrigation is taken from the AQUASTAT database (Siebert *et al*., 2006). During the optimization process, the model can endogenously deploy additional irrigation infrastructure (see Supplementary Online Material). Irrigation costs comprise investment costs for the deployment of additional irrigation infrastructure as well as annual costs for operation and maintenance of irrigation systems (see Supplementary Online Material). Yield increases through technological change enhance land as well as irrigation water productivity (see Section Scenarios).

Scenarios

Food, livestock, and material demand (Figure S4) is calculated using the methodology described in Bodirsky *et al.* (under review, 2012), as well as SSP 2 population and GDP projections (IIASA, 2013). SSP2 population and GDP projections belong to a 'Middle of the Road' scenario from the Shared Socio-economic Pathways (SSP) scenario family (O'Neill *et al*., 2013) that is being developed as the successor of the widely used Special Report on Emissions (SRES) scenarios from the Intergovernmental Panel on Climate Change (IPCC) (Intergovernmental Panel on Climate Change, 2000) for use in climate change research.

Global primary bioenergy demand is obtained from Popp *et al.* (2011b), a study with a coupled version of MAgPIE and the global energy-economy-climate model REMIND (Leimbach *et al*., 2010). Within this modelling framework, primary bioenergy from dedicated bioenergy crops is used for electricity production via BioCHP (biomass combined heat and power; conversion efficiency 43%) and BIGCC (biomass integrated coal gasification combined cycle; conversion efficiency 31–42%), and liquid fuel production (conversion efficiency 40%). Bioenergy demand is calculated under climate policies that limit greenhouse gas emissions in the energy sector to 1100Gt CO2 equivalent up to 2095 and accounts for 25% of total primary energy in 2095 (Popp *et al*., 2011b). Other renewables (wind, solar, hydropower) also contribute 25% to global primary energy. Demand for primary bioenergy from dedicated bioenergy crops starts at 7 EJ yr^{-1} in 2015, strongly increases in mid-century and reaches a level of ~ 300 EJ yr^{-1} in 2095 (Table 1).

Table 1 Trajectory of global dedicated primary bioenergy demand (EJ yr^{-1}) from Popp *et al.* (2011b)

	2015	2025	2035	2045	2055	2065	2075	2085	2095
EJ yr^{-1}	7	6	12	38	109	225	301	310	307

We investigate two bioenergy production scenarios (Table 2). In BE, dedicated bioenergy cultivation is unrestricted, i.e., the model can endogenously decide to use rainfed and irrigated production systems for bioenergy crops. BE_RF represents a water protection policy, where only rainfed bioenergy cultivation is allowed. Since the focus of this analysis is on resource requirements of bioenergy production and not on climate change mitigation targets, we assume that no climate change policy (e.g., emission pricing) is implemented in the land-use sector. In our standard model implementation, yield increasing technological change increases both, land productivity (output per hectare – tons ha^{-1}) and irrigation water productivity (output per m^3 of irrigation water – tons m^{-3}). For this setup, we assume that per hectare irrigation water requirements (m^3 ha^{-1}) are constant. Thus, technological change enhances irrigation water productivity (tons m^{-3}) by increasing the yield (tons ha^{-1}). To test the stability of our results, we perform a sensitivity analysis with static irrigation water productivity (static WP). For this setup, we assume that the irrigation water demand per ton output (m^3 ton^{-1}) is constant so that yield increases from technological change increase per hectare irrigation water demand (m^3 ha^{-1}). In our standard model, we assume that bioenergy crops can profit from technological change in the same way as food crops. Krausmann *et al.* (2013) however estimate that almost half of the past yield increases were due to increasing the share of harvested biomass to total plant biomass. This is more difficult to achieve for second generation bioenergy crops since all aboveground biomass can be used for energy production. We therefore conduct a sensitivity analysis where we assume that the effect of yield increases from technological change on second generation bioenergy crops is reduced by 50% compared to conventional crops (low Yields).

Results

Bioenergy production

Irrigation plays a key role for bioenergy provision in the BE scenario (Fig. 1). In 2095, 58% of global bioenergy supply stems from irrigated production. The region with the highest share of irrigated bioenergy production is South Asia with an irrigation share of 95% in 2095. Further regions with high irrigation shares are North America (71% in 2095), Sub-Saharan Africa (73%) and Latin America (50%). These high irrigation shares are driven by large differences between irrigated and rainfed yields (Figure S2). China is the only region, where bioenergy is mostly produced in rainfed systems (90% in 2095). Spatial allocation of bioenergy production (Fig. 2) to different world regions is mainly driven by spatial differences in bioenergy yields and varies between the scenarios. In the BE scenario, Latin America is the dominant production region contributing 160 EJ yr^{-1} in 2095. Further important bioenergy production regions are South Asia (40 EJ yr^{-1} in 2095), North America (35 EJ yr^{-1}), Sub-Saharan Africa (30 EJ yr^{-1}), and China (CPA, 30 EJ yr^{-1}). The remaining five regions do not contribute significantly to global bioenergy production (together 8 EJ yr^{-1} in 2095).

In the BE_RF scenario, irrigation of bioenergy areas is prohibited and consequently all bioenergy feedstock is provided from rainfed agriculture. Bioenergy production in Latin America and North America is similar to the BE scenario (160 EJ yr^{-1} and 30 EJ yr^{-1} in 2095 respectively). In South Asia on the contrary, rainfed bioenergy production is not competitive due to low yields (Figure S2) and no bioenergy is produced in the BE_RF scenario. This necessitates additional bioenergy production in other regions compared to BE. Bioenergy production increases significantly in Africa with an additional 35 EJ yr^{-1} in 2095 compared to BE. In China, bioenergy production increases by 5 EJ yr^{-1} and the remaining five regions provide an additional 3 EJ yr^{-1} in 2095. Even though bioenergy yields are high in Pacific Asia (Table S1), no bioenergy is produced there because forest requirements for wood production and

Table 2 Scenario definitions. In the standard model, yield increases affect land and irrigation water productivity. In the sensitivity runs, irrigation water productivity is static (static WP) and bioenergy crop yield increases are reduced to 50% compared to conventional crops (low Yields). Bioenergy cultivation is unrestricted in BE but limited to rainfed production systems in BE_RF

	MAgPIE model	Productivity increases	Bioenergy cultivation
BE	Standard model	Land and water	Rainfed and irrigated
BE_RF			Rainfed only
BE_staticWP	Static WP	Land only	Rainfed and irrigated
BE_RF_staticWP			Rainfed-only
BE_lowYields	Low Yields	Land and water; 50% penalty on bioenergy crops	Rainfed and irrigated
BE_RF_lowYields			Rainfed-only

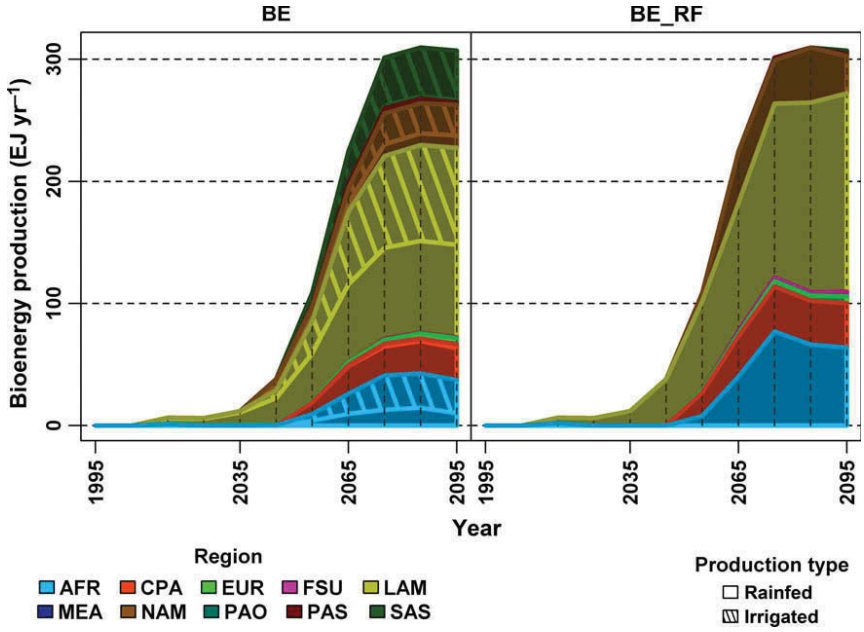

Fig. 1 Regional rainfed and irrigated bioenergy production for BE and BE_RF. AFR = Sub-Saharan Africa, CPA = centrally planned Asia including China, EUR = Europe, FSU = former Soviet Union, LAM = Latin America, MEA = Middle East and North Africa, NAM = North America, PAO = Pacific OECD, PAS = Pacific Asia, SAS = South Asia.

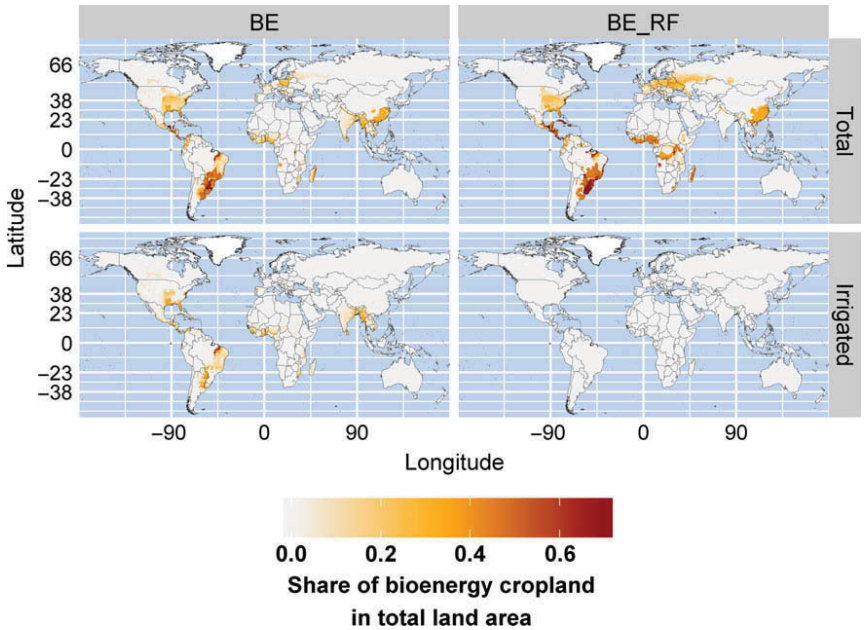

Fig. 2 Spatial allocation of bioenergy cropland for BE (left) and BE_RF (right). Colours indicate the share of bioenergy cropland in total land area. Top row: total bioenergy cropland. Bottom row: irrigated bioenergy cropland.

nature conservation limit land availability (see Section MAgPIE model).

Agricultural water withdrawals

Global agricultural water withdrawals (AWW) for the BE and BE_RF scenarios develop similarly until 2035 (Fig. 3) since almost no bioenergy is produced. Total AWW (energy crops + non-energy commodities) in 1995 are 2926 km^3 and increase to approximately 3250 km^3 in 2035 due to increasing food demand (Figure S4). The development of AWW in our projections in the near future is consistent with the historical trend (Shiklomanov, 2000), but in 1995 our estimate of AWW

is around 400 km^3 higher than the one by Shiklomanov. Our initial value is however consistent with historical data around the year 2000 given the uncertainty range from different irrigated area patterns and climate datasets (2200–3800 km^3), (Wisser *et al.*, 2008).

Due to additional water withdrawals for bioenergy crops, the BE scenario exhibits a steep increase in total AWW after 2035, reaching 6400 km^3 in 2075 and levelling off afterwards. Water withdrawals for irrigated food and material crop production and livestock production (nonenergy commodities) in the BE scenario increase moderately until the mid of the century (3250 km^3 in 2055) due to increasing demand for nonenergy commodities. In the second half of the century, the strong increase in bioenergy demand (Figure S4) leads to increased competition for water. Furthermore, demand for nonenergy commodities stagnates while ongoing yield increases from technological change (Fig. 7) continue to increase irrigation water productivity (see section Scenarios). Irrigation water withdrawals for nonenergy commodities therefore decrease slightly in the second half of the century and amount to 3050 km^3 (48% of total withdrawals) in 2095.

Agricultural water withdrawals in the BE_RF scenario in contrast are solely driven by food and material production and increase more slowly until 2055 (3460 km^3). In BE_RF, bioenergy does not compete with other crops for water, but rainfed bioenergy production can replace irrigated non-energy crops. Therefore, the development of AWW in the second half of the century in BE_RF is similar to water withdrawals for nonbioenergy commodities in BE. The BE_RF scenario requires 53% less irrigation water in 2095 than the BE scenario.

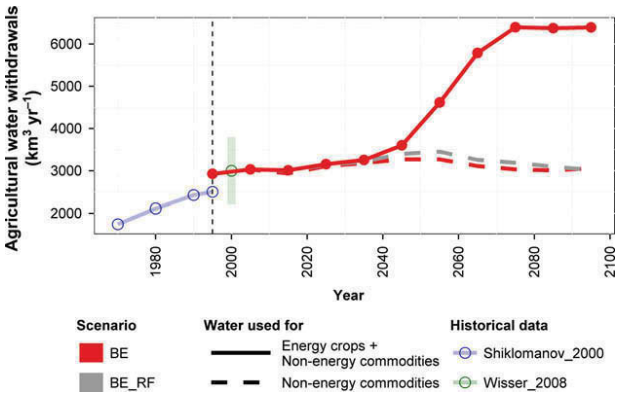

Fig. 3 Global agricultural water withdrawals for BE (rainfed and irrigated bioenergy production allowed) and BE_RF (only rainfed bioenergy production allowed). Total water withdrawals (energy crops + non-energy commodities) appear as a solid line for BE. Dashed lines depict water withdrawals for nonenergy commodities (irrigated food crops, livestock production, and irrigated material crops) only. In BE_RF, water withdrawals for nonenergy commodities equal total agricultural water withdrawals. Historical data from Shiklomanov (2000) and Wisser *et al.* (2008) is displayed for comparison. Wisser *et al.* provide an uncertainty range that is depicted as a shaded area. The vertical dashed line marks the start of the simulation period.

Regional water withdrawals for bioenergy crops in the BE scenario are highest in Latin America, Africa, South Asia, and North America (Fig. 4). AWW for nonenergy commodities in Latin America, Africa, and North America are similar or even higher in BE than in BE_RF indicating that additional water resources are tapped for bioenergy production. In South Asia, bioenergy crops compete directly for water with nonenergy crops indicated by the reduction in AWW for nonenergy commodities in BE compared to BE_RF. This competition for water in South Asia leads to a slight decrease in global AWW for nonenergy commodities in BE compared to BE_RF (Fig. 3).

