AGRICULTURAL NITROGEN POLLUTION: THE HUMAN FOOD-PRINT

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Summary

The provision of sufficient food for a growing world population is one of the most basic needs of humanity. The agricultural sector, which has the function to provide this food, is at the same time the main interface between anthropogenic and natural systems. The continuous expansion and intensification of agriculture has driven the nitrogen cycle, a central Earth-system process, out of the state it was residing in during the geological Holocene epoch. This disturbance leads to a number of undesirable feedbacks on human society, lowering the benefits of a multitude of ecosystem services and harming human health.

The overarching topic of this dissertation is how future food demand changes the global nitrogen cycle and nitrogen pollution. The research questions are: (1) What are plausible scenarios for global food demand in the 21st century? (2) What is the current state of the agricultural nitrogen cycle? (3) How will the nitrogen cycle evolve into the future? (4) Which additional pressure on the nitrogen cycle might evolve from the production of bioenergy? (5) How would the adoption of nitrogen mitigation action in food demand and in agricultural production change the nitrogen cycle and nitrogen pollution?

To answer these questions, this dissertation develops a new, simple and transparent method to create long-term scenarios of global food demand. These scenarios are subsequently used by the Model of Agricultural Production and its Impact on the Environment (MAgPIE). The model is a hybrid between a socio-economic and a biophysical land-use model that can be used to derive scenarios of the future development of the agricultural sector. The existing model was extended by a nitrogen mass-balance module, simulating the major nitrogen flows on cropland soils, in livestock production, in food processing and in household consumption. Finally, this extended model is used to create long-term scenarios of the agricultural nitrogen cycle and nitrogen related pollution, and to analyse the effect of nitrogen mitigation measures.

Reference scenarios without mitigation action indicate that until 2050, growing demand for food will expand most nitrogen flows in the agricultural sector, with global nitrogen pollution rising to unprecedented levels. An extension of bioenergy production would add additional pressure on the nitrogen cycle. Diverging from business-as-usual projections, this study shows that reduced food waste and livestock consumption, in combination with strong efficiency improvements in agricultural production, could reduce nitrogen pollution below current levels. However, even under such ambitious mitigation actions, the global nitrogen cycle will not return to pre-disturbance Holocene conditions and the remaining nitrogen pollution will continue to be detrimental to human welfare.

Zusammenfassung

Die Versorgung mit Nahrungsmitteln ist eines der grundlegendsten Bedürfnisse der Menschheit. Der landwirtschaftliche Sektor, der diese Nahrungsmittel herstellt, ist gleichzeitig die Hauptschnittstelle zwischen Mensch und Natur. Die kontinuierliche Ausdehnung und Intensivierung der Landwirtschaft hat einige grundlegende Prozesse des Erdsystems verändert - so auch den globalen Stickstoffkreislauf, der sich weit von seinem Zustand in der restlichen Ära des Holozäns entfernt hat. Die Störung des Stickstoffkreislaufs führt zu einigen unerwünschten Rückwirkungen auf die Menschheit. So schmälert Stickstoffverschmutzung den Nutzen einer Vielzahl von Ökosystemdienstleistungen und beeinträchtigt die menschliche Gesundheit.

Das übergreifende Thema der Dissertation ist die Auswirkung der zukünftigen Nahrungsmittelnachfrage auf den global Stickstoffkreislauf und auf Stickstoffverschmutzung. Die Forschungsfragen sind im Einzelnen: (1) Was sind plausible Szenarien für die globale Nahrungsmittelnachfrage im 21ten Jahrhundert? (2) In welchem Zustand befindet sich der landwirtschaftliche Stickstoffkreislauf aktuell? (3) Wie wird sich der Stickstoffkreislauf in der Zukunft entwickeln? (4) Welcher zusätzliche Druck könnte von der Bioenergie auf den Stickstoffkreislauf ausgehen? (5) Wie würde sich die Implementierung von Stickstoffmitigationsmaßnahmen in der Nahrungsmittelnachfrage und in der landwirtschaftlichen Produktion auf den Stickstoffkreislauf und auf Stickstoffverschmutzung auswirken?

Zur Beantwortung dieser Fragen wird in dieser Dissertation zuerst eine neue, einfache und transparente Methode entwickelt um Langzeitszenarien für die globale Nahrungsmittelnachfrage zu erstellen. Diese Szenarien werden anschließend vom „Modell für Landwirtschaft und seine Auswirkungen auf die Umwelt“ (MAgPIE) genutzt. Dieses Modell ist ein Hybrid zwischen einem sozioökonomischen und einem biophysikalischen Landnutzungsmodell, welches wiederum verwendet werden kann um Szenarien für die zukünftige Entwicklung der Landwirtschaft zu erstellen. Das bereits existierende Modell wurde um ein Stickstoffmassenbilanzmodell erweitert, welches die wichtigsten Stickstoffflüsse im Ackerland, in der Tierhaltung, in der Nahrungsmittelverarbeitung und in Haushalten simuliert. Schlussendlich wird dieses erweiterte Modell genutzt um Langzeitszenarien des landwirtschaftlichen Stickstoffkreislaufs und der Umweltverschmutzung durch reaktiven Stickstoff zu erstellen, sowie um den Effekt von Mitigationsmaßnahmen zu analysieren.

Die Szenarien zeigen dass bei unveränderten Rahmenbedingungen und ohne Mitigationsmaßnahmen die wachsende Nahrungsmittelnachfrage die meisten Stickstoffflüsse im landwirtschaftlichen Sektor bis 2050 stark ansteigen lässt, wodurch die globale Stickstoffverschmutzung nie dagewesene Höhen erreicht. Ein Ausbau der Bioenergieproduktion würde zusätzlichen Druck auf den Stickstoffkreislauf ausüben. Die Reduktion von Nahrungsmittelabfällen, die Senkung der Nachfrage nach tierischen Produkten, und starke Effizienzverbesserungen im Ackerbau sowie der Tierhaltung könnten jedoch in Kombination die Stickstoffverschmutzung wieder unter das aktuelle Niveau senken. Selbst in diesem Fall ambitionierter Mitigationsmaßnahmen verbleibt der Stickstoffkreislauf jedoch noch immer weit von seinem natürlichen Zustand im Zeitalter des Holozäns entfernt, so dass das Wohlergehen der Menschen auch weiterhin durch die mit Stickstoff zusammenhängende Umweltverschmutzung beeinträchtigt wird.

Chapter I

Introduction

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Content of chapter I: Introduction

1 Overview

The current state of the Earth system cannot be understood without considering human activity. Therefore, scientists suggested to name the current geological epoch Anthropocene, superseding the Holocene that started about 12 000 years ago (Crutzen 2002; Rockström et al. 2009; Steffen et al. 2011). One of the most important Earth system processes altered by human activity is the nitrogen cycle. Since pre-industrial times (Vitousek et al. 2013), human activity has doubled the global sources of reactive nitrogen (Nr) that are injected into the global nitrogen cycle every year, and more than quadrupled the terrestrial Nr sources (Fowler et al. 2013).

This has detrimental effects on human welfare. Natural systems are destabilized and more vulnerable to all sorts of external stress, as the "nitrogen cycle is generally tuned to operate with nitrogen in short supply" (Sutton et al. 2009). Ecosystem services like fish production or disease control are severely impaired. Nitrogenous water and air pollution also directly harms human health, causing for instance cancer and respiratory diseases. Finally, Nr interacts in various ways with the climate system and contributes to the depletion of stratospheric ozone. (Sutton, Howard, et al. 2013; Sutton et al. 2011)

The agricultural sector is the major human interface to many natural systems (Steffen 2004), in particular in regard to the nitrogen cycle (Fowler et al. 2013). Nr is an irreplaceable nutrient for crop and livestock production. The low natural availability of Nr restricted for a long time agricultural production, forcing farmers to efficiently use and recycle nutrients. The development of the Haber-Bosch ammonia synthesis in the 20th century however made it possible to provide agriculture with cheap Nr in unlimited amounts, which eliminated many incentives for efficient management. At the same time, agricultural production was enlarged to provide food for a growing world population and to fulfil its demand for an increasingly affluent diet. (Smil 2004)

This dissertation analyses how future food demand alters the agricultural nitrogen cycle and Nr pollution.

The introduction will first elaborate on the broader background of the topic, then present the main research question of this thesis. Subsequently, a description of the methodological approach is followed by a line-out of the whole thesis.