Land-use change

By the end of the century, bioenergy production will require substantial amounts of land (Fig. 5). In the BE scenario, dedicated bioenergy cropland reaches 490 Mha in 2095. Prohibiting irrigated bioenergy production increases this value by 200 Mha or 41% in the BE_RF scenario. Additional pressure from increasing food demand (Figure S4) drives expansion of cropland for food, feed, and material production (nonenergy cropland) that amounts to 200 Mha (BE) and 180 Mha (BE_RF) until 2095. Increasing bioenergy and nonenergy cropland requirements are fulfilled at the expense of natural forests and pasture. In the BE scenario, global forest and pasture areas decrease by 420 and 470 Mha respectively until 2095. Other land increases by 70 Mha until 2095 due to abandonment of agricultural land. Intensification in the livestock sector leads to reduced demand for animal feedstock from pasture and a reduction in pasture area. Since not all abandoned pasture area is suitable for cropping activities, this process is the main driver for the abandonment of agricultural land. Between 2075 and 2095, bioenergy demand and demand for nonenergy commodities stagnates (Figure S4) while continued technological improvements continue to increase agricultural yields (Fig. 7). Therefore, reductions in bioenergy cropland and nonenergy cropland further contribute to the increase in other land during this period.

The general pattern in the BE_RF scenario is similar to the BE scenario. Additional forest losses are 160 Mha and pasture decreases by an additional 140 Mha until 2095 (Fig. 5). On a regional level, additional forest losses in BE_RF compared to BE are highest in Africa (70 Mha in 2095), Latin America (50 Mha) and North America

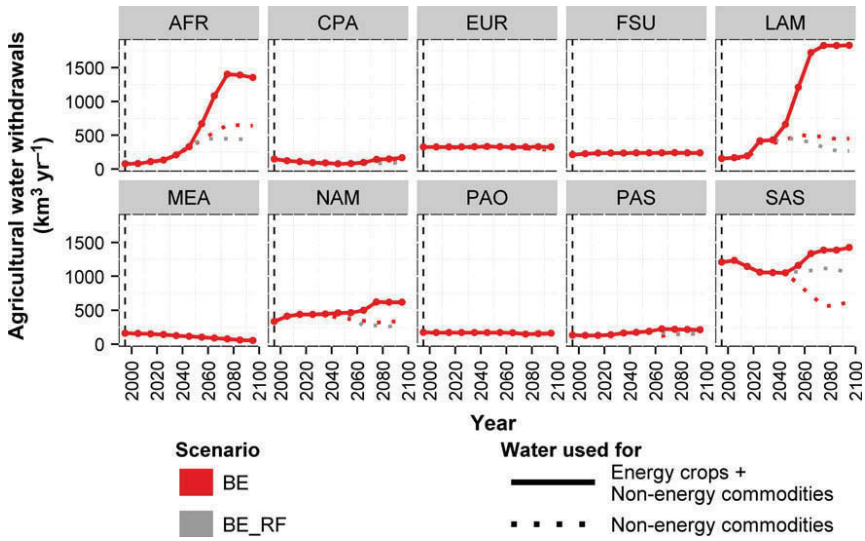

Fig. 4 Regional agricultural water withdrawals for BE and BE_RF. Total water withdrawals (Energy crops + non-energy commodities) appear as a solid line for BE. Dashed lines depict water withdrawals for nonenergy commodities (irrigated food crops, livestock production, and irrigated material crops) only. For BE_RF, irrigation of bioenergy crops is prohibited so that water withdrawals for nonenergy commodities equal total water agricultural withdrawals. The vertical dashed line marks the start of the simulation period.

(40 Mha) (Figure S5). In BE_RF, more land is abandoned between 2075 and 2095 than in BE due to stronger agricultural intensification at the end of the century.

Carbon dioxide emissions from land-use change (Figure S6) in BE_RF amount to 455 Gt CO2 between 1995 and 2095. Compared to emissions in BE of 316 Gt CO2 over the century, they are 44% higher because of increased agricultural land requirements and associated land-use change.

To test our results, we compare regional MAgPIE projections for cropland and pasture with FAO data (FAO, 2013) (Figures S7, S8). Deviations of regional MAgPIE cropland in 1995 from FAO data stay below 12% and deviations in regional pasture area are below 20%. The near term trend in the MAgPIE projections is in general similar to historical trends. The only exception is in the Middle-East and North Africa, where MAgPIE cropland and pasture are lower than FAO data by ~30%. The reason for this behavior is that MAgPIE prefers a more intensive production pathway in the Middle-East than observed in reality. Thus, early investments into technological change increase yields above real world levels and lead to reduced land requirements.

Bioenergy prices

Bioenergy supply prices as calculated by MAgPIE reflect the marginal costs of producing one additional unit of bioenergy given the current bioenergy demand.

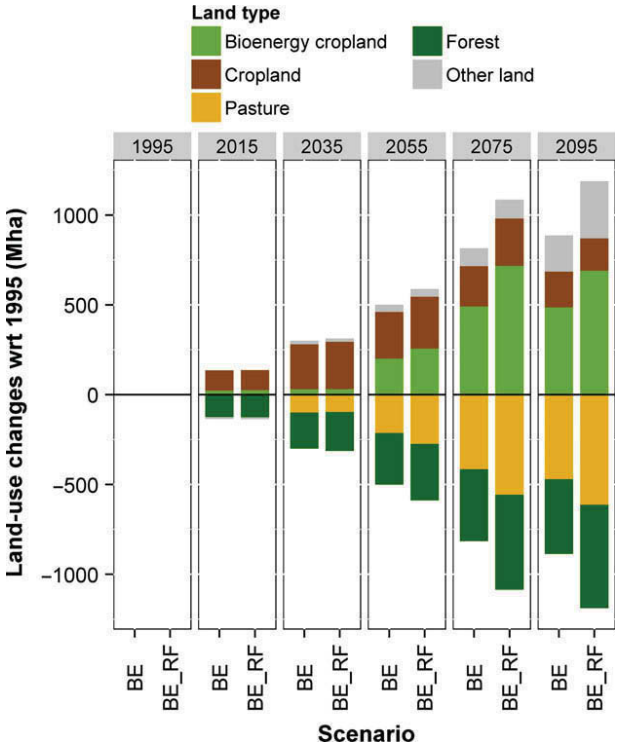

Fig. 5 Global land-use change over the century with respect to 1995 for BE and BE_RF for the land types represented in MAgPIE. Total cropland is split into bioenergy cropland and cropland for food, feed and material production (Cropland). Positive values indicate an increase, negative values a decrease in the corresponding land pool.

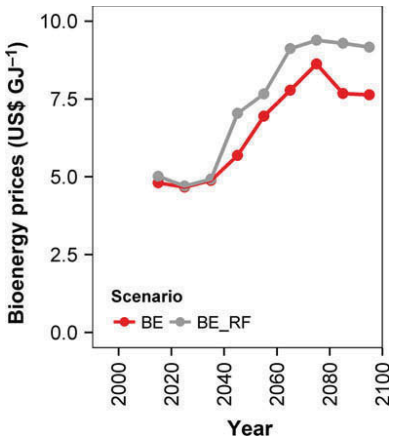

Fig. 6 Bioenergy prices for BE and BE_RF. Prior to 2015, no prices can be calculated since no bioenergy is produced.

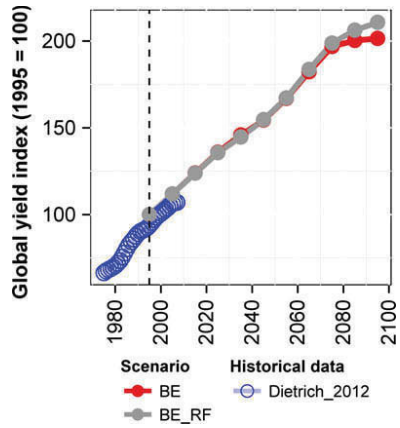

Fig. 7 Global yield index (1995 = 100) for BE and BE_RF. Changes reflect yield increases due to technological change on regional level. The global average is calculated using crop area as a weight. Historical data from Dietrich *et al.* (2012) are displayed for comparison. The vertical dashed line marks the start of the simulation period.

In the BE scenario, supply prices for primary energy from dedicated bioenergy crops increase from ~5 US $/GJ in 2015 to 8.6 US$/GJ in 2075 due to increasing bioenergy demand (Fig. 6). Afterwards, bioenergy supply prices slightly decline due to demand stagnation and reach 7.6 US$/GJ in 2095 for a production of ~300 EJ yr^{-1} of bioenergy. In BE_RF, bioenergy supply prices follow a similar trajectory and are always higher than in BE. In 2095 BE_RF exhibits a bioenergy price of 9.2 US $/GJ, around 20% higher than in BE.

Yield growth due to technological change

Yield growth for agricultural crops followed a linear trend in the past (Hafner, 2003; Fischer & Edmeades, 2010). The calculation of average annual growth rates would thus be misleading since it would suggest exponential growth. We therefore report yield growth due to technological change by calculating a global yield index (1995 = 100) (Fig. 7). Investments into yield increasing agricultural research and development are an endogenous model decision on regional level. Until 2075, increasing demand for agricultural products (Figure S4) leads to an approximately linear increase in global yield levels by ~12 points per decade. Afterwards, the yield trajectory flattens out, especially for BE, because demand stagnates and there is no further incentive for the model to invest into technological change. Regional yield increases (Table 3) are highest in the Middle East, Africa, Latin America, and South Asia. Initial land-use intensities in these regions are low and yield improvements can be achieved by closing yield gaps at low costs. In Europe and North America in contrast, initial land-use intensities are high. Further yield increases therefore require pushing the technology frontier, are expensive, and therefore less attractive. Historical data from Dietrich *et al.* (2012) shows global yield increases due to technological change by ~14 points per decade after 1960. Fischer & Edmeades (2010) find that yields for the important food crops maize, rice and wheat increased at about 8 to 16 points per decade between 1988 and 2007. Historical corn yield levels in the United States increased at ~14 points per decade between 1960 and present (Egli, 2008). Thus, our productivity pathway is compatible with historical data at the global scale. It is however unclear, whether historical yield trends can be maintained after current yield gaps have been closed (Cassman, 1999). In our projections, bioenergy yields stay within the yield potential achievable under current best management as simulated by LPJmL for most regions (Figure S9, left). In Latin America however, MAgPIE bioenergy yields at the end of the century exceed LPJmL potential yields by 12%.

Sensitivity analysis

In the standard implementation, yield gains from technological innovation increase land and irrigation water productivity simultaneously (see Section Scenarios). If technological change only increases land productivity and leaves irrigation water productivity unchanged (static WP), water withdrawals for BE and BE_RF are significantly higher than in the standard model (Table 4). The relative difference in AWW between BE and BE_RF in 2095 is however comparable for the standard model (110%) and the static WP model (100%).

Since water is less productive in static WP at the end of the century, less bioenergy area can be irrigated in the BE scenario for static WP. Therefore, static WP requires more total bioenergy area in the BE scenario than the standard model and forest losses until the end

Table 3 Regional yield index in 2095 for BE and BE_RF (1995 = 100). The global average is calculated using crop area as a weight

	WORLD	AFR	CPA	EUR	FSU	LAM	MEA	NAM	PAO	PAS	SAS
BE	201	317	167	110	109	267	409	146	113	150	239
BE_RF	211	331	167	110	108	295	409	152	111	148	219

Table 4 Results of the sensitivity analysis. The first row contains results for the standard model for the scenarios BE and BE_RF. The second row depicts results for a model version with no improvements in irrigation water productivity (static WP). The third row contains results for a model version where bioenergy crop yields increase at half the rate of conventional crops (low Yields). Per cent numbers indicate the difference between the sensitivity results and the corresponding standard model results

Model run	Agricultural water withdrawals 2095 (in km^3 yr^{-1})		Total bioenergy area in 2095 (in million ha)		Irrigated bioenergy area in 2095 (in million ha)		Forest lost between 1995 and 2095 (in million ha)	
Standard model								
BE	6393		486		228		416	
BE_RF	3031		689		0		576	
Static WP								
BE_staticWP	8879	+38%	568	+16%	142	–38%	516	+24%
BE_RF_staticWP	4456	+47%	683	−1%	0	+0%	560	−3%
Low Yields								
BE_lowYields	8306	+30%	740	+52%	350	+53%	556	+33%
BE_RF_lowYields	2446	−20%	1002	+45%	0	+0%	690	+20%

of the century are higher. Bioenergy area in the BE_RF scenario is not affected by different irrigation water productivity assumptions.

If yield increases from technological change are only half as efficient for bioenergy crops as for traditional crops (low Yields), bioenergy land requirements increase substantially and reach up to 1002 Mha for BE_RF. With lower yields, irrigation becomes even more attractive and AWW in BE increase by 30% compared to the standard model. In BE_RF, AWW are reduced because rainfed bioenergy cropland replaces irrigated food cropland. Bioenergy yields are lower in the low Yields model than in the standard model and stay within yield potentials as simulated by LPJmL (Figure S9, right).

To test the sensitivity of our results with respect to bioenergy demand, we conduct a sensitivity analysis where we reduce bioenergy demand in 15% steps from the original demand of 307 EJ yr^{-1} in 2095 (Figure S10). While for both scenarios, bioenergy area decreases with decreasing demand, bioenergy area in BE_RF is higher than in BE over the range of demand scenarios considered. For a demand reduction of 30%, BE_RF requires the same amount of bioenergy area as BE with the full demand of 307 EJ yr^{-1}.

Discussion

Several studies have highlighted that large-scale irrigated bioenergy production may require significant amounts of water and fundamentally alter the state of global freshwater resources (Berndes, 2002; Beringer *et al.*, 2011; Chaturvedi *et al.*, 2013). The present study investigates the trade-offs between water and land resources for producing ~300 EJ of bioenergy per year in 2095 from dedicated energy crops using a spatially explicit global land and water-use allocation model. We compare a scenario where irrigation water use for bioenergy production is allowed within the limits of available water (BE) to a rainfed-only bioenergy production scenario that aims at minimizing the impacts of bioenergy production on water resources (BE_RF). This experimental setup allows us to determine the water-use implications of large-scale bioenergy production as well as the land-use implications of water-saving bioenergy production strategies.

Implications of bioenergy production for water resources

Our results suggest that irrigation will play a key role in providing bioenergy feedstock if no policy restrictions are imposed. In contrast to comparable studies (Berndes, 2002; Chaturvedi *et al.*, 2013), the decision between rainfed and irrigated bioenergy production is treated here as an endogenous process. Thus, the large irrigated fraction in total bioenergy production (58%) in the BE scenario reflects comparative advantages, especially significant yield improvements through irrigation in important bioenergy production regions such

as India, Latin America, North America, and Africa. This is in line with results by Beringer *et al.* (2011) who found that natural water limitations reduce bioenergy yields in large parts of these regions by up to 100% of the yield achievable without water limitations.

In this study, irrigation water requirements associated with the production of 300 EJ/yr of second generation bioenergy crops reach 3350 km^3 if no policy restrictions on irrigated bioenergy production are imposed. This number is comparable to current global agricultural water withdrawals (Shiklomanov, 2000; Wisser *et al.*, 2008) and is consistent with previous studies on bioenergy water withdrawals. Beringer *et al.* (2011) estimates irrigation water requirements of large-scale bioenergy production of 1500–3900 km^3. Chaturvedi *et al.* (2013) explore the water-use implications of several climate change mitigation scenarios. They project water withdrawals for bioenergy production between 670 and 5200 km^3 depending on the bioenergy deployment (<100–850 EJ yr^{-1}). Given the substantial associated water withdrawals, the question whether irrigated bioenergy production will impair freshwater ecosystems needs to be addressed. Experience from past and present human influence on water resources suggests that even current levels of human water withdrawals pose a major threat to aquatic ecosystems. It has been estimated that freshwater vertebrate populations have declined by 54% globally and that 32% of the world's amphibian species are threatened with extinction due to human interference (Dudgeon *et al.*, 2006). Hoekstra *et al.* (2012) have estimated that human water use exceeds the sustainably allowed level at least 1 month per year in 223 of 405 large river basins globally. A recent special issue has highlighted that direct human influence will pose the biggest threat to freshwater ecosystems in the coming decades (Vörösmarty *et al.*, 2013). Since agriculture contributes around 70% to current human water withdrawals (Rost *et al.*, 2008), these studies suggest that the projected doubling of agricultural water withdrawals due to large-scale bioenergy production will have substantial adverse impacts on freshwater ecosystems. Especially projected irrigated bioenergy production in South Asia (40 EJ yr^{-1}), a region facing severe water scarcity and overexploitation of groundwater resources (de Fraiture *et al.*, 2008; Biewald *et al.*, 2014) is worrisome in this context.

Implications of a water-saving bioenergy production strategy

Can ambitious bioenergy targets be reached without threatening global water resources? In our experimental setup, it is possible to produce 300 EJ yr^{-1} of bioenergy – an amount comparable to current total human appropriation of primary biomass production (Haberl *et al.*, 2007) – without tapping additional blue water resources for irrigation. There are, however, caveats associated with such a rainfed-only bioenergy production scenario.

Land requirements for producing ~300 EJ yr^{-1} of bioenergy increase significantly if irrigated bioenergy production is prohibited since rainfed bioenergy yields are lower than irrigated yields. In our simulations, an additional 200 Mha of bioenergy cropland will be required if irrigated bioenergy production is prohibited. This corresponds to the extent of current total cropland in the United States and Australia together (~210 Mha) (FAO, 2013). A recent model intercomparison exercise projects land requirements for the production of 150–250 EJ yr^{-1} of bioenergy feedstock with three integrated assessment models (Popp *et al.*, 2014; Rose *et al.*, 2014). The projected bioenergy cropland from this study is 450 to 550 Mha, comparable to our bioenergy cropland projections for producing 300 EJ yr^{-1} of 490 Mha (BE) to 690 Mha (BE_RF).

Prohibiting irrigated bioenergy production leads to an increase in bioenergy supply prices. In a market economy, such price increases would lead to a decreased demand for bioenergy and could reduce the additional land requirements in a rainfed-only bioenergy production scenario. The feedback of increased bioenergy prices on bioenergy demand depends crucially on the willingness-to-pay for bioenergy in the energy system. Rose *et al.* (2014) suggest that bioenergy is an economically attractive energy carrier, especially in combination with carbon capture and storage under climate change mitigation scenarios. Klein *et al.* (2014) find a high willingness-to-pay for bioenergy in case of stringent climate targets. Our sensitivity analysis demonstrates that bioenergy demand would need to decrease by 30% in BE_RF compared to BE to avoid bioenergy area expansion compared to BE.

Additional land requirements for bioenergy production in the rainfed-only case are partly fulfilled at the expense of pasture areas. While bioenergy expansion into pasture areas can lead to the loss of important ecosystems featuring high biodiversity (Alkemade *et al.*, 2013) and carbon storage potential (Conant *et al.*, 2001), the impact on natural forests is even more worrisome. Our results suggest that a rainfed-only bioenergy scenario would lead to substantially increased losses of natural forests (580 Mha) compared to the unrestricted bioenergy scenario (420 Mha), especially in tropical regions where additional forest losses in BE_RF compared to BE amount to 120 Mha by 2095. Tropical rainforests are high priority conservation targets since they are major biodiversity hotspots (Barlow *et al.*, 2007) and provide a number of important ecosystem services such as carbon sequestration and water flow regulation

(Onaindia *et al.*, 2013). It is thus likely that protecting freshwater ecosystems from degradation due to bioenergy production will accelerate the loss of important land ecosystems if no strict land-use change regulations are implemented.

Aside from the trade-off between water and land resources for bioenergy production, economic considerations may form an obstacle to the implementation of water-saving bioenergy production policies. With rising energy prices, bioenergy production may become an important source of income for farmers (Walsh *et al.*, 2003), and can play a key role for economic development in developing countries (Demirbas & Demirbas, 2007). We find that restricting irrigation changes the comparative advantages between regions and leads to reallocation of production and associated economic benefits.

Assumptions and limitations

This study investigates implications of different bioenergy production strategies on land and water resources in a cost optimization framework with a fixed bioenergy target of ~300 EJ yr^{-1} in 2095. While this setup allows us to investigate the cost optimal resource allocation as well as bioenergy supply prices for a fixed bioenergy demand under different production scenarios, we are not able to quantify price-induced changes in bioenergy demand between the scenarios. Thus, this study provides insights into the implications of substituting water resources with land resources for large-scale bioenergy production, but does not claim to provide a comprehensive picture of future bioenergy related resource requirements under different scenarios.

The influence of bioenergy production on water resources is not restricted to irrigation water requirements. First, water is also needed during the conversion of biomass into final energy (e.g., electricity, fuel, heat, Singh *et al.*, 2011). Processing requirements are however small compared to water requirements during feedstock production (Berndes, 2002; Gerbens-Leenes *et al.*, 2009; Gheewala *et al.*, 2011) and have therefore been neglected in this analysis. Second, bioenergy plantations will to some extent alter the balance between runoff and evapotranspiration, thereby changing available blue water in rivers, lakes and aquifers (Berndes, 2002). The magnitude of this effect is however highly uncertain (Haddeland *et al.*, 2011) and depends on the location and the type of vegetation that is replaced by bioenergy crops (Berndes, 2002). Quantifying the overall effect of bioenergy on water availability would therefore require a full coupling to a global vegetation and hydrology model such as LPJmL, which is beyond the scope of this analysis. Third, we focus on water quantity and do not investigate bioenergy implications for water quality.

Land requirements for bioenergy production crucially depend on bioenergy yields (Creutzig *et al.*, 2014). Observed bioenergy yields on test sites in Europe range between 120 and 280 GJ ha^{-1} yr^{-1} (Chum *et al.*, 2011). Average European bioenergy yields in 1995 in our model of 115 GJ ha^{-1} yr^{-1} (rainfed) and 220 GJ ha^{-1} yr^{-1} (irrigated) are consistent with these observations. It is however unclear, if the yields that were achieved under test conditions can be realized over large areas (Johnston *et al.*, 2009). Due to investments into agricultural research and development, all agricultural yields – including bioenergy yields – in our projections approximately double until the end of the century on global average. This yield projection is consistent with historical data on yield increases for conventional crops. It is however unclear, whether plant physiological limits will limit future yield increases (Cassman, 1999). Moreover, almost half of the past yield increases can be attributed to harvest index improvements (Krausmann *et al.*, 2013). In the case of lignocellulosic bioenergy crops, all aboveground biomass can be used for energy production, so that increasing the harvest index is hardly possible (Searle & Malins, 2014). On the other hand, breeding of lignocellulosic bioenergy crops has just started, fostering the hope that significant yield improvements are possible (Chum *et al.*, 2011). Several studies argue that natural productivity poses an upper limit to bioenergy yields (Erb *et al.*, 2012; Smith *et al.*, 2012; Haberl *et al.*, 2013). Within our model, bioenergy yields in 2095 stay within the potential yield achievable under current best management as simulated by LPJmL, except for Latin America. In summary, rainfed bioenergy yields within our model of up to 450 GJ ha^{-1} yr^{-1} in 2095 are within the range reported by Haberl *et al.*(2010) (69–600 GJ ha^{-1} yr^{-1} in 2055), but bioenergy yields remain a key uncertainty of our analysis.

Restrictive land-use change policies that mainly aim at conserving natural forests (REDD) are discussed as an option to mitigate climate change (Angelsen *et al.*, 2009). Our scenarios do not contain a REDD policy and therefore allow conversion of forests and other natural vegetation into bioenergy plantations. Under scenarios with a REDD policy, such expansion would be strictly limited and could lead to stronger land productivity increases, reduced land-use implications of bioenergy but potentially higher bioenergy prices.

Aside from land productivity, water productivity is a key determinant of the resource requirements for large-scale bioenergy production (King *et al.*, 2013). In our standard model implementation, agricultural research and development is assumed to increase both, land and irrigation water productivity. This assumption is supported by various studies on crop water productivity (Kijne *et al.*, 2004; Rosegrant *et al.*, 2009; Molden

et al., 2010) and can be achieved by: minimizing losses in the water distribution system; increasing the ratio of transpiration to evaporation on the field; increasing plant water-use efficiency by breeding and improved management of all inputs. The extent of possible irrigation water productivity improvements is however uncertain, especially in already highly intensified agricultural systems. Our sensitivity analysis shows that agricultural water requirements are significantly higher if no improvements in irrigation water productivity can be realized. The competitiveness of irrigated bioenergy production and the possible doubling of agricultural water withdrawals due to bioenergy production is, however, robust with respect to different assumptions on water productivity.

Policy implications and conclusions

In the context of the presented results, it is important that policies to protect freshwater ecosystems from degradation due to bioenergy production are carefully designed and address the trade-off with land ecosystems and the economic incentives opposing sustainable water use. Certification schemes are one possibility to manage the water implications of bioenergy production (Fehrenbach, 2011). A certificate for rainfed bioenergy production would allow consumers to make an informed choice and could create a market incentive for less water intensive production. Governments could furthermore create direct incentives for rainfed bioenergy production through taxes and subsidies. South Africa has for example already decided to stop the support for bioenergy crops under irrigation (Moraes *et al.*, 2011).

Policies that aim at incentivizing rainfed bioenergy production are useful to protect water resources but neglect the trade-off with land resources and may therefore endanger land ecosystems. Rather than promoting rainfed-only bioenergy production, one could therefore restrict water use for bioenergy production to sustainable levels. Such an approach would require site specific estimates of how much water is required for a functioning ecosystem. Estimates of how much water needs to be reserved for environmental purposes – also called environmental flows – are already available (Smakhtin *et al.*, 2004; Poff *et al.*, 2010), although more research is needed to increase the accuracy of such estimates (Pastor *et al.*, 2013). The implementation of comprehensive water management strategies that take into account the different types of human water use and environmental flow requirements (Pahl-Wostl *et al.*, 2013) would allow irrigation of bioenergy where enough water resources are available. Thus, negative impacts of bioenergy production on water resources could be prevented while irrigation could still contribute to reducing land requirements for bioenergy crops. Ideally, such sustainable water management policies would be accompanied by forest protection policies that can further reduce the negative impacts of bioenergy production on natural land ecosystems (Popp *et al.*, 2011b).

Producing 300 EJ yr^{-1} of bioenergy from dedicated bioenergy crops is a very ambitious scenario (Creutzig *et al.*, 2014). Lower second generation bioenergy production will of course have less implications for land and water resources and may raise less sustainability concerns. Even with a lower contribution from dedicated crops, bioenergy can make an important contribution to the future energy mix since forestry and residues can provide 35–125 EJ yr^{-1} in 2050 already (Creutzig *et al.*, 2014). It is furthermore likely that market forces lead to a decreased bioenergy demand because of higher prices if irrigation of bioenergy crops is prohibited. This process could mitigate the negative impacts of water-saving bioenergy production strategies on land resources. Our results however suggest that a price-induced demand reduction of 30% would be necessary to fully compensate additional land requirements if irrigated bioenergy production is prohibited.

In summary, our results indicate that without dedicated water protection policies, large-scale bioenergy production from dedicated 2nd generation energy crops may lead to severe degradation of freshwater ecosystems. It is therefore crucial that the focus of bioenergy strategies shifts from land-use efficiency (Gheewala *et al.*, 2011) to a broader sustainability perspective including water resources. We find that prohibiting irrigated bioenergy crop production for water resources protection can lead to the loss of important natural land ecosystems, especially tropical forests. Policies that balance water- and land-use implications of large-scale bioenergy production are therefore needed. The concept of environmental flow protection is a promising avenue since it protects freshwater ecosystems while still allowing for irrigated bioenergy production to increase yields and thereby decrease the pressure on land ecosystems. Further research should aim at investigating additional implications of water-saving bioenergy production strategies that were not covered here. Those include feedbacks on bioenergy deployment in the energy system, as well as implications for the water cycle due to changes in evapotranspiration on bioenergy plantations.

Acknowledgements

We thank the anonymous reviewers for their valuable comments. The research leading to these results has received funding from the European Union's Seventh Framework Program FP7/2011 under grant agreements no 308329

(ADVANCE), no 282846 (LIMITS) and no 603542 (LUC4C). Additional funding was provided by the BMBF through the INNOVATE project (grant agreement 01LL0904D).

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Data S1. Additional model description and results.

Trade-offs between land and water requirements for large-scale bioenergy production

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Supporting information

Contents

Additional model and scenario description

All simulations for this analysis have been carried out with the MAgPIE model revision 7423.