2 Background

The knowledge about the global nitrogen cycle is still incomplete and connected to large uncertainty[1]. However, the scientific community made in recent years large progress in assembling, interconnecting and summarizing the current state of knowledge, most importantly through the European Nitrogen Assessment (Sutton et al. 2011), the report "Our nutrient World" (Sutton, Howard, et al. 2013), and as a side-product of various reports of the International Panel of Climate Change (e.g. IPCC 2013; IPCC 2007).

[1] In the following, only the best guess estimates will be quoted for most numeric quantifications. Please note that all estimates are subject to large uncertainty ranges (see section 8.1.2).

The following subsections will start by giving a short overview on the current state of the natural and agricultural nitrogen cycle. Subsequently, the effects Nr has on the environment are described. The last section analyses the Nr efficiency of the agricultural supply chain, highlighting major inefficiencies that are at the same time levers for mitigation.

2.1 Nitrogen in nature and agriculture

Nitrogen is a nutrient that is a prerequisite for all life on Earth. Every deoxyribonucleic acid (DNA), ribonucleic acid (RNA), protein and peptine contains nitrogen, as well as the chlorophyll pigments that are crucial for photosynthesis (Kramer 2005). Despite its affluent availability in the Earth's atmosphere (78% of the air consists of dinitrogen (N_2)), the nutrient is very scarce and a limiting factor to life as the unreactive molecular form of N_2 cannot be absorbed by most organisms. Splitting up the triple bound that connects the two N atoms of N_2 requires large amounts of energy, such that N_2 is only "fixed" into reactive forms (Nr) like ammonia (NH_3) in very limited amounts through lightning or geological processes. Furthermore, a range of micro-organisms (diazotrophs) developed in their evolutionary process the ability to fix dinitrogen from the atmosphere. Some of them live in symbiosis with plants, most importantly with leguminous crops, but also with other species like sugar-cane or alder trees. The bacteria infect roots or stems of these plants and receive energy in the form of sugar in exchange for Nr. (Smil 2004; Vitousek et al. 2013)

Despite the existence of biological Nr fixation, Nr remains in short supply for many natural ecosystems, especially in temperate and boreal zones, as well as tropical savannahs[2]. There are various reasons for this: Nr is more mobile than other nutrients, and disturbances like fires deplete Nr availability more than those of other nutrients. Another major reason is that biological fixation is very energy intensive, and Nr-fixing plants often have significantly lower net primary production. Therefore, most organisms instead try to gather the Nr available in their environment. Nr is therefore often recycled in the food supply chain and detained in the ecosystem to prevent or at least slow down its denitrification into N_2. (Vitousek and Howarth 1991)

The low availability of Nr does not only limit the growth of natural ecosystems, but also of agricultural production. Agricultural activity is therefore shaped by management activities to overcome this nutrient scarcity. Nutrient management practices can be divided mainly into six categories:

- **Capturing the Nr provided naturally by the environment.** For this purpose, crop cultivation and livestock grazing can be extensified to capture the Nr that is naturally deposited on the area or fixed biologically by free-living diazotrophs. Moreover, shifting cultivation and slash-and-burn practices allow to capture Nr that accumulated over longer time-periods in vegetation or soils.
- **Increasing biological Nr fixation.** Cultivating crops that live in symbiosis with diazotrophs (such as leguminous plants or sugarcane), and initialising the Nr fixation through inoculation can reduce the requirements of other Nr fertilizers. Management practices include also the cultivation of Nr-fixing cover crops that are not harvested but ploughed into the soil to provide nutrients for the subsequent crop.

[2] Tropical forests as well as freshwater and most marine ecosystems are usually not limited by Nr, but by other nutrients, most importantly phosphorus.

- **Recycling Nr.** Crop residues, animal manure and human sewage can be recycled to agricultural land.
- **Increasing nitrogen efficiency.** Fertilization requirements can be lowered through reducing nitrogen losses on the field. This can be achieved by fertilizing the right type and right amount of Nr at the right time at the right space (4R).
- **Mining Nr.** Nitrogen can be mined from nitrogen rich natural deposits and applied on the fields as fertilizers. The largest natural deposits are Peruvian seabird-guano and Chilean saltpetre.
- **Applying industrial fertilizer.** As alternative to the above organic fertilizers, the Haber-Bosch ammonia synthesis allows the production and use of inorganic fertilizers. The industrial process uses the energy of natural gas to fix Nr from atmospheric dinitrogen.

Until the beginning of the 20th century, only the first five practices existed, which were costly in terms of labour and land. The scarcity of land set limits to extensification and to shifting cultivation practices (Weaver 1979), lower yields of Nr fixing plants and worse digestibility of leguminous proteins limited the potentials of biological fixation (Smil 2000), and the deposits of natural Nr suitable for mining were declining rapidly in the end of the 20th century (Hager 2008).

Today, agricultural production would be certainly limited by Nr shortages if the Haber-Bosch ammonia synthesis would not have been developed in the early 20th century (Smil 1997; Erisman et al. 2008). Originally promoted to provide Nr for explosives to war-time Germany, technological improvements after the wars made this chemical process an inexhaustible source of cheap Nr fertilizer (Smil 1997), and sparked the Green revolution of agriculture by removing nutritional limitations. Without this technology, the simultaneous population growth and dietary affluence observed in the 2nd half of the 20th century would not have been realizable (Smil 1997).

At the same time, the importance of most other nutrient management options declined in the last 50 years. Natural Nr sources which were captured by extensification became less important, as the largest increase of agricultural production was achieved by intensification instead of extensification (Huber et al. 2014). Mined Nr is more costly than industrial fertilizers and today only supplies some niche markets. Recycling of manure declined with increasing geographical concentration of livestock production (Sutton, Howard, et al. 2013), which was made possible through the cheap availability of concentrate feedstock (instead of pasture, crop residues or non-eaten food). The global nitrogen uptake efficiency on agricultural land (Nr withdrawals by plants divided by total Nr inputs to plants) fell from 63% in 1900 to 44% in 2000 (Bouwman 2011, table 1). Only the cultivation of Nr fixing crops is a continuously used nutrient management option. Compared to other crops, the production of Nr-fixing crops increased over-proportionally in the last 50 years (FAOSTAT 2013), mostly driven by soybean, groundnut and sugarcane production used for livestock feed and bioenergy.

In effect, human activity (in particular agriculture) altered the global nitrogen cycle fundamentally in the last century (Erisman et al. 2008). Natural processes fix every year about 58 Tg Nr in terrestrial ecosystems (Vitousek et al. 2013) and 157 Tg Nr in oceans (Voss et al. 2013). In contrast, humans fix currently via the Haber Bosch Synthesis 108 Tg Nr for synthetic fertilizers (IFA 2013, consumption in 2011) and 20 Tg Nr for industrial purposes (Fowler et al. 2013). Herridge, Peoples and Boddey (2008)

estimate that biological fixation on croplands and pastures contributes further 60 Tg Nr[3]. Combustion of fossil fuels in the transport, energy and industry sectors releases about 60 Tg Nr in the form of nitrogen oxides (NO_x) (EC-JRC/PBL 2011, aggregated from NO_x emissions by sector in 2008), which were either stored in the fuel or fixed by the combustion process. Similarly, large-scale biomass burning also releases about 18 Tg Nr in the form of nitrogen oxides (NO_x) and ammonia (NH_3) (EC-JRC/PBL 2011, aggregated from NO_x and NH_3 emissions by sector in 2008), and a probably even larger quantity of Nr that is directly denitrified into N_2. Finally, an often neglected source is the release of Nr when soil organic matter depletes after the conversion of natural soils (in particular wetlands) to croplands. This process may release about 30 Tg Nr per year (Vitousek and Matson 1993). In total, human activity more than doubled global Nr fixation and quintupled terrestrial fixation.

There are hardly any long-term storages in the nitrogen cycle (Sutton, Howard, et al. 2013), such that all fixed Nr is released into the environment. Nr fixed or released from combustion is directly volatilized into the air, while the Nr entering the agricultural sector passes through the agricultural supply chain before it is lost mainly at three stages:

- The largest losses occur on the field during and after fertilization of soils with inorganic fertilizers, manure, or crop residues. Nr that is not taken up by plants leaches into the groundwater, volatilizes in the form of nitrogen oxides (NO_x), ammonia (NH_3) and nitrous oxide (N_2O), is carried away by wind and water erosion, or is simply denitrified into N_2.
- Losses occur also during the management of manure in stables and animal waste management systems, mostly through volatilisation in the form of NO_x, NH_3, and through denitrification.
- Also the nutrients reaching households are ultimately lost. Household waste decomposes on landfills or is burned, while sewage is either directly discharged into rivers and oceans, or is treated in sewage works where most Nr is denitrified to N_2.