MAgPIE world regions

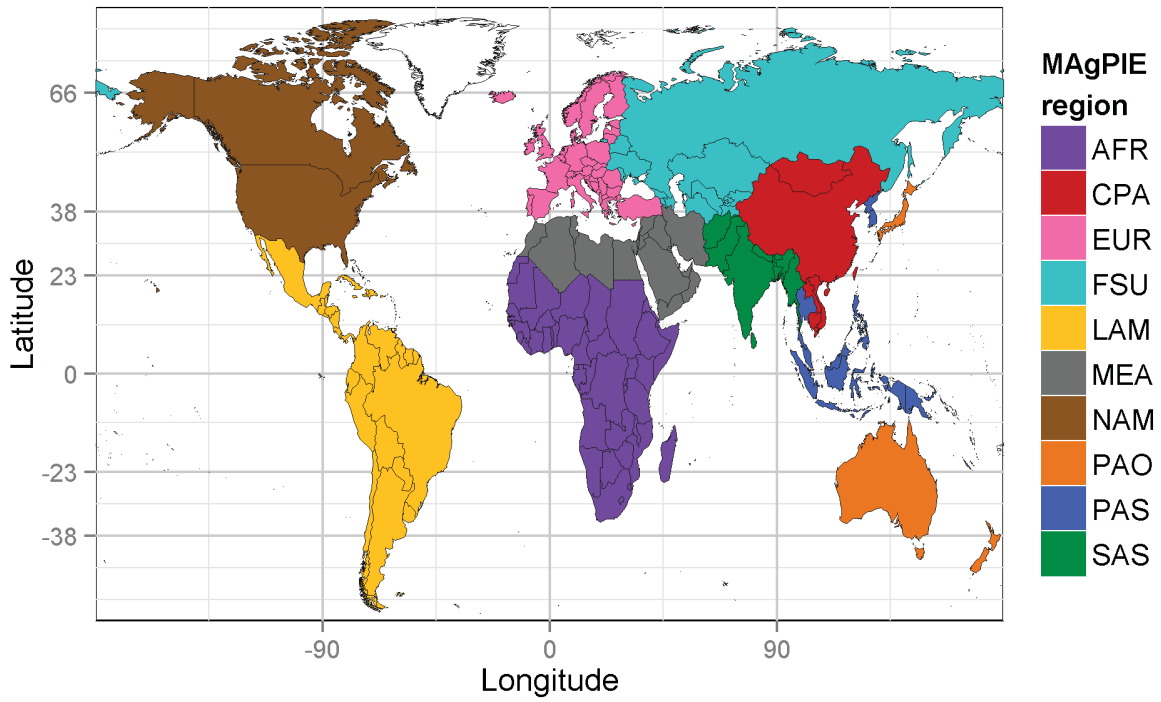

Figure S1 MAgPIE world regions. AFR=Sub Saharan Africa, CPA= centrally planned Asia including China, EUR=Europe, FSU=former Soviet Union, LAM=Latin America, MEA=Middle East and North Africa, NAM=North America, PAO=Pacific OECD, PAS=Pacific Asia, SAS=South Asia. Greenland and Antarctica are not covered by MAgPIE.

Bioenergy yields

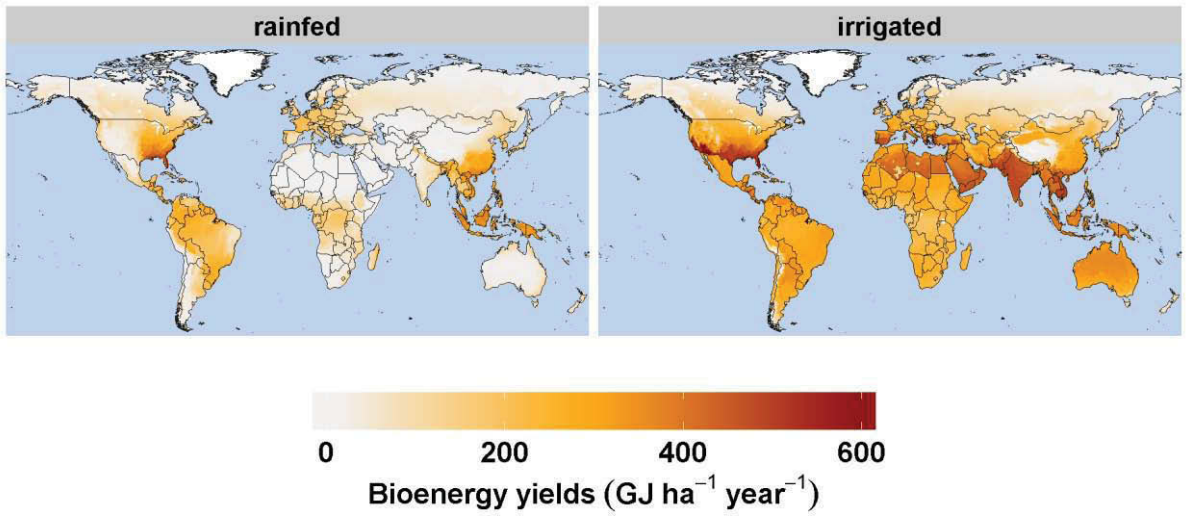

Figure S2 Yields of herbaceous lignocellulosic bioenergy crops in 1995. Rainfed (left) and irrigated (right). Bioenergy yields are derived from LPJmL potential yields (Beringer et al., 2011) and calibrated using information about observed land-use intensity (Dietrich et al., 2012) and agricultural yields (FAO, 2013) to arrive at actual yields. Note that actual yields are displayed for all land area except for Greenland and Antarctica, irrespective whether bioenergy is grown in a given location.

	WORLD	AFR	CPA	EUR	FSU	LAM	MEA	NAM	PAO	PAS	SAS
Rainfed (MAgPIE)	74	51	102	115	25	152	9	80	40	293	98
Irrigated (MAgPIE)	203	249	215	220	58	289	372	141	311	352	394
Calibration factor	-	0.26	0.58	0.71	0.37	0.38	0.41	0.72	0.34	0.47	0.46

Table S1 Global and regional average actual herbaceous bioenergy yields (GJ/ha/yr) in 1995. Top row: rainfed yields; middle row: irrigated yields. The average is obtained by calculating the non-weighted mean yield over all simulation units in a specific region, irrespective of whether bioenergy is actually grown. Bioenergy yields are derived from LPJmL potential yields (Beringer et al., 2011) and calibrated using information about observed land-use intensity (Dietrich et al., 2012) and agricultural area (FAO, 2013) to arrive at actual yields. Regional calibration factors are shown in the bottom row.

Non-agricultural water withdrawals

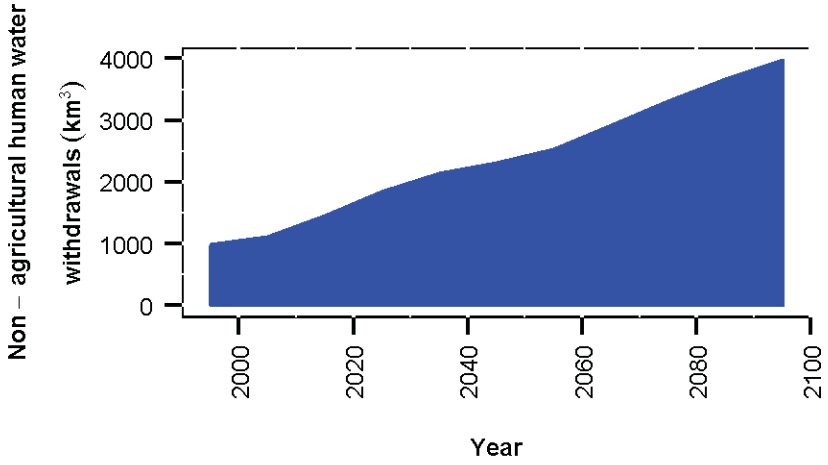

Figure S3 Global non-agricultural human water withdrawals including industrial production, domestic use and electricity production for the SSP2 scenario. Data is obtained from WaterGAP (Flörke et al., 2013), (Alcamo et al., 2003).

Demand

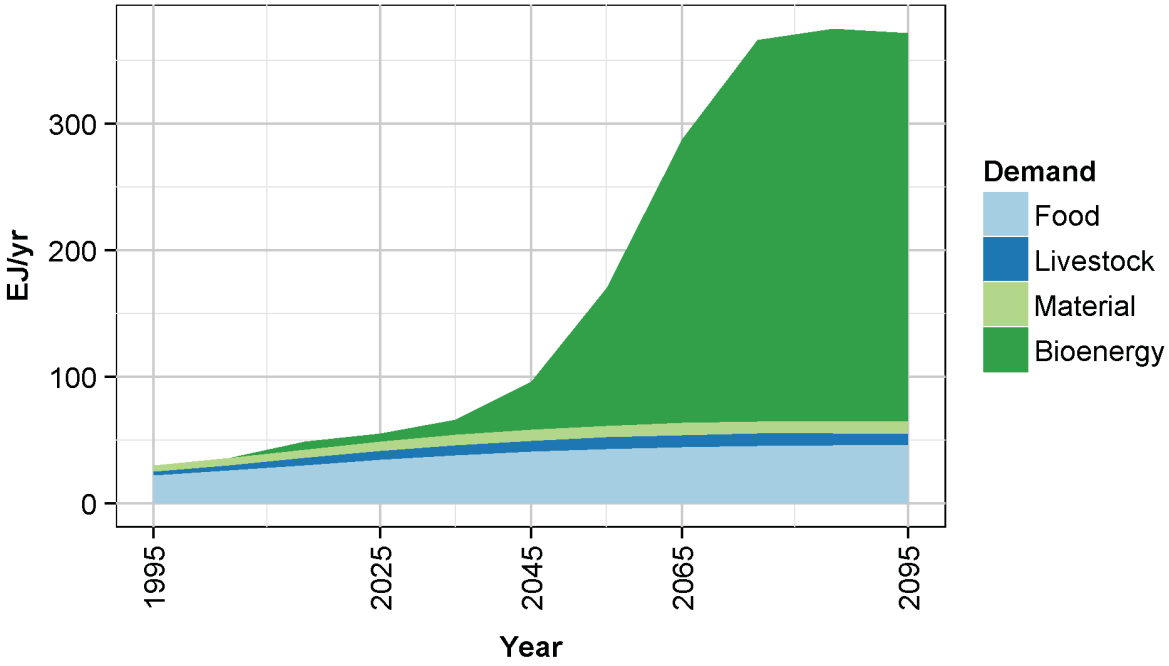

Figure S4 Time-series of global food, livestock, material and bioenergy demand in EJ per year. Food, livestock and material demand is calculated according to the methodology in Bodirsky et al. (under review) and Bodirsky et al. (2012) for the SSP2 scenario. Bioenergy demand is taken from Popp et al. (2011).

Available water calculation

Available water is based on simulations by the global hydrology, vegetation and crop growth model LPJmL (Rost et al., 2008; Müller and Robertson, 2014). LPJmL takes climate and precipitation data from the CRU 3.0 dataset (Mitchell and Jones, 2005) as input and simulates global runoff (water that enters rivers, lakes and aquifers) and river discharge on a 0.5 degree grid. Within MAgPIE, monthly runoff (also called "blue" water Rost et al., 2008), i.e. the renewable freshwater resource excluding fossil groundwater and discharge from melting glaciers constitutes available water in one month. Distribution of available water to individual grid cells follows an approach by Schewe et al. (2013) and uses river discharge as a weight. First, monthly available water is aggregated to river basin level (basins according to Döll and Lehner, 2002). Each grid cell in a river basin then receives a fraction of available water at basin level that corresponds to the ratio of the cell's monthly discharge over the total monthly discharge within the basin. It has been highlighted that a considerable fraction of blue water is inaccessible to humans due to seasonal distribution (Postel et al., 1996). It is common to assume that a fixed fraction of blue water is inaccessible and this fraction varies considerably among studies (Gerten et al., 2013). The focus of this analysis is on irrigation, thus we restrict the available water for irrigation to the growing period of the plants (based on LPJmL simulations). The annual water availability in each grid cell is thus the sum of the monthly values during the growing period except for cells where dams are present (Biemans et al., 2011) where the blue water resource of the whole year is available. We impose no further restrictions of available water such as ecosystem protection (Smakhtin et al., 2004). In total, 27000km^3 per year of water are available for human use globally.

Irrigation water requirement calculation

Irrigation water requirements for food, material and bioenergy crops is simulated by LPJmL based on the soil water deficit below optimal growth of the plants (Rost et al., 2008) on a 0.5 degree grid. These net irrigation requirements on the field are corrected for losses during the transport of the water from the source to the field. Based on a study by Rohwer et al. (2007), these conveyance losses were estimated at 36% of water withdrawn from the source.

Irrigation cost calculation

Irrigation costs comprise investment costs for deploying new irrigation infrastructure and annual costs for operating irrigation systems. Investment costs for irrigation for 1995 are based on Worldbank data (Jones, 1995) and enter the model on a regional level ranging from 1900 to

37200 US$ / ha. Jones (1995) finds that differences in costs between regions are largely driven by implementation problems (construction quality, procurement problems, and funding shortage). We assume that regional differences in institutional capacity will be reduced in the future. Therefore, costs linearly converge to the European level of 5700 US$ / ha until 2050 (Table S2).

	1995	2005	2015	2025	2035	2045	2055	2065	2075	2085	2095
AFR	37173	31444	25715	19986	14257	8528	5663	5663	5663	5663	5663
CPA	8781	8214	7647	7080	6514	5947	5663	5663	5663	5663	5663
EUR	5663	5663	5663	5663	5663	5663	5663	5663	5663	5663	5663
FSU	5663	5663	5663	5663	5663	5663	5663	5663	5663	5663	5663
LAM	12237	11042	9846	8651	7456	6261	5663	5663	5663	5663	5663
MEA	5933	5884	5835	5786	5737	5688	5663	5663	5663	5663	5663
NAM	5663	5663	5663	5663	5663	5663	5663	5663	5663	5663	5663
PAO	5663	5663	5663	5663	5663	5663	5663	5663	5663	5663	5663
PAS	2078	2730	3382	4033	4685	5337	5663	5663	5663	5663	5663
SAS	1899	2584	3268	3952	4637	5321	5663	5663	5663	5663	5663

Table S2 Investment costs for installing new irrigation infrastructure (US$ / ha) for the MAgPIE regions.

Since no global dataset for annual costs for the operation and maintenance of irrigation systems is available, we use GTAP data (Narayanan et al., 2008) to calculate annual operation and maintenance costs. Calzadilla et al. (2011) have proposed an approach to split the GTAP land rent into the rent associated with irrigable land and the rent associated with irrigation water application. For this analysis, we use the rent associated with irrigation water application – calculated according to the approach by Calzadilla et al. – as a proxy for the operation and maintenance costs of irrigation infrastructure (Table S3). Local case studies in Africa and the US find annual irrigation costs of 10 – 404 US$ / ha and 167 – 392 US$ / ha respectively (Palanisami 1997; Schaible and Aillery 2013). 90% of the values resulting from our calculation are in the range reported by the local studies of 10 – 400 US$ / ha. High values of over 1000 US$ / ha only occur for potato in NAM and PAO.