2.2 Nr pollution

While being a nutrient in the food supply chain, Nr becomes a pollutant when it reaches the environment in large amounts. Once released to the environment, Nr can travel large distances downstream in water bodies and even several hundred kilometres in the air (Sanderson et al. 2008). Thereby, the same Nr particle can have subsequently a cascade of effects on the environment (Galloway et al. 2003) before it eventually denitrifies. Reactive nitrogen is a pollutant to the air, water, soil, and the atmosphere.

2.2.1 Air pollution

According to Moldanova et al. (2011), Nr in the air is a pollutant in various ways. Nitrogen dioxide (NO_2) is a toxic gas that harms human and animal health, causing short-term and chronic diseases like asthma and cancer. It is also light-scattering and aggressive to materials, corroding buildings, technical structures, including objects of cultural heritage. NO_2 catalyses the formation of tropospheric ozone, which is in turn an oxidant damaging human, animal and plants cells, as well as being aggressive to several types of materials. Furthermore, NH_3 and NO_x significantly contribute to the formation of particular matter. Once particular matter is inhaled, it may lead to a number of

[3] Actual fixation might be lower as their estimates for fixation by leguminous crops are too high (see chapter 4)

respiratory and cardiovascular diseases, including lung cancer, inflammation of airways, or infarcts. Finally, volatilized Nr compounds can travel large distances in the air and reach remote ecosystems, altering them by acidification and eutrophication. (Moldanova et al. 2011)

2.2.2 Water pollution

According to Grizetti et al. (2011), Nr that reaches drinking water can be toxic and may lead to the blue baby syndrome or colon cancer. Water being a transport medium for many soluble Nr compounds, nutrients can travel down-stream in surface water, ground water, aquifers and oceans. Together with other nutrients, most importantly phosphorus, Nr can destabilize aquatic ecosystems. It may lead to eutrophication, anoxia, toxic algal blooms, habitat loss and general vulnerability to other stresses. Thereby, it impairs ecosystem services like fish provision, recreational value or the purification of water. (Grizzetti et al. 2011)

2.2.3 Soil pollution

Nr reaching natural ecosystems through atmospheric deposition may have negative consequences on these soils, acidifying the soils and lowering micro-organism and plant diversity. In certain soil types, Nr deposition releases heavy meals or sulphates that may subsequently reach drinking water. In contrast, positive effects on soil quality dominate in managed agricultural soils. (Velthof et al. 2011)

2.2.4 Atmospheric pollution

Nr emissions have effects on the ozone layer and on the Earth's climate system.

Nitrous oxide (N_2O) is known to catalyse the destruction of the stratospheric ozone layer. Today, N_2O is the single most important ozone-depleting substance (Ravishankara, Daniel, and Portmann 2009). Stratospheric ozone depletion leads to an increased exposure of life to UVB-radiation, increasing the risk of cancer.

The inter-linkages of the nitrogen cycle with the Earth's climate are complex and in parts still subject so large uncertainty.

Public and broader scientific perception mostly focuses on the best-understood process, the direct radiative forcing of N_2O that warms the climate, and which is currently the third most important well-mixed greenhouse gas (IPCC 2013). Capturing radiation in other spectral windows than carbon dioxide (CO_2) and methane (CH_4), N_2O is a powerful greenhouse gas with a 100yr global warming potential of 265 CO_2-equivalents per ton Nr (IPCC 2013). With an atmospheric lifetime of 121 years, it is a long-lived greenhouse gas (IPCC 2013). And while there is uncertainty concerning the rate of atmospheric destruction and thereby lifetime, as well as the precise global warming potential, its net direct radiative forcing is little controversial (IPCC 2013). The main uncertainty for this process actually lies with the N_2O emission factors from soils and water bodies where even the order of magnitude is uncertain (Butterbach-Bahl, Eggleston 2006). A recent study also shows that emission factors are climate-sensitive and may increase under climate change (Sutton, Reis, et al. 2013).

Besides its direct global warming potential, Nr has also strong so-called indirect radiative effects, which stem from interactions of Nr with non-nitrogenous greenhouse gases (IPCC 2006). NO_x for instance may lead to the formation of tropospheric ozone, a greenhouse gas that in turn lowers the lifetime of the greenhouse gas CH_4. With its contribution to aerosol formation, Nr has a cooling

effect on the climate. Also, by fertilizing natural ecosystems with Nr, CO_2 is removed from the atmosphere and bound in biomass (see IPCC 2013, Erisman et al. 2009; Butterbach-Bahl et al. 2011 for a detailed list of these indirect effects).

Both cooling and warming processes strongly influence the climate. However, they are currently largely cancelling out. Erisman et al. (2011) estimate a net-cooling effect (-0.24 W/m²) from all Nr related processes, with an uncertainty range covering also a light net-warming. The individual processes may however evolve unevenly in the future. The warming N_2O has a long lifetime and will accumulate in the atmosphere. At the same time, air pollution regulations might lead to a decline of NO_x and NH_3 compounds with a net-cooling effect. Also the fertilization of natural ecosystems may saturate in the long-term.

2.2.5 Energy requirement

Beside its polluting effects once released into the environment, the fixation of Nr itself is not without environmental effects, as it is very energy intensive. Inorganic fertilizers require 1-2 per cent of global energy production in the form of natural gas for the Haber-Bosch synthesis (Sutton, Howard, et al. 2013; Ramírez and Worrell 2006), a process that also releases large amounts of CO_2. Also the biological fixation is problematic, as Nr fixing crops with comparatively low yields require a larger cultivation area.

2.3 Inefficiencies in the agricultural sector and potential mitigation actions

The high amount of Nr losses to the environment is the consequence of the low Nr efficiency of the agricultural sector. Agriculture is receiving approximately 250 Tg Nr[4], of which only 19 Tg Nr are actually eaten by consumers as food (Billen, Garnier, and Lassaletta 2013), a mere 7.6%. The major inefficiencies, the socio-economic factors that cause them, and potential mitigation options are discussed in the following.

2.3.1 Field losses

Currently, only about half of the Nr applied on global croplands is actually taken up by crops. Losses occur first as volatilisation losses when inorganic fertilizers or manure are applied on fields (Eggleston et al. 2006). Moreover, in the time that the nutrients reside on the field before being taken up by the plant biomass, they are subject to denitrification losses, as well as leaching losses in irrigated systems or in soils where water-holding capacities are exceeded (Eggleston et al. 2006). Hence, one important mitigation option is to increase fertilizing efficiency and thereby reduce Nr losses on the field.

High fertilization efficiency can be reached by monitoring the nutrient dynamics on the field and by adjusting the selection, timing, placing and dosing of fertilizers accordingly. Nutrient losses can be avoided through reducing tillage, by combating erosion, by planting cover- and nutrient catch-crops, or by using biological or synthetic nitrification inhibitors. As the most limiting factor determines plant growth, removing water shortage, fighting pests and diseases, or improved plant varieties usually also improves nitrogen efficiency. Rather challenging is the integration of inhomogeneous Nr inputs like manure, biological fixation, atmospheric deposition or crop residues, as their amount and

[4] 108 Tg Nr as fertilizers (IFA 2013), 60 Tg Nr from biological fixation (Herridge, Peoples, and Boddey 2008), 30 Tg Nr from depletion of natural soils (Vitousek and Matson 1993), and about 50 Tg Nr of atmospheric deposition on croplands and pastures, stemming originally from combustion or volatilized Nr (Sutton, Howard, et al. 2013).

chemical composition as well as their spatial and temporal availability is difficult to estimate for farmers. (see e.g. IFA 2007; Snyder and others 2009)

The level of targeted fertilization efficiency is chosen by the farmer through his management efforts. Measuring, multiple applications, spreader maintenance or education are connected to labour and capital costs that de-incentivize efficient fertilization. On the other side, there are also incentives for efficient management: Nr is a costly fertilizer, and some of the environmental impacts of Nr pollution impede the ecosystem services farmers are using themselves. Moreover, farmers may want to label their products as sustainably produced or have a personal preference to protect the environment. (Bodirsky 2009)

However, market and state failure lead to too low fertilization efficiencies. As a large part of the impacts of Nr pollution also damage other people than the farmer, the private costs of farmers' behaviour do not match the social costs – a classic case of a negative externality (Bodirsky 2009). Moreover, agriculture is subject to large subsidies that distort markets. In particular, many states subsidize the use of inorganic fertilizers (Liao 2008; Banful 2011) with the aim to improve food security, while lowering private costs of fertilization and incentives for efficiency.

2.3.2 Livestock feeding and animal waste management

Large Nr losses stem from the inefficient transformation of plant-based into animal-based calories and proteins. For gross energy, transformation efficiencies range between 4-5% for beef meet and 20-25 % for milk (Smil 2000). In the case of Nr, efficiencies range between 5-8% efficiency for beef up to 30-40% for milk (Smil 2002). Therefore, while animal-based products make up only one fifth of global caloric food demand, their production consumes one third of the valuable global cereal harvest (FAOSTAT 2013), almost all proteins from oilcrops (in the form of oilcakes) (Bodirsky et al 2012), roughage that is grown on one tenth of cropland area as well as a number of other crops (FAOSTAT 2013), and crop residues like straw (Smil 1999). Finally, ruminants also receive feed from grazing, with pasture occupying two thirds of agricultural land (FAOSTAT 2013).

The induced Nr losses of inefficient livestock production occur mostly down-stream on the fields where feed-crops are cultivated. But also the nutrients in animal feed which are excreted as manure are a source of Nr pollution. A large fraction of manure Nr is not recycled to fields, but lost through volatilisation and denitrification in manure management, and when applied on fields. There are large differences in the resource-efficiency of different livestock management systems (Herrero et al. 2010; Herrero et al. 2012): Mixed livestock-crop systems are rather resource-efficient, as they can make better use of marginal crop products as livestock feed, and as they efficiently recycle the nutrients in manure on their cropland. In contrast, landless industrial management systems require high-quality feed products that underwent a long supply-chain with losses. Also, landless systems have problems to get rid of their nutrient surplus, as they tend to be geographically concentrated close to harbours and markets instead of cropland.

Different mitigation options exist to improve the nutrient efficiency of livestock production. Better breeds, veterinary medicine and animal housing can increase the efficiency of nutrient uptake of animals. The composition of animal diets can be fine-tuned to reduce the excretion of surplus nutrients. Also feed additives like certain, scarce amino acids can reduce feed requirements and Nr excretion. NH3 volatilisation and denitrification in animal housing can be mitigated for instance by acidifying manures, by drying manure or by cleaning ventilation air in stables. Losses in storage and

transport of manure can be lowered by covering storage facilities or by shortening the time of storage. The quality of manures can be improved and adjusted to field requirements before application on the fields, for example amending or separating nutrients, by adding chemicals that reduce Nr losses, or by pelletizing manures into uniform and transportable organic fertilizers. Finally, the spreading technique and the general recommendations of efficient fertilization (see section 2.3.1) also influence the recycling efficiency of manures. (Sutton, Howard, et al. 2013)

Similar to fertilization efficiency, there are diverse reasons for the current efficiency in livestock production. Efficiency gains translate into lower costs for farmers if they can save feedstock and substitute inorganic fertilizers by manure. However, these efficiency improvements are connected to management costs as well as investment costs into breeding, manure management facilities, knowledge and education. Again, the existence of externalities distorts incentives. Livestock farmers (in particular landless producers) bear in comparison to crop farmers an even smaller share of their environmental damages themselves as they depend less on ecosystem services of their direct environment. Also the state distorts incentives by subsidizing feedstock and inorganic fertilizers, or by actively fostering livestock production, for example through fixed prices (de Wit, Westra, and Nell 1996).

2.3.3 Food demand: intake, hunger, overconsumption, food waste & sewage

While food is one of the most basic needs for humans, food is not efficiently allocated in the sense that actual food demand (defined here as the amount of food purchased by consumer households) does not match well with food requirements or healthy levels of consumption. As will be pointed out in more detail in this section, a large fraction of human population is undersupplied with food, while the food demand of other parts of the population exceeds requirements because of unhealthy overconsumption and food waste on the household level. Overconsumption and food waste do not only unnecessarily increase food production, but also increase the Nr losses in sewage systems and on landfills.

The food energy requirements to sustain the basic metabolism are 1200-1600 kilocalories (kcal) per day for adult females, and 1400-2000 kcal per day for adult males (Smil 2000). Taking into account moderate physical activity, energy requirements rise to 2000-2600 and 2500-3600 kcal respectively (Smil 2000). Caloric requirements also differ by age, with an adult requiring almost twice the calories of a three year old child (FAO 2004). Therefore, the global average intake lies probably around 2000-2100 kcal per capita per day (Bender 1994; Smil 2000). Beside energy, humans also require nutrients like Nr. Humans are not only incapable of absorbing Nr from the air; they are also incapable of synthesizing all amino acids themselves. Certain essential amino acids have to be taken up by the food. A diet should therefore not only provide enough Nr (about 3 kg per person per year (de Vries et al. 2013)), but also a balanced composition of amino acids (WHO, FAO, and UNU 2007). However, the diet of large parts of human population diverges from these dietary recommendations.

On the one side, more than 800 million people in the developing world are suffering from hunger and undernourishment (FAO 2004). While most undernourished live in the least developed countries (Figure 1), even in high-income countries like the United States of America, 11% of the households are not food-secure for the whole year, and 3.5% even suffer from hunger (Nord and Andrews 2003). Undernourishment often overlaps with vitamin and mineral deficiencies, which concern worldwide almost 2 billion people (FAO 2004). Consequences are an increased risk of dying in

infancy, stunting of physical and cognitive growth, lower immunity, giving birth to low-weight children, and lower working power (FAO 2004). The World Health Organisation (WHO 2009) estimates that childhood and maternal undernutrition causes the loss of 192 million disability adjusted life years (DALYs).

Major causes of mass undernourishment include failed harvests, armed conflicts, economic crisis and refugees (FAO 2004), which often coincide with a low economic and social capacity for adaptation. In developed countries, food insecurity is mostly found in poor and low-income households, in particular in families headed by single women or men (Nord and Andrews 2003).

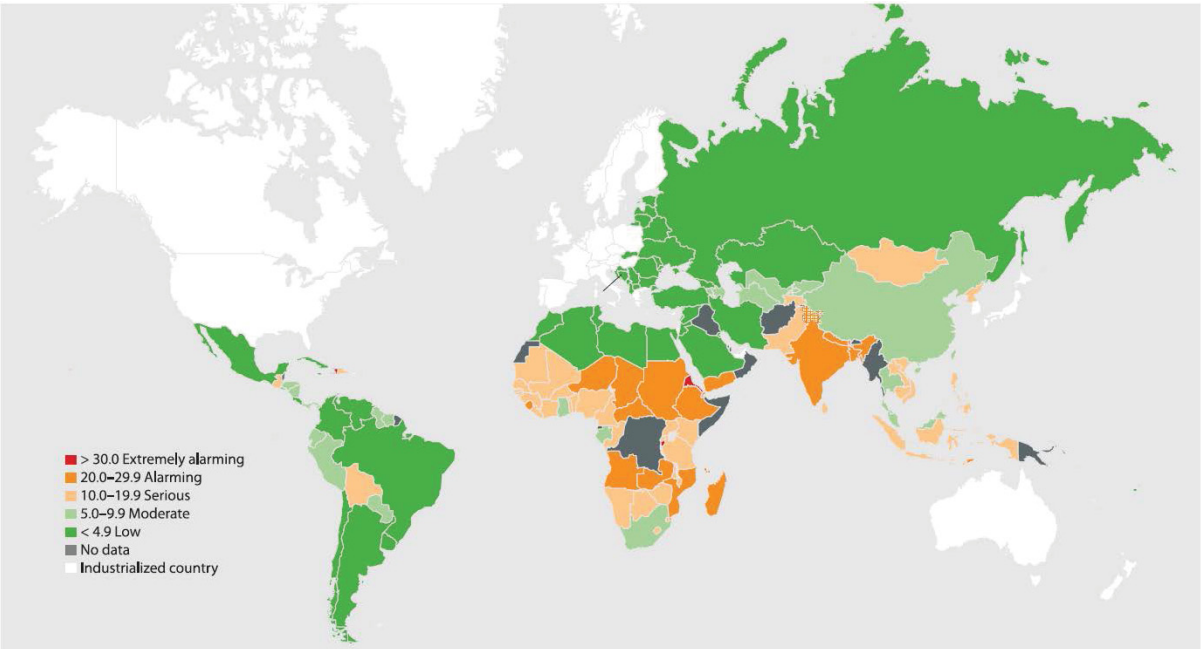

Figure 1: Global hunger index, indicating the share of population undernourished. The map was published in Wheeler and Braun (2013), and is based on data of Grebmer et al. (2012).