	AFR	CPA	EUR	FSU	LAM	MEA	NAM	PAO	PAS	SAS
temperate cereals	31	109	96	34	48	39	51	70	97	56
tropical cereals	10	71	67	11	30	22	52	52	30	15
maize	10	55	46	19	17	52	53	78	20	15
rice	26	118	117	43	48	123	88	195	57	57
others	83	441	295	187	178	273	482	965	138	203
potato	332	695	805	389	396	764	1098	2011	609	696
cassava	46	162	94	72	72	397	92	230	67	129
pulses	59	271	280	177	81	125	220	248	153	104
soybean	21	80	105	20	77	76	71	105	36	40
rapeseed	222	400	597	92	304	325	246	439	728	229
groundnut	36	188	119	87	70	193	120	175	66	66
sunflower	105	398	243	141	286	140	199	229	196	125
oil palm	75	381	193	193	266	193	193	193	278	193
sugar beet	77	65	111	37	117	80	84	212	77	56
sugar cane	218	401	321	319	287	883	301	765	318	361
cotton	60	280	256	145	111	284	127	423	200	103

Table S3 Annual costs for operation and maintenance of irrigation systems (US$ / ha). Costs differ across regions and crops.

Additional results

Regional land-use change

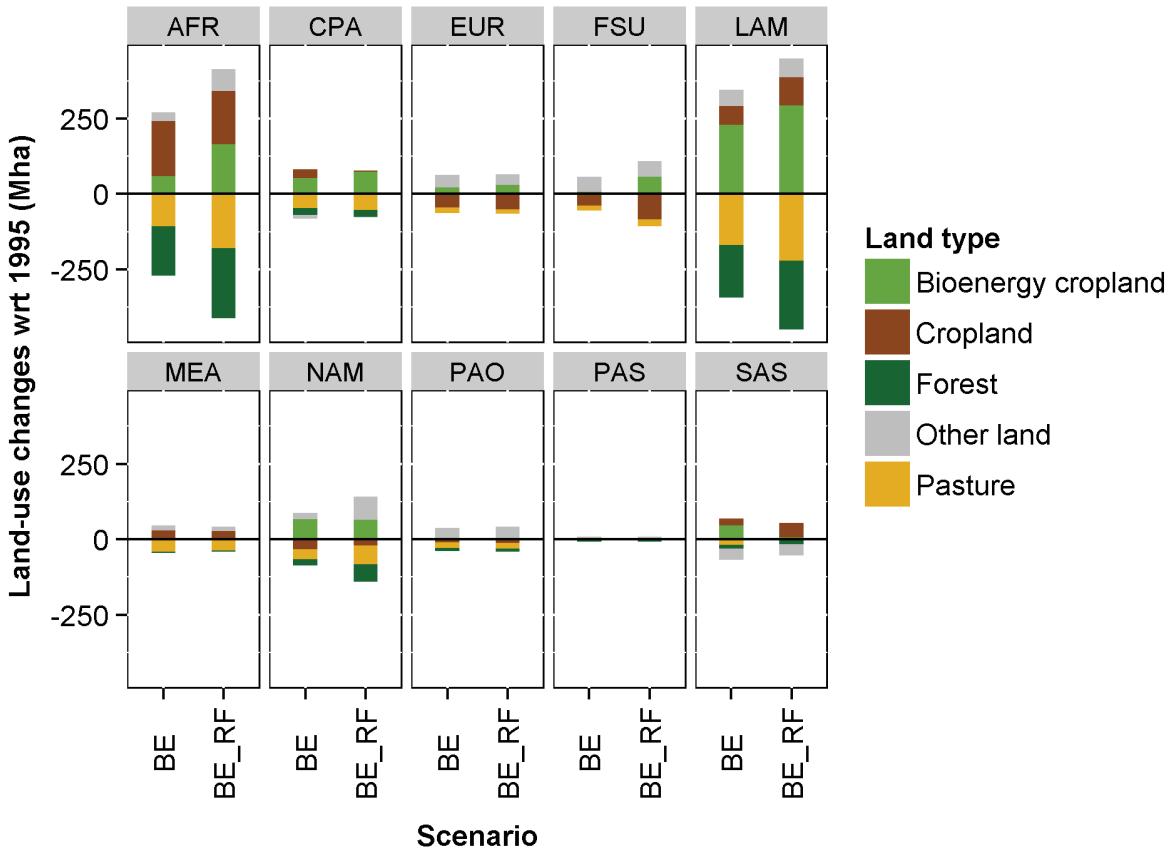

Figure S5 Regional land-use changes by the year 2095 with respect to 1995 for BE and BE_RF for the land types represented in MAgPIE. Total cropland is split into bioenergy cropland and cropland for food, feed and material production (Cropland). Positive values indicate an increase, negative values a decrease in the corresponding land pool.

Carbon emissions from land-use change

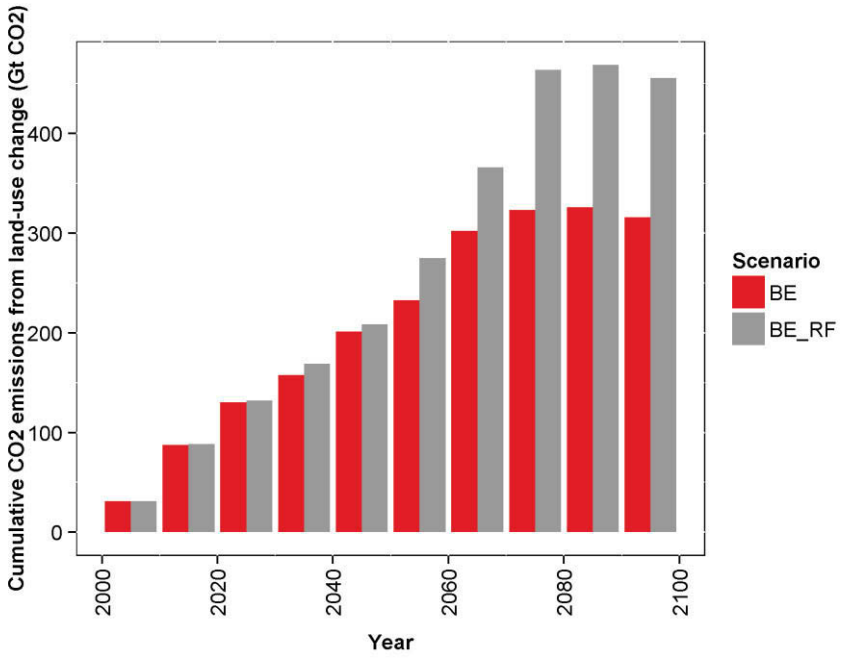

Figure S6 Global cumulative carbon dioxide emissions from land-use change for BE and BE_RF. The reference year is 1995.

Cropland validation

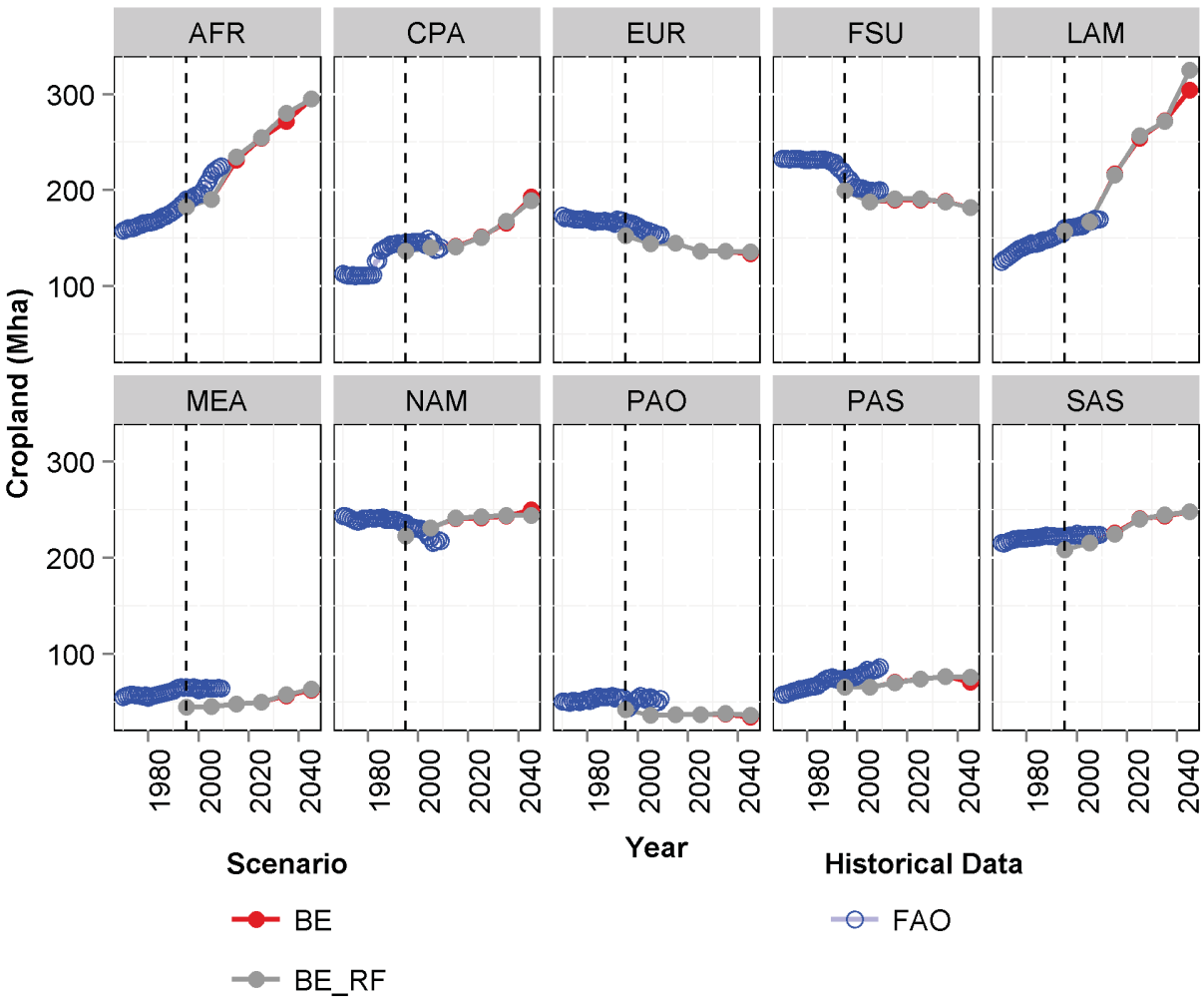

Figure S7 Regional cropland for BE and BE_RF. Historical data from FAO (FAO, 2013) is displayed for comparison. The vertical dashed line marks the start of the simulation period.

Pasture validation

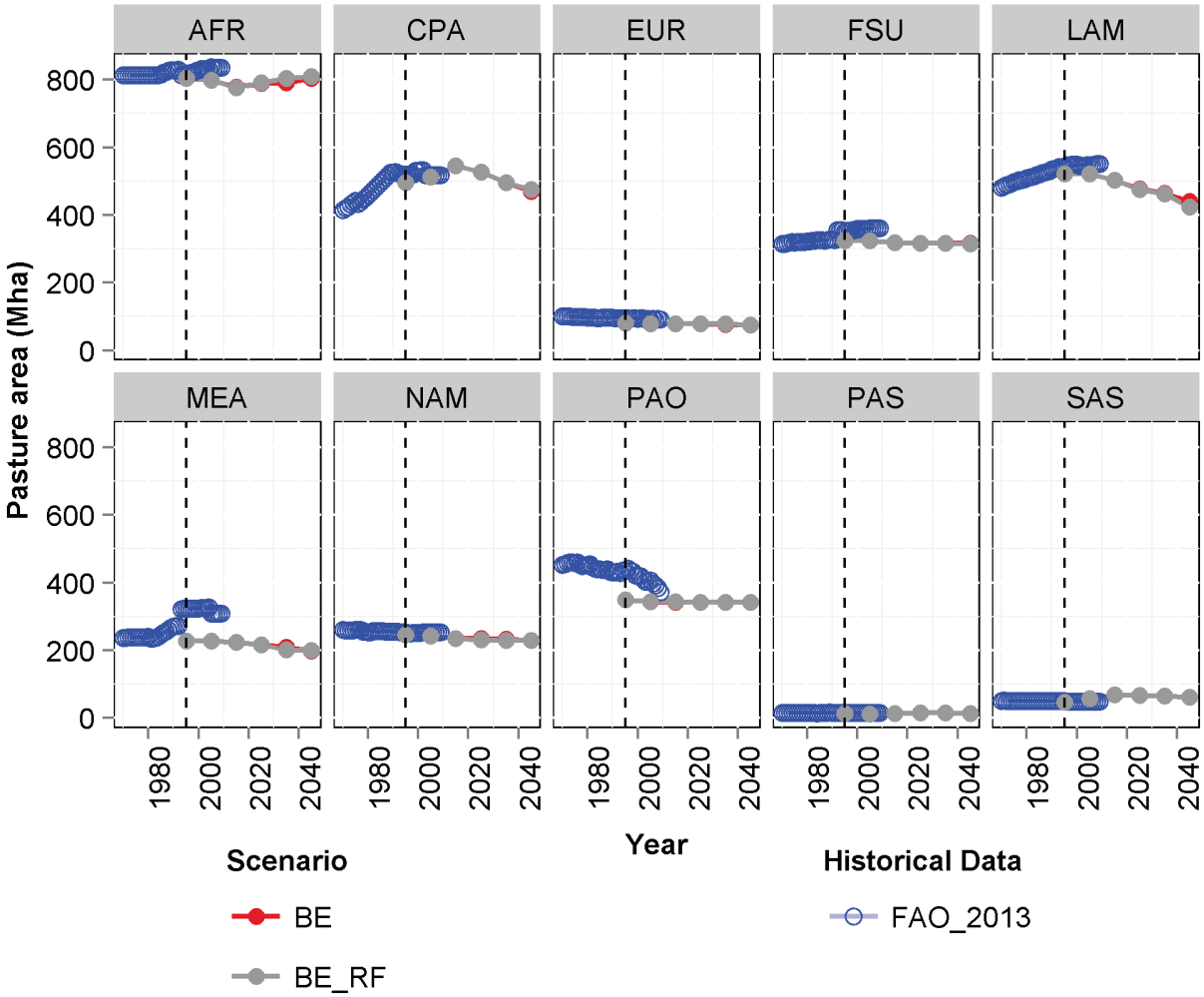

Figure S8 Regional pasture area for BE and BE_RF. Historical data from FAO (FAO, 2013) is displayed for comparison. The vertical dashed line marks the start of the simulation period.