On the other side, food demand substantially exceeds intake recommendations for a large part of global population (FAOSTAT 2013; Smil 2000) due to unhealthy overconsumption and household food waste.

Food intake often exceeds the recommended levels of caloric intake for the degree of physical activity. Overconsumption is often supplemented by a shift in dietary structure from dietary fibre towards animal products, oils, and sugar (Drewnowski and Popkin 1997). In combination, this leads globally to an estimated 1.4 billion overweight adults, of which 500 million are obese (WHO 2013), with an increasing share also stemming from developing countries (FAO 2004). Obesity, unbalanced diets, and low physical activity lead to a number of health problems like diabetes, cardio-vascular diseases, musculoskeletal disorders, and cancer (FAO 2004; WHO 2013). The World Health Organisation (WHO 2009) estimates the casualties at 212 million disability adjusted life years (DALYs), even exceeding the casualties from undernutrition.

Overnutrition and imbalanced diets are strongly connected to modern and urbanized lifestyles. Low physical activity at work, increasing integration into a specialized labour market with little time for

food preparation and consumption, distortion of natural food preferences by marketing, and the cheap availability of highly refined products like livestock products, sodas and sweets are some contributing factors (Drewnowski and Popkin 1997; Popkin 1993).

Overconsumption is only responsible for a minor part of the overly high food demand. Taking into account also overconsumption, plausible daily per-capita intake in developed countries lies most likely between 2000 and 2300 kilocalories, and certainly does not exceed 2500 kilocalories (Smil 2000). However, the average food demand in OECD countries of 3444 kilocalories (FAOSTAT 2013) by far exceeds these plausible intakes. The reason for this is that 30-40% of purchased food in developed countries is estimated to be thrown away on household level (Gustavsson, Cedersberg, and Sonesson 2011).

Similarly to overconsumption, high food waste in household can be connected to the socio-economic situation in modern urbanized societies (Parfitt, Barthel, and Macnaughton 2010; Kantor et al. 1997; Smil 2000): The number of single person households is rising, the labour market captures higher shares of the population, so food is increasingly bought pre-prepared or consumed outside the house. This leads to less precision in the portioning and high leftovers. Higher incomes also lead to a lower relative valuation of leftovers. Moreover, consumer's knowledge about the preparation and the perishability of food is decreasing, such that flawless food is thrown away. Finally, the externalities and state interventions into agricultural production also lower food prices and incentivize careless food demand patterns.

Mitigation in the food demand sector has to be cautious not to aggravate hunger and malnutrition. Instead it should concentrate on the reduction of food waste and unhealthy overconsumption. By making parts of the production redundant, this would reduce down-stream Nr pollution through the whole food supply chain. But also post-consumer losses (Nr lost as food waste on land-fills or as slurry in sewage systems and rivers), offers potentials to improve efficiency. Currently, only a small share of Nr in sewage is recycled to agricultural land, while most Nr is denitrified in sewage plants or enters surface waters (Morée et al. 2013). Similarly, only about one fifth of organic waste is composted, and it is unclear how much of this composted waste is actually returned to agricultural fields (Hoornweg and Bhada-Tata 2012). Higher recycling shares of sewage waste and organic wastes would be therefore a further mitigation measure to complement reduced food demand.

2.3.4 The share of livestock products within the diet

Also the consumption of livestock products diverges from requirements and health recommendations. In general, vegetarian diets can settle all dietary requirements and have positive health effects (Smil 2000; Key, Appleby, and Rosell 2007). Vegan diets can also be balanced as long as they are complemented with supplements of certain nutrients like vitamin B12 that are scarce in vegan food (Key, Appleby, and Rosell 2007). In few cases, an increase of consumption of livestock products (but also a more diverse vegetarian diet) could be beneficial for nutrition, for example in cassava dominated diets in Africa (Smil 2000). In contrast, the high consumption of animal products in large parts of the world is linked to a number of health problems (Cross et al. 2007; Corpet 2011; Fraser 1999; Kelemen et al. 2005; Sinha et al. 2009; WCRG and AICR 2007). For example, diets containing more than 25% of calories from animal products are likely to increase obesity (Smil 2000).

Livestock products make up 18% of calories (39% of proteins) within global diets, with many developed countries receiving almost 30% of their calories (60% of proteins) from meat, milk, eggs

and fish (FAOSTAT 2013). A reduction of this share would displace the inefficient production chain of livestock products, requiring a multitude of plant calories as feed (see section 2.3.2), by more efficiently produced plant products. While plant proteins are less digestible than animal proteins, vegetarian diets still score substantially higher Nr efficiencies than carnivorous diets if the whole supply chain is regarded (Smil 2000). Similarly to a reduction of total livestock consumption, a consumption shift from less efficiently produced livestock products (e.g. ruminant meat) towards high efficient products (e.g. milk) could reduce feed requirements. While there is a clear ranking of caloric transformation efficiencies between animals, this does not necessarily mean higher efficiency from a system perspective. Pigs can convert a broad range of processing and household by-products that would otherwise be wasted. Similarly, only the complex digestive systems of ruminants are able to effectively digest pasture and straw. Finally, couple products exist (ruminant milk and meat, bird eggs and meat) which make a full substitution impossible. Within these complex limitations, consumption shifts within livestock products can still contribute substantially to mitigation (Wirsenius, Azar, and Berndes 2010).

The consumption of livestock products is determined by a large number of cultural, religious, genetic, economic and social factors. Pig consumption is traditionally low in Muslim and Jewish cultures, and cow meat is not eaten among Hindus. Milk consumption in contrast is limited by lactose intolerance, a widespread phenomenon outside of Europe and North America (Smil 2000). Economic drivers of livestock consumption are rising incomes, as livestock products are often seen as superior goods. Moreover, the modernization of food industry, various market failures and state interventions lead to low prices for livestock products (see section 2.3.2) and high consumption. Livestock consumption is also shaped by major social transitions. Urbanisation leads to a convergence of diets around the world (Regmi and Unnevehr 2005). Increasing urbanization, labour force integration and marketing lead to a higher consumption of pre-prepared meals and outside eating (Drewnowski and Popkin 1997; Popkin 1993; Smil 2000), which is often connected to a high share of livestock products. Finally, health awareness or environmental and ethical concerns may lead to a reduction of livestock consumption, or even to vegetarianism (Fox and Ward 2008).

3 Research Questions

As was highlighted in the background section, the nitrogen cycle is intertwined with human welfare in complex ways. It is therefore important to acquire knowledge about the current state of the nitrogen cycle, to understand the dynamics like food demand that determine its development, and to explore quantitatively how the nitrogen cycle might evolve in the future.

The topic of this dissertation is therefore:

How will future food demand alter the agricultural nitrogen cycle and Nr pollution?

Five research questions stand in the centre of this study:

3.1 What are plausible scenarios for global food demand in the 21st century?

(Chapters II,III)

The rising food demand, driven by population growth and more affluent diets, has been the major driver of the land-use sector in the past (Huber et al. 2014). Understanding the dynamics of food

demand is therefore crucial to any projection of the land-use sector.
This thesis investigates how the main dynamics that determine food demand can be represented in long-term models, and how the long-term perspectives of global food demand look alike. Within food demand, the plant-based and animal-based calories shall be distinguished, as the production of animal-based products is more resource intensive. Of similar interest is the breaking up of food demand into food intake and food waste. Analysing and comparing projections of different models gives insights about the overall pressure on the agricultural sector and the environment in the future, as well as insights on questions of food security and overconsumption.

3.2 What is the current state of the agricultural nitrogen cycle?

(Chapter IV)

The magnitude of major nutrient flows in the agricultural sector is still rather uncertain even for the present state. Only few Nr flows like inorganic fertilizer production are accounted for in official statistics and can be determined comparatively precise with an error +-10% (Fowler et al. 2013). Most nitrogen flows instead can only be estimated indirectly, and have therefore a much higher uncertainty, ranging often in the order of +-50% (Fowler et al. 2013), or in the case of emission parameters even being uncertain by one order of magnitude (Eggleston et al. 2006). Some important Nr flows have not even been quantified at all.
This study therefore provides a new set of estimates for the major Nr flows in the agricultural sector. By proposing new methods to estimate Nr flows and by providing different sets of parameters, this study improves the understanding of the nitrogen cycle.

3.3 How will the nitrogen cycle evolve into the future?

(Chapters IV,VII)

Understanding the current and future dynamics of the agricultural nitrogen cycle is necessary to estimate nitrogen pollution and its impact on the Earth system as well as on human welfare. As the agricultural nitrogen cycle is driven by human activity, projections of the agricultural nitrogen cycle require both an understanding of the major socio-economic developments of the agricultural sector as well as their biophysical consequences on the nitrogen cycle.
In a first step, scenarios on the future trajectories of the agricultural sector are developed, which consider the dynamics of dietary transition described above, but also account for the emerging demand for bioenergy crops, the development of agricultural technology on the field and in livestock management, the economic choices made in regard to the allocation of natural and financial resources, as well as policies regarding trade liberalisation and land-use planning.
In a second step, scenarios for the nitrogen cycle are derived which are consistent with the above dynamics in the agricultural sector. This concerns how agricultural activity shapes nitrogen flows, e.g. through the harvest of crops, but also how nitrogen flows determine agricultural activity, e.g. how the availability of organic fertilizers influences the fertilizer management of farmers. The scenarios shall include trajectories for all major Nr flows in the agricultural sector as described in section 3.2.

3.4 Which additional pressure on the nitrogen cycle might evolve from the production of 2nd generation bioenergy?

(Chapter V)

Bioenergy will become a competitive substitute for fossil fuels in the near future, especially under climate change mitigation (IPCC 2012). Depending on the scenario, bioenergy crops may even become the dominant driver of future change in the agricultural sector (Neher 2013).
Therefore this thesis investigates the effects that bioenergy production, additional to food production, could have on the nitrogen cycle and nitrogen pollution. The study will thereby focus on so called 2nd generation bioenergy crops like fast-growing grasses and trees which have promising attributes in regard to crop yields and nutrient efficiency.

3.5 How would the adoption of nitrogen mitigation measures change the nitrogen cycle and Nr pollution?

(Chapters V,VI)

Sutton et al. (2013) collected a number of mitigation measures that could be adopted to reduce nitrogen pollution. In regard to agriculture, these measures target fertilization efficiency, livestock feeding, animal waste management, human consumption patterns and sewage. This study investigates how these measures would change Nr flows and by how much they could reduce Nr pollution. In the end stand the questions on how high the residual pollution would be after implementing all mitigation measures, and whether this residual pollution would be below critical environmental thresholds.

4 Methods overview

4.1 Research Approach

For this dissertation, the approach of computer model simulations is chosen. The aim of modelling exercises is to create simplified images of reality that are able to capture the important dynamics determining the outcomes relevant for the respective research question. The process of mathematical formalization renders explicitly the underlying processes, makes the results reproducible, and eventually reveals inconsistent or conflicting assumptions. The use of computational power of modern supercomputers supplements our mental capabilities and allows more complex models and faster estimates. (Forrester 1994; Costanza and Ruth 1994; Verburg 2004)

Answering the research questions for this thesis requires a model that combines the scientific disciplines of social and natural sciences. As indicated by the term “Anthropocene” (Steffen 2004), the current state of Earth system processes is strongly shaped by human behaviour. In particular the current and future state of the nitrogen cycle and the atmospheric concentration of greenhouse gases cannot be simulated without taking into account the human component (Fowler et al. 2013). Vice versa, the world’s ecosystem services are of precious economic value, in particular to the agricultural sector (Costanza et al. 1997; Millennium Ecosystem Assessment 2005), and the impact of pollution on human welfare can only be estimated accurately if the underlying physical processes are taken into account. The methodological approach of this dissertation can be best located into the discipline of ecological economics. This discipline fusions social and natural sciences and emanates from the thought that “human economy is embedded in nature, and economic processes are also always natural processes in the sense that they can be seen as biological, physical and chemical processes and transformations” (Ropke 2004, p. 312). While there is a seamless transition between the disciplines ecological economics and environmental economics, ecological economics tends to represent biophysical processes more explicit by looking at the laws of natural sciences. As

it will be explained in the following, this fusion of economics and biophysics is well reflected in the land-use model MAgPIE (Model of Agricultural Production and its Impact on the Environment) that is central to this analysis.

It is a global land-use model that can be used to create long-term projections of the land-use sector with a special focus on agriculture. (Lotze-Campen et al. 2008; Lotze-Campen et al. 2009; Popp, Lotze-Campen, and Bodirsky 2010; Popp, Dietrich, et al. 2011; Popp, Lotze-Campen, et al. 2011; Bodirsky et al. 2012; Dietrich et al. 2012; Popp et al. 2012; Dietrich et al. 2012; Schmitz et al. 2012; Dietrich et al. 2014; Krause et al. 2013; Schmitz et al. 2013)
On the one hand, MAgPIE is a socio-economic model. It estimates land-use activities based on a partial equilibrium approach (even though with inelastic demand), representing important social dynamics like the optimization of trade and global production patterns. For this thesis, MAgPIE is extended by a food demand model to represent the dynamics of the dietary transition with increasing economic development. Using an econometric approach, it estimates global food demand, the share of animal products within the diet, and global food waste. The approach requires only very little data and can be adjusted to varying storylines of scenarios, which makes it very suitable for the creation of multiple consistent scenarios (Chapters III,IV).
On the other hand, MAgPIE is simultaneously a spatially explicit biophysical model, representing physical quantities like land, water, biomass, nutrients, and emissions. For this thesis, MAgPIE is extended by a mass-balance module for nutrient flows (Chapters V, VI, VII). This module captures all major flows from Nr entering the agricultural sector (e.g. as fertilizer), its transformations within the sector (e.g. from feed to animal protein), up to the point where it leaves the agricultural sector to enter the environment (e.g. as N_2O emissions). A mass-balance approach ensures the consistency of the flows.
Within MAgPIE, physical and social processes do not coexist, but are intertwined and influence each other. Agricultural goods are simultaneously economic values and physical quantities. Physical scarcities of land, water or nutrients translate into economic prices. Thereby, nutrient flows are not calculated ex-post as a consequence of human behaviour, but are taken into account by the economic processes. This endogenous implementation is for instance crucial when the taxation of Nr emissions is simulated, as Nr flows then strongly alter the incentive structure of economic actors (Bodirsky 2009).