Bioenergy yield development

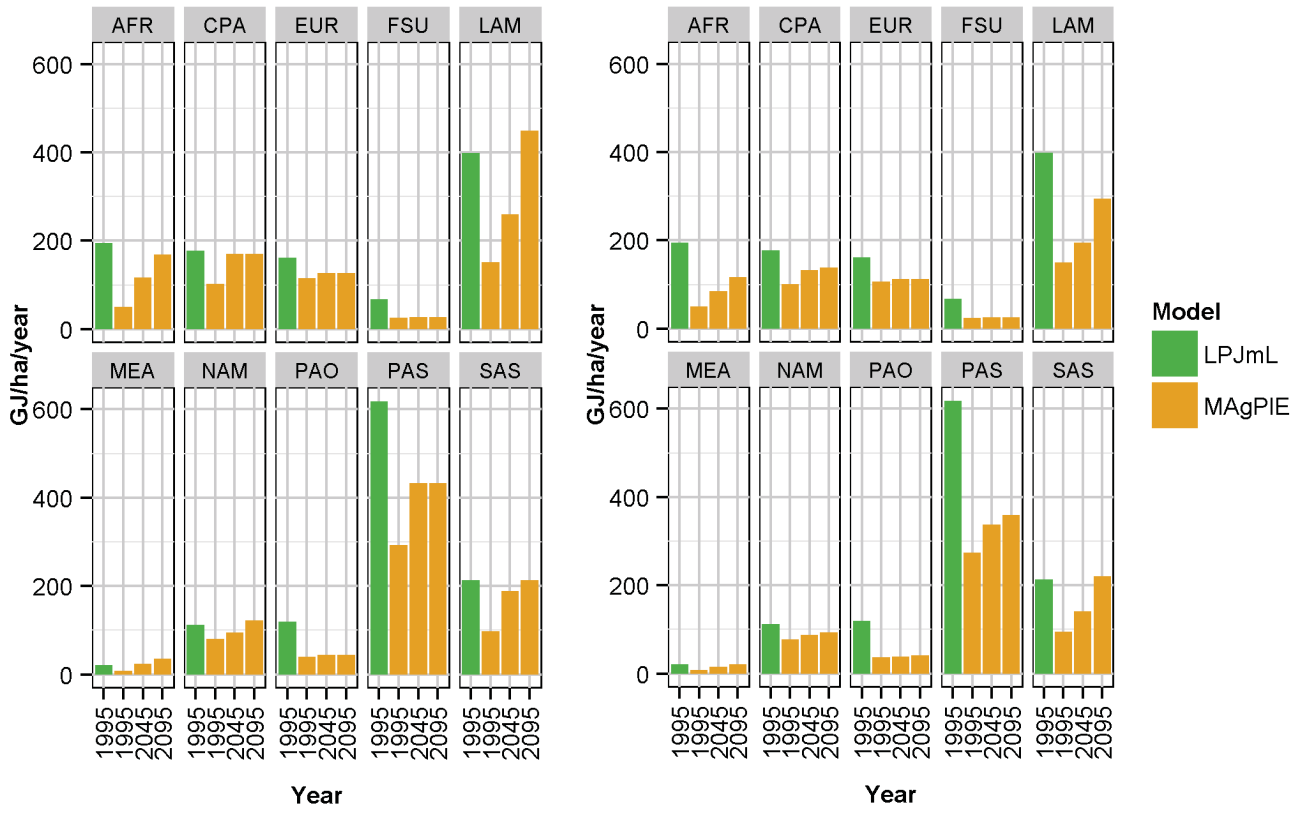

Figure S9 Herbaceous rainfed bioenergy yields for BE_RF over time for the 10 MAgPIE world regions. Standard model results (left) and results from the low Yields sensitivity model (right). Regional averages reflect the unweighted mean across all simulation units within a region. MAgPIE yields (yellow) increase over time due to technological change. LPJmL simulates potential yields under best currently available management options.

Bioenergy demand sensitivity

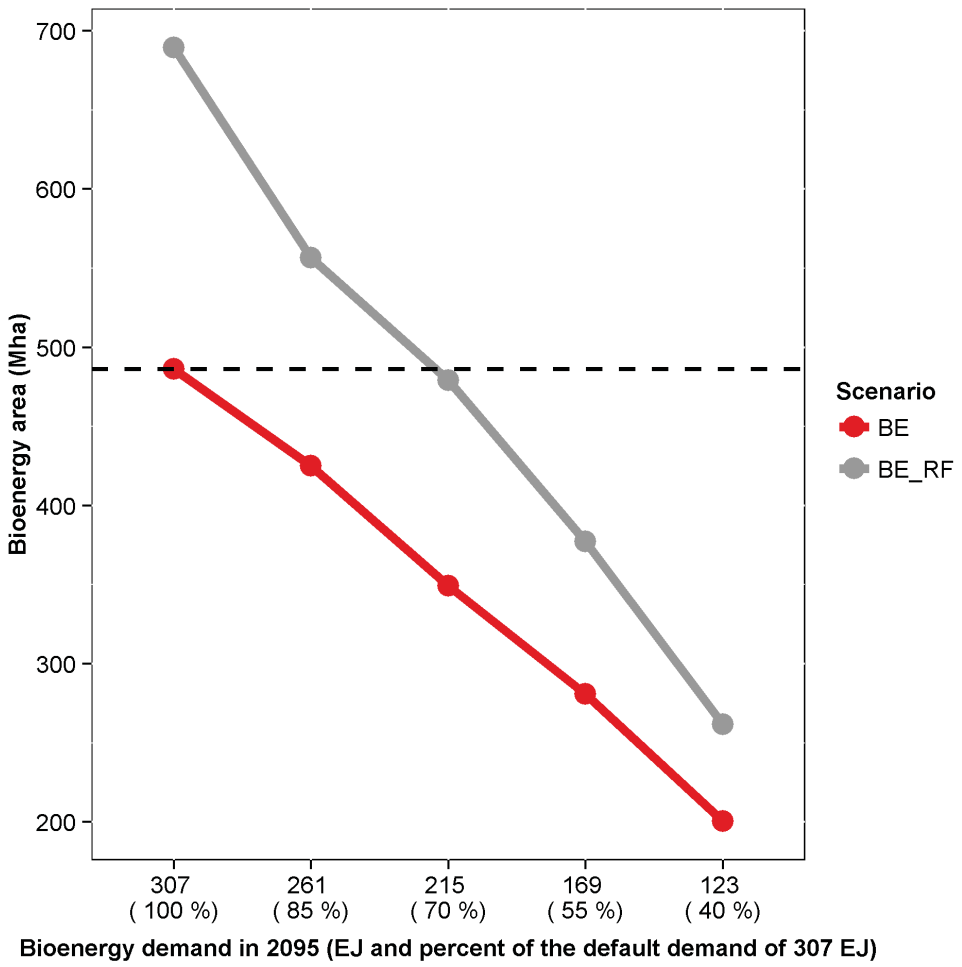

Figure S10 Global total bioenergy area in 2095 for BE and BE_RF under different bioenergy demand scenarios. Bioenergy demand scenarios are constructed by reducing the original demand in 15% steps from 100% (307 EJ/yr) to 40% (123 EJ/yr). The horizontal dashed line marks the bioenergy area for a demand of 307 EJ/yr in the BE scenario. In BE_RF, bioenergy area at a demand corresponding to 70% of the default value (307 EJ/yr) is equal to bioenergy area in BE with the full demand.

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Chapter VI

Summary, synthesis and outlook

Markus Bonsch

Content of Chapter VI: Summary, synthesis, and outlook

The main objective of this thesis is to investigate the overarching research question of how agriculture will affect global land and water resources in the future. In order to structure the presented work, the overarching research question has been divided into four specific research questions that focus on climate policies, bioenergy and sustainable water strategies as drivers of agricultural land and water-use (see Chapter I, Section 5). These research questions have been explored in detail in Chapters II to V of the thesis and a summary of the key findings is presented in the first section of this chapter. The second section of this chapter provides a synthesis of the results from Chapters II to V alongside the last two research questions presented in Chapter I, Section 5. Finally, the last section of this chapter provides an outlook for future research approaches that can help to improve our understanding of agricultural land and water-use.

1 Summary of key findings of the individual studies

1.1 What are the land-use implications of policies that aim at reducing emissions from land-use change (Chapter II)?

Ambitious climate change mitigation targets will require significant emission reductions. Carbon emissions from land-use change (LUC) currently contribute 1-18% of total anthropogenic CO_2 emissions (IPCC, 2014a). Despite this high uncertainty, avoiding LUC emissions is discussed as a cost efficient mitigation option (Kindermann et al., 2008).

Chapter II of this thesis uses the MAgPIE model to investigate the implications of mitigation policies that include LUC emissions into a global carbon tax regime aiming at ambitious emission reductions (RCP2.6) (IPCC, 2014a). The land-use consequences of these policies are assessed in the context of rising food demand driven by population growth and economic development according to a middle of the road socioeconomic scenario (SSP2) (O'Neill et al., 2014) and climate change impacts consistent with the RCP2.6 emission trajectory.

The results suggest that the main driver of future LUC emissions is expansion of cropland into natural forests and pasture areas. A carbon tax could be a powerful tool for halting cropland expansion, already at moderate tax levels of 25 US$ per ton CO_2. If all emissions from LUC are included into the global tax regime, residual cropland expansion is marginal (35 Mha until 2100 compared to 240 Mha in the absence of a climate policy). The current political debate is , however, focusing on mitigation options for the land-use sector that only target emissions from deforestation (REDD) (Nepstad et al., 2013). Results from Chapter II indicate that a REDD policy that only covers emission from deforestation and disregards emissions from the conversion of other land types does not effectively avoid LUC emissions. While forests are effectively protected by the tax, cropland expansion at the expense of pasture areas is amplified. Thus, global cropland expansion is only marginally reduced by a REDD policy compared to the reference case (by ~35Mha until 2100) and only about half of LUC emissions are avoided. Coverage of all land types will, therefore, likely be a prerequisite for the effectiveness of policies that aim at avoiding LUC emissions.

According to MAgPIE results, avoiding LUC emissions will require a substantial transformation of the agricultural sector and may therefore be challenging to achieve. In the reference case without a climate policy, increasing demand for agricultural products leads to gross land-use changes of almost 800 Mha until 2100. In contrast, future land-use change has to be stopped almost completely if LUC emissions are to be avoided (gross changes of 64 Mha until 2100). Thus, the feasibility of ambitious emission reduction policies in the land-use sector crucially depends on the potential to increase productivity of cropland and pasture areas via technological progress and improved management.

From an environmental perspective, taxing LUC emissions is highly beneficial since it effectively halts agricultural expansion. This is a prerequisite for conserving important land ecosystems like natural forests, but also extensively managed pasture areas (Foley et al., 2011), that feature high biodiversity and provide essential services like water regulation and erosion protection (Bradshaw et al., 2007; Foley et al., 2007). It has to be noted, however, that the necessary intensification of agricultural production can have negative side-effects such as water pollution and soil degradation (Foley, 2005; Grizetti et al., 2011).

These results are of course conditional on the underlying assumptions. A sensitivity analysis shows that the uncertainty of land-use change associated with climate model uncertainty is 10-20% in the Reference scenario (one standard deviation, Chapter II, Table SI-1) and even higher in the mitigation scenarios. Moreover, land-use dynamics in the Reference case are highly dependent on food demand and investment costs for yield increasing technological change while the biggest influence in the comprehensive mitigation scenario is the level of the carbon tax (Chapter II, Table SI-2). While numbers are uncertain, the general finding is robust across the investigated sensitivity scenarios: avoiding LUC emissions will require a significant transformation of the agricultural system in terms of halting agricultural expansion.

1.2 How may terrestrial carbon sequestration policies influence agricultural land-use (Chapter III)?

Terrestrial mitigation policies can go beyond avoiding LUC emissions by sequestering carbon through afforestation and bioenergy in combination with carbon capture and storage (BECCS) (Tavoni and Socolow, 2013).

Chapter III of this thesis investigates the land-use implications of afforestation and BECCS. As in Chapter II, the MAgPIE model is used to project global land-use patterns, food demand is driven by an SSP2 socioeconomic scenario, and the same global carbon tax is applied in the mitigation scenarios. In contrast to Chapter II, the carbon tax does not only penalize LUC emissions from all land types, but also rewards negative emissions from afforestation and BECCS. For the sake of simplicity, climate change impacts are not considered.

Results suggest that terrestrial carbon sequestration may be a very powerful mitigation measure. A combination of afforestation and BECCS can sequester more than 1200 Gt CO_2

until 2100 in the scenarios from Chapter III. This could effectively double the global cumulative CO_2 emission budget that is still available under stringent climate policies (630-1180 Gt CO_2 until 2100) (IPCC, 2014a). Both measures have a similar long term carbon storage potential, but afforestation is cost efficient at lower tax levels than BECCS and may therefore be an option for the near term already. A sensitivity analysis with respect to five model parameters shows that the potential for terrestrial carbon sequestration crucially depends on the level of the carbon tax. The afforestation potential is furthermore strongly influenced by the time horizon that is used to estimate the expected amount of sequestered carbon and the associated financial reward. The main uncertainty about BECCS on the other hand is the potential for geological carbon storage. The CCS technology has not yet been applied at large scales and it is still uncertain whether it can meet the high expectations (Chalmers et al., 2013).

Afforestation has very strong impacts on global land-use dynamics since it may require up to 2800 Mha of land. This is almost twice the amount of current global cropland (1500 Mha) and will require increasing current forestry area by 80% (FAO, 2010; FAOSTAT, 2013). Land requirements of BECCS are lower (500 Mha in 2095) but still correspond to 30% of current global cropland. Land requirements in the combined setting where both measures are allowed are similar to the afforestation-only scenario, but the carbon storage potential is 40% higher. In order to free land for terrestrial carbon sequestration, agricultural areas have to be reduced significantly below their current levels, resulting in less than 1000 Mha of cropland (currently 1500 Mha) (FAOSTAT, 2013) and less than 1500 Mha of pasture (currently over 3000 Mha) (FAOSTAT, 2013).

As already indicated by the very strong impact on land-use dynamics, terrestrial carbon sequestration will require a substantial transformation of agriculture. The feasibility of terrestrial carbon sequestration strategies hinges on significant yield improvements. There is currently a controversial debate in the scientific community, whether average yield increases of 1.4 % yr^{-1} over the 21st century – as required in the most ambitious mitigation scenario – will be feasible (e.g. (Cassman, 1999; Haberl et al., 2013). Afforestation will require a significantly stronger transformation of agriculture than BECCS in terms of land-use change and yield improvements.

Concerning the environmental impacts, there is a fundamental difference between afforestation and BECCS. Afforestation generally has positive environmental impacts because it incentivizes the creation of large-scale forestry areas that are extensively managed and can be important for biodiversity conservation, water regulation, erosion protection, and other ecosystem services (Barlow et al., 2007). BECCS on the other hand will require large areas for 2nd generation bioenergy crop production under intensive management. Thus, BECCS is likely to aggravate the environmental damages associated with intensively managed cropland, including biodiversity loss, soil degradation, and water pollution (Foley, 2005; Grizetti et al., 2011; Jangid et al., 2008).