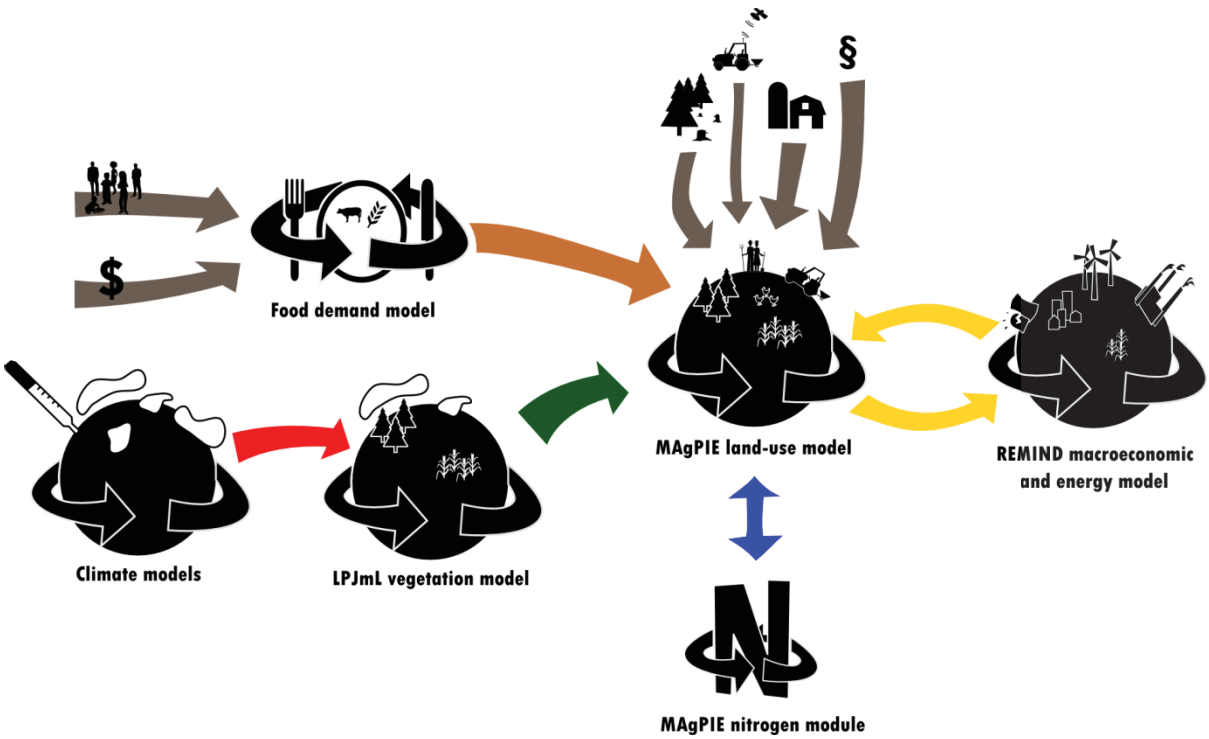

Figure 2: Parts of the model universe developed at the Potsdam Institute for Climate Impact Research. Models are linked one way (single arrows), by iterative soft-coupling (double arrows), or by integrated hard-coupling (arrow with two arrowheads).

MAgPIE is embedded in a network of models that are developed at the Potsdam Institute for Climate Impact Research (Figure 2). This allows a better integration of information flows, and higher consistency of the results. In this network, climate models provide information on temperature and precipitation to the global dynamic vegetation model LPJmL (Lund-Potsdam-Jena managed Land model)(Sitch et al. 2003; Bondeau et al. 2007). LPJmL delivers information on carbon stocks for different vegetation types, water availability and crop productivity to the land-use model MAgPIE at a spatial resolution of 0.5 degree (Dietrich, Popp, and Lotze-Campen 2013). Information on bioenergy demand and agricultural GHG emissions from MAgPIE are exchanged against information on bioenergy and GHG prices from the macroeconomic and energy model Remind in the REMIND-MAgPIE coupling framework (Klein et al. 2013). MAgPIE furthermore receives estimates for the demand of crop and livestock products that are estimated by a food demand model based on population and income projections (Chapters II, III of this dissertation). Directly integrated into the model is a module to estimate nitrogen flows (Chapter IV of this dissertation), as well as various other optional modules to estimate for example CH_4 and CO_2 greenhouse gases (Popp, Lotze-Campen, and Bodirsky 2010; Popp et al. 2012), water use (Schmitz et al. 2013) or crop yield growth (Dietrich et al. 2014). Finally, a number of parameters like forest protection policy or the degree of globalization are independent scenario drivers for the MAgPIE model. The drivers and models are harmonized based on qualitative storylines (Chapter II, IV of this dissertation).

4.2 Uncertainty analysis and validation

Models are only abstractions of reality (Müller 2011), which simplify the dynamics of the regarded system. According to Forrester (1994), "… there is no possible proof of the validity of any model, whether they are mental or computer models. Models are to be judged by their comparative usefulness. Assumptions about structure and policies should be compared with any available

information. Computer simulation results should be compared with behaviour in the real system being represented. Discrepancies lead to improving both mental and computer models."

To inspect the quality of the utilized models and their results, and to provide insights into the model uncertainty, different methods are used in this dissertation. A first set of approaches targets the uncertainty that may enter the model through the input parameterization.

- A Monte Carlo method is used for an uncertainty analysis in Chapter IV. It estimates the impact of emission factor uncertainty on the estimated emissions. Unfortunately, such a sensitivity analysis is not suitable to be applied to the whole sets of input parameters of a model like MAgPIE. Firstly, the uncertainty ranges of many input parameters are unknown, secondly many input parameters cannot be assumed to be independent, and thirdly the computational requirements of a model containing so many parameters would be inappropriate costly.
- Where sensitivity analysis is not feasible, a scenario analysis can help to provide insights into the reaction of model dynamics on changing input parameters, even though it can only provide a lower estimate for the uncertainty range. Typically, parameters are chosen as scenario parameters if their uncertainty as well as their potential impact on model results is high. Scenario analysis are applied in chapters II, III, IV, V, VI and VII.
- Storylines are a qualitative tool for scenario design. They are used to consider complex correlations of parameters in a model, which exist in reality but which are not represented in a model. By excluding scenario parameterisations that have a low probability to coexist, they help to narrow down the uncertainty range. A positive side-effect is that storylines facilitate communication and the consistent use of output data. Storylines are used for the scenario design of chapters II, III, IV and VII.

A second set of approaches targets the uncertainty stemming from a simplified or a possibly incorrect representation of processes within the model.

- In the food demand model (chapter II), functional relations in the regression analysis are adapted to the assumptions of the underlying storyline. Therewith, an example is provided that scenarios can not only be created by using different input data for a model, but also by exchanging representations of processes.
- Different modelling approaches for the same process are used. In chapter III, different food demand modelling approaches and their outputs are presented and compared. Similarly, in the discussion of chapter IV, different approaches to estimate Nr flows are compared. The approaches have their individual strength and weaknesses, and a comparison helps to identify models appropriate for individual research questions. Finally, chapters VI and VII provide different approaches to estimate the impact of supply-side mitigation measures.

A third set of approaches is used to validate our results.

- A first validation approach compares model results to historical data. On the one hand, it is analysed whether our results match the historical trend and the processes are identified that are responsible for eventual deviations (chapters II, III, IV). On the other hand, the model is run for a period in the past, and it is subsequently evaluated how well the historical dynamics are matched (chapter II and IV).

- Our model results are also compared to the projections of other studies with methodologies ranging from similar modelling approaches to statistical models or expert estimates (chapter II, III, IV, VI).

5 Structure of the dissertation

The main part of this dissertation consists of 6 articles that have been published in scientific journals or are currently under review. The articles are the result of a scientific cooperation between various authors, whose individual contributions are listed in the Statement of Contribution at the end of this dissertation. The 6 articles each form self-contained chapters, having their own references and their own layout. Nevertheless, the articles are interlinked concerning the topic, method, data and conclusions. Please also note that the articles were not written and published in the same chronological order as they appear in this dissertation. Chapter 4 has therefore a more sophisticated methodology than chapter V and VI.

Chapter II develops a plain and transparent method to project global food demand for the 21st century, using only GDP and population projections as input data. The demand model is based on regressions, and allows creating scenarios with different functional types depending on the assumptions on the underlying processes. The method is exemplarily applied to the SRES storylines (Nakicenovic et al. 2000) and the results are compared to other studies.

Chapter III is devoted to a comparison of ten models of global food demand, also featuring the model described in chapter II. First, the different methodological approaches of the models are presented and discussed. Subsequently, the parameterisation and the model projections are compared and interpreted.

Chapter IV introduces the MAgPIE model. The food demand scenarios from chapter II are combined with further major land-use drivers and are fed into MAgPIE to derive scenarios for the agricultural sector in the 21st century. Moreover, MAgPIE was extended by a nitrogen budget module, which estimates the current state and the future development of the agricultural nitrogen cycle. This module is described qualitatively in the main text, as well as mathematically and in more detail in the Appendix of this chapter.

Chapter V analyses how the large-scale cultivation of 2nd generation bioenergy crops like fast-growing grasses and trees would change nitrogen fertilizer requirements and N_2O emissions. Different scenarios of bioenergy demand and nutrient uptake efficiencies are compared.

Chapter VI simulates mitigation scenarios for the non-CO_2 emissions methane (CH_4) and nitrous oxide (N_2O). The effects of supply-side and demand-side mitigation options as well as their joint implementation are compared.

Chapter VII uses a slightly extended version of the model described in Chapter IV to create mitigation scenarios for Nr pollution. We simulate the impact of different Nr mitigation options as described by Sutton et al. (2013). These mitigation options include more efficient nutrient management on the producer side, but also an alteration of diets and food consumption behaviour on the food demand side.