1.3 Do we have to expect a conflict between sustainable water management strategies and sustainable land management strategies for agriculture (Chapter IV)?

Robust evidence of freshwater ecosystem degradation and impairment of associated ecosystem services has led scientists to call for sustainable water management strategies that explicitly account for environmental water needs, also called environmental flows (Pahl-Wostl et al., 2013; Vörösmarty et al., 2013).

Chapter IV of this thesis investigates how a global environmental flow protection (EFP) regime might affect agriculture by projecting global agricultural land and water-use until the middle of the century with the MAgPIE model. Results suggest that irrigation water withdrawals will continue to grow in the absence of an EFP regime. By 2045, up to 19% of the global land surface could be subject to violations of environmental water requirements. This indicates that implementing EFP strategies will likely be mandatory in order to protect ecosystems from continued degradation. However, reduced water availability for irrigation due to EFP could reduce crop yields and trigger cropland expansion into natural land ecosystems. The scenarios suggest that up to 30 % of global agricultural production might be subject to reduced irrigation water availability due to EFP. Indeed, the introduction of a global EFP regime might reduce agricultural water withdrawals by 20 to 30 % of the current level. Consequences in terms of land-use change are, however, insignificant at the global scale. Across the investigated scenarios, there is a tendency towards cropland expansion due to EFP, but the effect varies between socioeconomic scenarios and is subject to large uncertainties associated with the amount of water required to sustain ecosystems, irrigation costs and the potential for technological yield improvements. Over the full range of investigated sensitivities, global cropland change due to EFP ranges between -90 and +107 Mha. The land-use impact of increasing food demand until 2045 is by a factor 3.5 to 9 stronger than the effect of EFP. This result is robust across the full sensitivity range, including very different assumptions for the socioeconomic drivers of food demand. Thus, there is no indication that sustainable water management strategies are in conflict with efforts to protect valuable land ecosystems.

EFP is designed to enhance environmental sustainability since it protects freshwater ecosystems from degradation. MAgPIE results indicate that negative implications for natural land ecosystems are moderate. Therefore, the overall environmental consequences of EFP can be considered highly beneficial. It has to be noted, however, that only sufficient water quantity for EFP is considered. Thus, freshwater ecosystems in the EFP scenarios may still be prone to degradation because of insufficient water quality and flow timing alterations (Poff et al., 2010).

The transformation of the agricultural system required to achieve EFP appears to be moderate. Reduced water availability for agriculture reduces the potential of irrigation to boost crop yields. This, however, has no significant effect on the agricultural system as a

whole, especially when compared to the transformation required to produce food for a growing world population.

1.4 How may large-scale bioenergy production affect global land and water resources (Chapter V)?

The analysis of terrestrial carbon sequestration policies in Chapter III of this thesis has shown that bioenergy could be a powerful mitigation tool. Its versatility makes bioenergy an attractive energy carrier even in scenarios without climate policies (Rose et al., 2014).

Chapter V of this thesis takes a closer look at the resource requirements of large-scale 2nd generation bioenergy crop production. To this end, the MAgPIE model is used to estimate land and water requirements for the production of 300 EJ of bioenergy from dedicated bioenergy crops at the end of the century. As in Chapters II and III, socioeconomic drivers develop according to an SSP2 scenario. In order to investigate the uncertainty of results, sensitivity analyses with respect to the potential for increasing irrigation efficiency and bioenergy yields are conducted.

Results suggest that cost-efficient large-scale bioenergy production will require substantial land, and especially water resources. Land requirements between 500 and 740 Mha in 2095 could lead to the loss 420 – 560 Mha of natural forest ecosystems and additional pasture areas. Maybe even more worrisome, water requirements for irrigated bioenergy production could increase agricultural water withdrawals to 6400 – 8900 km^3 yr^{-1}. This corresponds to an increase of 70 – 300 % compared to today when considering the uncertainty on current agricultural water withdrawals (Wisser et al., 2008).

There is hope, however: results from Chapter V suggest that it can be possible to avoid negative impacts of large-scale bioenergy production on scarce water resources. Within the MAgPIE modelling framework, it is feasible to produce large amounts of bioenergy without tapping additional water resources for irrigation. The downside of such an approach is, however, that land requirements might increase substantially (+20 – 41%), potentially reaching 1000 Mha or two thirds of current total cropland (FAOSTAT, 2013). Thus, prohibiting irrigated bioenergy production for water resources protection may come at the cost of valuable land ecosystems.

The analysis in Chapter V confirms the results from Chapter III that large scale bioenergy will likely require a fundamental transformation of global land-use patterns (Chapter VI, Section 1.2). An additional finding is that bioenergy cultivation may alter agricultural water-use in an unprecedented way. These findings are in line with other studies on the impacts of large-scale bioenergy on agriculture (Berndes, 2002; Chaturvedi et al., 2013; Popp et al., 2014). Thus, there are strong indications that a fundamental transformation of the agricultural system as a whole will be required in order to provide large amounts of bioenergy.

The potential environmental impacts of large-scale bioenergy production are a reason for serious concern. Already today, freshwater ecosystems are degraded by human intervention (Dudgeon et al., 2006; Vorosmarty et al., 2010) and associated livelihoods are imperilled (UNESCO, 2009). Therefore, a doubling of current agricultural water withdrawals due to bioenergy production – as projected in Chapter V – will likely have severe adverse environmental and social consequences. Furthermore, bioenergy production may substantially increase the competition for land, suggesting that it will be difficult to reconcile ambitious bioenergy strategies with efforts to conserve valuable land ecosystems. The good news is that there is ample space for management strategies to balance the trade-off between land and water resources for bioenergy production. However, even under optimal strategic decisions, environmental impacts will likely be profound.

2 Synthesis of results

2.1 How do different agricultural strategies compare in terms of environmental implications for land and water resources and required transformation of the system?

In order to develop sustainable strategies for agriculture, it is important to explore the potential consequences of different management strategies. One key aspect of such an approach is the evaluation of environmental implications. Furthermore, it is important to investigate the potential societal challenges of a particular agricultural strategy. These can include for example institutional requirements, distributional effects, and food security concerns. While a detailed global analysis of societal challenges is very difficult due to the complexity of the problem, the required transformation of the agricultural system can be used as a proxy for the challenges associated with implementing a particular agricultural strategy. Scenarios that will require a fundamental transformation of the agricultural system are likely to be more difficult to achieve than scenarios that only require minor adjustments.

Synthesizing the results from Chapters II to V (Table 1) suggests that afforestation for terrestrial carbon sequestration would require the strongest transformation of the agricultural system, followed by large-scale bioenergy deployment, and strategies to avoid LUC emissions. Environmental flow protection for sustainable water-use on the other hand might only need a moderate transformation of agriculture as a whole.

Based on the results from this study, it can be expect that all the investigated management strategies would have positive environmental consequences in terms of protecting or even creating valuable land and freshwater ecosystems (Table 1). The only exception is large-scale bioenergy production. Even though 2nd generation bioenergy crops – as considered in this analysis – have a high energy yield (Gerbens-Leenes et al., 2009), they will likely require large areas under intensive agricultural production and could fundamentally increase the human impact on freshwater resources.

Table 1 Comparison of agricultural strategies with respect to required transformation of agriculture and environmental sustainability.

Strategy	Required transformation of the agricultural system	Environmental implications
Afforestation	**very strong** • very high land requirements • requires highly intensified agricultural production	**positive** + restores and creates large forest areas with associated ecological benefits – potential negative consequences of agricultural intensification
Bioenergy	**very strong** • high land requirements • high water requirements • requires highly intensified agricultural production	**negative** – threatens natural land ecosystems – threatens freshwater ecosystems
Avoiding carbon emissions from land-use change	**strong** • requires halting agricultural expansion • requires intensified agricultural production	**positive** + avoids the loss of valuable natural land ecosystems – potential negative consequences of agricultural intensification
Sustainable water-use strategies (environmental flow protection)	**moderate** • reduce irrigated production • marginal consequences in terms of land-use change and land-use intensity	**positive** + avoid freshwater ecosystem degradation + marginal negative side-effects for land ecosystems

In summary, environmental flow protection emerges as a promising strategy to increase environmental sustainability without fundamentally altering the agricultural system. Terrestrial climate change mitigation on the other hand will likely require a fundamental transformation of the agricultural system and may therefore be more challenging to achieve. The environmental implications differ between mitigation strategies. Afforestation and strategies to avoid LUC emissions have a strong overlap with sustainable land

management strategies that aim at conserving pristine land ecosystems and biodiversity (Foley et al., 2011). Bioenergy on the other hand can be expected to have serious negative implications for land and water resources. From an environmental perspective, it would thus be beneficial if terrestrial mitigation policies concentrated on afforestation and avoiding LUC emissions. From an agricultural perspective, however, bioenergy could be favoured over afforestation since the required transformation of the system is stronger for the latter.

The conclusions of this section are of course conditional on the underlying modelling assumptions and on the study design. The major caveats according to the author's personal judgement are summarized in the following.

First, the assignment of environmental consequences to different strategies mainly considers global quantities of land and water. Considering other important aspects, such as water and soil quality, but also specific local environmental conditions, is beyond the scope of this analysis. Therefore, the results only provide a rough picture that can help to identify drivers that may have distinct environmental consequences at large scales.

Second, the investigated set of drivers is not complete (see Chapter I, Section 4). Most prominently, potential impacts of climate change on agriculture are missing. They are highly uncertain (see Chapter I, Section 4.2) and therefore difficult to incorporate into quantitative analyses. In the worst case, negative impacts on agriculture could be substantial, while in the best case, climate change would increase agricultural yields significantly (Rosenzweig et al., 2014). We thus have to consider climate change impacts as one of the biggest uncertainties for future agricultural land and water-use.

Third, the focus on the required transformation of the agricultural system as a proxy for the societal challenges associated with different strategies is only an approximation. In reality, societal challenges of implementing an agricultural policy are influenced by a variety of factors including market structure, political decisions, property rights, but also individual and cultural customs and preferences (Holmes et al., 2007; Rozelle and Swinnen, 2004). Thus, the actual societal challenges for implementing for example sustainable water-use strategies can be high, even though this analysis finds that the required transformation of the system is moderate.

2.2 How large is the operating space for land and water management strategies in agriculture?

It is instructive to investigate the range of potential future agricultural land and water-use across all scenarios that were conducted in Chapters II to V because such a synthesis of results can provide an idea about the operating space for agricultural management strategies (Table 2).

Table 1 Global cropland, pasture and agricultural water-use across all MAgPIE simulations conducted in Chapters II to V. Present situation according to the literature.

Projected range of agricultural resource requirements			
	Present (around 2000)	**Mid-century (2045 – 2050)**	**End of century (2095 - 2100)[A]**
Global cropland (10^6 ha) [B]	1500 [C]	1010 – 2080	720 – 2540
Global pasture area (10^6 ha) [D]	3400 [C]	2110 – 3150	1300 – 3150
Global agricultural water withdrawals (km^3 yr^{-1}) [E]	2200 – 3800 [F]	2540 – 4640	2450 – 8670

A: Does not include results from Chapter IV since the simulation period ends in 2045
B: Can include 2nd generation bioenergy plantations
C: Data source FAOSTAT (2013)
D: Does not include results from Chapter IV since pasture was not explicitly considered there
E: Does not include results from Chapters II and III because water withdrawals were not explicitly considered there
F: Data source Wisser et al. (2008)

Two important conclusions can be drawn from this analysis.
First, according to MAgPIE simulations, biophysical resource availability leaves ample space for increasing land and water inputs for agriculture in the future. This suggests that we do not have to expect agriculture to soon hit biophysical resource limits that would make it impossible to increase global food, energy, and material production. While this picture emerges from an aggregate global perspective, locally, biophysical limits will likely constrain agricultural activity and food production. This suggests that enough food can be provided for a growing world population, but it is very important to improve food distribution and access (Cohen and Garrett, 2009) in order to overcome local resource constraints.

Second, the results suggest that it is possible to reduce the environmental footprint of agriculture in terms of land and water appropriation compared to today, while still providing sufficient food for a growing world population. Thus, sustainable land and water management strategies that aim at the conservation of important ecosystems (Foley et al., 2011; Pahl-Wostl et al., 2013) seem feasible from a global perspective. It has to be noted, however, that a reduction of land and water appropriation for agriculture will require significant technological improvements that increase resource efficiency in crop and livestock production. While such improvements have been achieved in the past (Foley, 2005), it is unclear whether progress can continue at the same pace (Cassman, 1999).

In summary, the analysis in this section suggests that the operating space for agriculture is large enough to support a variety of strategies, ranging from nature conservation to ambitious climate change mitigation and large-scale bioenergy production. Thus, there is hope that the agricultural system is resilient with respect to future challenges – including unforeseen ones – in the sense that a wide range of adaptation mechanisms are feasible.

While being based on a large scenario analysis using a state of the art land and water-use model, these conclusions could be affected by several limitations that can be assigned to three categories:

Overestimation of natural resource capacity

The rather low spatial (500 to 1000 simulation units globally) and temporal (annual) resolution of this analysis may hide local and time specific resource scarcity. By pooling land and water resources over large areas, it is implicitly assumed that water can be transported over long distances. Seasonal water shortages can be missed since available water for irrigation is pooled over the growing period of agricultural crops (see Chapter IV for details). Furthermore, this analysis does not consider extreme weather events that can adversely impact agricultural productivity (IPCC, 2014b). Therefore, agricultural activity in this study can take place in locations where, in reality, frequent weather extremes render farming inefficient. Thus, the upper bound of possible anthropogenic water and land appropriation may be overestimated.

Ignorance of crucial driving factors

Several factors that may influence agricultural production have not been considered in this analysis. For example, soil degradation and insufficient water quality may have adverse impacts on agricultural productivity, while expansion of urban areas may compete for scarce land resources. The evolution of crop yields is a key determinant of agricultural resource use. In this study, it is assumed that technological innovation can continue to increase land and water productivity of crops. While such improvements have been possible in the past (Fischer and Edmeades, 2010; Kijne et al., 2004), some authors argue that there may be natural boundaries for crop productivity (Cassman, 1999; Haberl et al., 2013). A sensitivity analysis in Chapter V has revealed that more pessimistic pathways for crop productivity can substantially increase the resource requirements compared to the default assumptions in this study.

Overestimation of the adaptive capability

Many scenarios require a strong and fast transformation of the system. In reality, this would require transformations of institutions, processing chains, market structures, political and legal frameworks, and individual behaviour. Within MAgPIE, constraints such as institutional

capacity, social and cultural preferences, and market failures are not modelled explicitly, so that the magnitude and the speed of achievable transformations may be overestimated.

3 Outlook for further research

Within this thesis, the results of 182 simulation runs with a state of the art agricultural land and water-use model (MAgPIE) have been evaluated in order to gain insights about the future of agricultural land and water-use. Still, there are several opportunities for extending and improving the here presented analysis. The first part of this section concentrates on potential avenues for future research that are directly connected to the main research objective of this thesis. The second part of this section is devoted to general aspects of model based analyses of future pathways for agriculture. Three key challenges for such analyses are discussed and potential long-term research avenues for addressing these challenges are presented.

3.1 Land and water for agriculture at the global scale: directions for further research

The main research objective of this thesis is to analyse, how agriculture might affect global land and water resources in the future. In the following, I will highlight three potential avenues for further research that can help to improve our ability to address this main research objective.

Evaluating the sustainability of agricultural water-use: moving beyond quantities

Within this thesis, the evaluation of agricultural water-use mainly relies on quantitative consequences for global water resources. However, Poff et al. (2010) highlight that the functioning of important freshwater ecosystems depends on adequate water quantity, quality and timing. Including the aspect of water quality into MAgPIE requires two steps. First, agricultural activity needs to be linked to available data on threats for water quality, as used in Vörösmarty et al. (2010). This can be achieved via econometric regression analysis. Second, the distribution of pollutants throughout the river network needs to be represented. To this end, MAgPIE should be extended by a river routing scheme that allows for downstream migration of pollutants. Since such an approach is computationally very demanding, the river routing can be applied in a post-processing step so that the optimization performance is not affected. Including the aspect of flow-timing within MAgPIE requires increasing the temporal resolution from annual to at least monthly time-steps. From an input data perspective, this is no problem since LPJmL provides water availability and demand data on a monthly basis. However, increasing the temporal resolution increases the complexity of the computational problem. This issue can be partly circumvented by sticking to an annual resolution, but considering twelve water balances, one for each month, instead of one water balance for the whole year. Finally, water is a local resource. Analyses of water quantity, quality and timing would therefore strongly benefit from an increased spatial resolution. Achieving higher spatial resolutions is

computationally highly demanding and will therefore require significant advances with respect to the computational efficiency of MAgPIE.

Feedbacks of land-cover changes on water availability

Available water for irrigation and other anthropogenic uses is highly dependent on evaporation and transpiration from vegetation and land-cover (Haddeland et al., 2011). Within the current analysis framework, the feedback of land-use changes on water availability is not considered. In order to include this feedback, the interaction between MAgPIE and the global vegetation and hydrology model LPJmL needs to be extended by returning MAgPIE land-use information to LPJmL and deriving updated water availability patterns. Such an enhanced analysis framework could be used to tackle the interesting research question of how large-scale land-use changes due to afforestation might affect global freshwater availability.

Interaction between agricultural production and demand

In Chapter V, it has been shown that a strategy that prohibits irrigated bioenergy production can have consequences for the price of primary bioenergy. In the same way, environmental flow protection strategies that were investigated in Chapter IV could have impacts on food prices. More generally, agricultural production strategies can change commodity prices and trigger changes in food demand and food basket composition (Valin et al., 2014). Within the current MAgPIE version, demand for agricultural products is income inelastic, i.e. feedbacks of supply side strategies on demand are not reflected. This could be changed by introducing explicit price elasticities of food, material, and bioenergy demand. In this context, it is important to note that food demand is strongly shaped by consumption habits and preferences that can change in the long run (Allais and Nichèle, 2007). Available food price elasticities from literature (USDA, 2014) can therefore not be applied for long-term analyses. The work by Valin et al. (2014) who review different approaches for incorporating price elasticities into agricultural models should therefore be the starting point for estimating price elasticities within MAgPIE.

3.2 General long-term prospects for model based analyses of future agricultural pathways

In the previous paragraphs, it has been discussed how MAgPIE could be specifically improved for analyses of agricultural water and land-use. There exist, however, also challenges that are common to any model based assessment of future agricultural pathways. As will be discussed below, addressing these challenges is not possible within a single modelling framework, but requires new modelling paradigms and a community-wide effort for data collection and joint analyses. The following paragraphs discuss three key challenges for modelling the future of agriculture and propose long-term strategies for dealing with them.

3.2.1 Key challenges for projecting future pathways for agriculture

The three challenges that are highlighted in this section are selected as the most important ones by the author of this thesis. There are of course many other challenges as discussed by e.g. Beven et al. (2002), Sohl et al. (2013), and Brown et al. (2013).

Unpredictable behaviour

The future development of the agricultural system is conceptually unpredictable since it involves non-deterministic elements such as human decision making (see Schwanitz, 2013). The challenge of unpredictable behaviour can be addressed by conducting scenario analyses, i.e. by asking the question "How would the system evolve under the assumptions that define the investigated scenario?" (Schwanitz, 2013). Each individual scenario is, however, extremely unlikely to come true because many assumptions have to be made due to the complexity of the system. Therefore, it is important to consider a large set of scenarios in order to cover a large part of the assumption space.

Uncertainty

Data about the current state and historical evolution of the system is highly uncertain (see Chapter I, Sections 2 and 3). This is problematic since it is difficult to justify values of model parameters that are not estimated from empirical data (Jakeman et al., 2006). Parameters that are poorly constrained by empirical data have to be considered as an assumption. Therefore, a scenario analysis is required to test the sensitivity of results with respect to plausible variations of uncertain parameters.

Complexity

The agricultural system is highly complex. It is influenced by a large number of biophysical and socioeconomic driving factors that can vary across space and time. Modelling complex systems requires complex models that adequately represent important driving processes and their interactions. In the context of unpredictable behaviour and data uncertainty this implies a large number of assumptions concerning model structure and parameter values that require extensive sensitivity analyses. Moreover, simulating a large set of scenarios with complex models can become difficult due to long simulation time (up to several days in the case of MAgPIE). Finally, model transparency – a key aspect of model credibility (Schwanitz, 2013) – requires significant and continued efforts and resources for model documentation.

In summary, future research should aim at:

- better testing of assumptions via more comprehensive sets of scenarios,
- constraining values of model parameters via improved data quality and accessibility
- improving transparency via better model documentation.

3.2.2 Three steps towards improving projections of future pathways for agriculture

Addressing the challenges that were identified in the previous section will require new approaches towards model development, data availability, and study design.

Model development

A well-structured model development process is of vital importance for the usefulness of land-use models. Over the last years, MAgPIE has been improved significantly with respect to flexibility, user friendliness, and transparency. Further efforts in these directions are, however, required. First, the transparency of MAgPIE should be increased by making the model documentation – at the moment only for internal use – publicly available. A more ambitious step would be the publication of the source code under an open source licence. Second, the time required for a simulation run should be decreased in order to facilitate larger sets of scenario runs. Recent experience with MAgPIE shows that the acquisition of dedicated expertise in computational optimization problems can be important in this context. A more efficient model code would additionally allow increasing the complexity, e.g. spatial or temporal resolution. Third, model development is often triggered by the need to represent important processes within the model. At the same time, data availability is a key prerequisite for implementing new processes (see Jakeman et al. (2006)). Therefore, the availability of interesting data could be put into the focus of future model development strategies. This would imply that model development would be largely triggered by the availability of interesting data. In a second step, it would be decided, which conceptual structure of new model components would be most appropriate for integrating the information from this data into the existing model.

Data availability

Projecting agricultural land and water-use requires a large variety of biophysical and socioeconomic data. There is a large number of datasets available, comprising census data, remote sensing data and observations from local case studies and measurement stations. Local observations can provide accurate and reliable data but are of limited use for global modelling due to missing spatial coverage. Census data and remote sensing data on the other hand are prone to large – mostly not explicitly quantified – uncertainties (Fritz et al., 2010; Grainger, 2008; Miller et al., 2013).

One promising option for addressing the data issue is to establish a community knowledge hub. Such a hub should host as many relevant datasets as possible in a consistent format, comprising remote sensing data, census data, but also observations from case studies. This would facilitate meta-analysis in order to quantify uncertainties and would thus help to overcome the severe shortcoming that there exist a lot of quantitative data without quantified uncertainties. Moreover, data harmonization at the hub would allow for combining reliable data from local case studies in order to obtain high quality data with

good spatial coverage. The most promising project in this direction is the GEOSHARE initiative (Hertel and Villoria, 2014). It has been initiated by renowned scientists with the aim to provide a globally consistent, temporally opportune, and locally relevant database for better decision making. The significant effort of this task can only be made with the contribution of the whole research community, so that the contribution to GEOSHARE is a very promising avenue for further research.

Study design

Traditional analyses employ a single modelling approach to derive estimates of agricultural land and water-use. This approach can be viewed as being 'consolidative', i.e. it attempts to bring all known factors and relevant information together into a single package that is used as a surrogate of the real world (Bankes, 1993). Only quite recently, the land-use modelling community has started to conduct dedicated model intercomparison exercises that allow for gaining insights from an ensemble of simulations employing different modelling approaches (Popp et al., 2014; Rosenzweig et al., 2014, 2013; Smith et al., 2010). These exercises have revealed that results vary considerably between different models (Popp et al., 2014; Rosenzweig et al., 2014; Schmitz et al., 2014). In this context, the traditional approach of building one consolidated model or to consolidate model ensemble results has to be viewed highly critical. A promising approach to overcome this dilemma can be found in the concept of 'exploratory modelling' (Bankes, 1993). Under this paradigm, research questions are addressed by employing the full ensemble of existing models (or a sufficiently large subset) and conducting sensitivity analyses with respect to model parameter values. Thus, robust features can be identified and uncertainty can be quantified, resulting in a better understanding of where quantitative models can provide useful information and where they currently fail to do so. Again, a community knowledge hub could be pivotal in this context if it provides access to an ensemble of models that can be used for exploratory analyses by a large group of users.

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Statement of contribution

Introduction and conclusion were written by Markus Bonsch with valuable comments by Alexander Popp and Benjamin Bodirsky.

Chapter II: Land use protection for climate change mitigation

State: published in Nature Climate Change

A.P. designed the overall study; F.H. and M.Bonsch. handled the MAgPIE model runs. A.P. wrote the manuscript with important contributions from F.H., B.L.B., C.M. and M.Bonsch.; A.P., F.H., M.Bonsch. and B.L.B. analyzed results; F.H., I.W., B.L.B., M.Bonsch., J.P.D., A.P., M.S. A.B. and H.L.C. contributed in developing and improving the MAgPIE model; C.M. and S.R. provided biophysical input data from LPJmL; all authors discussed and commented on the manuscript.

Chapter III: Investigating afforestation and bioenergy CCS as climate change mitigation strategies

State: published in Environmental Research Letters

F. Humpenöder, A. Popp and J.P. Dietrich designed the overall study; F. Humpenöder handled the MAgPIE model runs; F. Humpenöder and J.P. Dietrich analyzed results; F. Humpenöder wrote the manuscript with important contributions from A. Popp, B.L. Bodirsky and M. Bonsch; C. Müller provided biophysical input data from LPJmL; F. Humpenöder, M. Bonsch, I. Weindl, J.P. Dietrich, D. Klein, B.L. Bodirsky, M. Stevanovic, H. Lotze-Campen and A. Popp contributed in developing and improving the MAgPIE model; all authors discussed and commented on the manuscript.

Chapter IV: Environmental flow provision: Implications for agricultural water and land-use at the global scale

State: published in Global Environmental Change

M. Bonsch designed the overall study; M. Bonsch handled the MAgPIE model runs; M. Bonsch, S. Rolinski, A. Biewald, and A. Popp analyzed the results; M. Bonsch wrote the manuscript with important contributions from S. Rolinski, A. Biewald, A. Popp, J. Heinke, C. Schmitz, and B. Bodirsky; M. Bonsch, C. Schmitz, K. Högner, A. Biewald, S. Rolinski, J. P. Dietrich, I. Weindl, B. Bodirsky, F. Humpenöder, M. Stevanovic, H. Lotze-Campen, and A. Popp contributed in developing and improving the MAgPIE model; S. Ostberg provided biophysical input data from LPJmL; all authors discussed and commented on the manuscript.

Chapter V: Trade-offs between land and water requirements for large-scale bioenergy production

State: published in Global Change Biology – Bioenergy

M. Bonsch and A. Popp designed the overall study with important contributions by F. Humpenöder; M. Bonsch handled the MAgPIE model runs; M. Bonsch, F. Humpenöder and J.P. Dietrich analyzed results; M. Bonsch wrote the manuscript with important contributions from F. Humpenöder, A. Popp and J.P. Dietrich; M. Bonsch, F. Humpenöder, J.P. Dietrich, S. Rolinski, A. Biewald, I. Weindl, M. Stevanovic, B. Bodirsky, H. Lotze-Campen, and A. Popp contributed in developing and improving the MAgPIE model; all authors discussed and commented on the manuscript.

Tools and Resources

The following tools were used for this thesis:

- R (R Core Team, 2014),
- GAMS (Rosenthal, 2014) with the CONOPT solver (Drud, 1994),
- LaTeX (Lamport et al., n.d.),
- Open Office,
- Microsoft Office,
- Zotero

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Acknowledgements

This work wouldn't have been possible without support from colleagues and friends, all of whom I want to thank here.

I want to thank Ottmar Edenhofer and Alexander Popp for giving me the opportunity to join PIK and to pursue my research interests that led to this dissertation. They were always very supportive and professional and helped me a lot to structure my work and thoughts. Many thanks also to Dorothe Ilskens and the whole administration team who arranged all administrative matters perfectly and were a great and friendly support. Special thanks go to Susanne Rolinski and Jan Philipp Dietrich who were always willing to provide help and guidance. Without them, I would not have managed to carry on during the most difficult times.

I am very grateful for fruitful discussions and valuable scientific exchange with all of my colleagues in the landuse group and beyond, including Alexander Popp, Susanne Rolinski, Jan Philipp Dietrich, Hermann Lotze-Campen, Christoph Müller, Miodrag Stevanovic, Florian Humpenöder, Christoph Schmitz, Isabelle Weindl, Benjamin Bodirsky, Anne Biewald, Xiaoxi Wang, Katharina Waha, Ulrich Kreidenweiss, Jens Heinke, Sebastian Ostberg, Yvonne Jans, Elmar Kriegler, and many more.

Many thanks to my office mates Misko and Florian for the great time that I was allowed to spend with them and for many enjoyable professional and private talks. I want to thank Susanne Rolinski, Anne Biewald, Jan Philipp Dietrich, Benjamin Bodirsky, Jessica Strefler, and David Klein for nice kicker and card playing events.

Finally, I want to thank my friends and family for their support and for putting up with me during times of frustration and despair. I owe thanks for that to my parents, my flatmates in U64, and especially to Teresa.