Chapter VIII summarizes and synthesizes the results of the individual chapters. Finally, a perspective is given on how the existing model of MAgPIE could be extended into an integrated model of agriculture, nitrogen and human welfare, and how the integrated view on nitrogen and human welfare could change the agricultural sector.

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Chapter II

Global food demand scenarios for the 21st century

Benjamin Leon Bodirsky • Susanne Rolinski • Anne Biewald • Isabelle Weindl • Alexander Popp • Hermann Lotze-Campen

Content of chapter II: Global food demand scenarios for the 21st century

Global food demand scenarios for the 21st century

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Abstract Long-term food demand scenarios are an important tool for studying global food security and for analysing the environmental impacts of agriculture. We provide a simple and transparent method to create scenarios for future plant-based and animal-based calorie demand, using time-dependent regression models between calorie demand and income. The scenarios can be customized to a specific storyline by using different input data for gross domestic product (GDP) and population projections and by assuming different functional forms of the regressions.

Our results confirm that total calorie demand increases with income, but we also found a non-income related positive time-trend. The share of animal-based calories is estimated to rise strongly with income for low-income groups. For high income groups, two ambiguous relations between income and the share of animal-based products are consistent with historical data: First, a positive relationship with a strong negative time-trend and second a negative relation with a slight negative time-trend. The fits of our regressions are highly significant and our results compare well to other food demand estimates.

The method is exemplarily used to construct four food demand scenarios until the year 2100 based on the storylines of the IPCC Special Report on Emissions Scenarios (SRES). We find in all scenarios a strong increase of global food demand until 2050 with an increasing share of animal-based products, especially in developing countries. At the same time, the results indicate that undernourishment is likely to persist for the coming decades.

Keywords Food demand · animal-based calories · food security · food waste · scenarios · SRES

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1 Introduction

Over the last 50 years, global food demand approximately trippled (FAOSTAT, 2011). This rapid growth was caused on the one hand by the doubling of world population from about 3 to more than 6 billion people and on the other hand increased per-capita demand due to rising living standards. In this article, we define food demand as equivalent to food supply according to FAOSTAT (2011). It describes the average availability of food in households in kcal capita^{-1} d^{-1}, comprising both food intake and non-eaten household food waste. Detecting a basic rule of microeconomics, the German statistician Engel observed already in the middle of the 19th century a positive correlation between per capita food demand and income that levels out for high incomes, the so called "Engel's Law" (Engel, 1857). Yet beside income, per capita demand is also influenced by other factors such as climate, age and gender composition of the population, food prices, development of the food industry, globalisation and openness to global markets, the degree of urbanisation, or changes in the role of women in society (Drewnowski & Popkin, 1997; Smil, 2000; Popkin, 1993).

At the same time, more and more animal-based products like meat, milk, eggs and fish are being consumed. In the last five decades, the global population-weighted average share of animal-based products rose from 15.4 to 17.7 % (FAOSTAT, 2011, years 1961-2009). A closer look however reveals, that this increase can mainly be attributed to rising consumption of animal-based products in developing and emerging economies, while in developed countries this share stagnated or even decreased in the last decades (FAOSTAT, 2011). As with total food demand, income is a major driver for the share of animal-based products, but also the development of the livestock industry, state interventions, food prices, cultural preferences, globalisation and openness to global markets, and health concerns are determining factors (Drewnowski & Popkin, 1997; Smil, 2000; Popkin, 1993).

Food demand can be used as a simple indicator for food security, even though such an index has to be interpreted with caution. According to FAO's definition (FAO, 1996), food security can be analyzed with respect to (1) food availability, (2) food stability over time, (3) access to food, including the potential of individuals to purchase food, and (4) utilization of food to create an adequate and healthy diet. Food demand as analyzed in this study is mostly an indicator for food availability, while one has to bear in mind that the FAO estimates do not cover food provision from hunting and gathering of wild products, which may supply a significant share of food calories to households in developing countries (see A.1). The information on food stability over time and access to food is partly lost as FAO estimates are averaged over one year and over the population of one country. For example, in the United States of America about 11% of households are not food secure for the whole year, and 3.5% suffer from hunger (Nord & Andrews, 2003), even though the per-capita demand in the United States is with more than 3800 kcal capita^{-1} d^{-1} one of the highest in the world. Hence, food wastage and hunger can coexist due to distributional inequality. Estimating the prevalence of undernourishment based on distribution corrected food availability as done by FAO (2004) was criticized as being unreliable as it is a top-down approach that has to rely on poor statistical data (Svedberg, 1999). Instead, Svedberg (1999) proposes to use anthropometric bottom-up measurements like the prevalence of stunting and underweight. Unfortunately, global data on food security is still deficient and incomplete (Wheeler &

Braun, 2013). One of the few available global datasets that includes anthropometric measurements is the Global Hunger Index (GHI) (von Grebmer et al., 2012) which is composed to equal shares of the prevalence of child underweight, child mortality and FAO (2004)'s top-down estimates. Despite this diverging approach, the GHI shows surprisingly strong correlation (Pearsons correlation coefficient of -0.84) with average per-capita demand per country (FAOSTAT, 2011). Within its limitations, food demand can therefore be considered as a useful food security indicator.

Scenarios of future food demand are an important tool to develop strategies for achieving the goal of global food security. Scenarios are used when uncertainty is too high to allow for predictions or forecasts. Instead, they are alternative images of the future that "enhance our understanding of how systems behave, evolve and interact" (Nakicenovic et al., 2000). They can be used to explore possible futures and to localize the regions where most effort is needed to combat hunger. Furthermore, they help to distinguish periods in which the problems can be expected to persist, such that strategies can be adapted accordingly.

Moreover, food demand scenarios are also used in different contexts, for example to estimate future cropland expansion (Wirsenius, 2000), agricultural greenhouse gas emissions (Popp et al., 2010; Stehfest et al., 2009), trade, technology and water prices (Schmitz et al., 2013), agricultural intensification (Trewavas, 2001), and the use of pesticides (Carvalho, 2006). For many of these studies, the amount of animal-based calories may play a special role, as their production requires a multitude of plant calories and puts a high pressure on the land-use system and the environment (Steinfeld et al., 2006; Tilman et al., 2011).

The rising computing power allows to explore more and increasingly diverse pathways into the future. Hereby, it becomes important that scenarios are easily adaptable to changing storylines to account for many possible future worlds. Additionally, flexibility regarding the aggregation of world regions and the possibility to expand the time horizon are desired.

A number of studies already estimate future food demand (Alexandratos & Bruinsma, 2012; Alexandratos et al., 2006; Valin et al., 2010; Dorin et al., 2011; Kruse, 2010). However, these approaches are not easily reproducible and hard to adapt to the specific needs of new scenario settings or new research questions. Many studies provide only data for a single scenario and cover a time-horizon until 2050. While scenarios beyond 2050 are inevitably connected to much larger uncertainty, they are strongly needed for climate change impact analysis (Nakicenovic et al., 2000).

Therefore, we provide a transparent method for creating specific food demand scenarios for total and animal-based calories, requiring only population and income projections as input, while no information on the food supply side is needed. The scenarios that can be created are flexible regarding the time horizon and provide information on the national level, which can be aggregated to regions of any size and location. Our method is exemplarily applied to the storylines of the SRES scenarios (Nakicenovic et al., 2000) to create country-based projections until the year 2100. The SRES storylines are broadly used in the literature and were further elaborated in the Millennium Ecosystem Assessment (2003, 2005). Beyond, our method could be applied to many other scenario projects: To the climate-related global IPCC storylines SA90 (IPCC, 1990), IS92 (Leggett et al., 1992) or the upcoming SSP (O'Neill et al., 2013) and SPA storylines (Kriegler et al., 2014);

to general long-term scenarios for sustainability research like the Club of Rome scenarios (Randers, 2012) or the Millennium Ecosystem Assessment (2003, 2005); or to scenarios on agriculture and land-use that are focussed on specific world regions like Öborn et al. (2011). They could also help as a tool for participatory scenario-building processes like CCAFS (Vervoort et al., accepted).

2 Material and Methods

Our approach uses country-specific historical data to fit regressions of food demand on income over time which we apply to scenarios of future income and population.

2.1 Data sources

We use data on historical food demand provided by the United Nations Organisation for Food and Agriculture (FAO) in the FAOSTAT database (Table 1). Demand is given in kcal capita^{-1} d^{-1} per country from 1961 to 2007 as total calorie (C_T) and animal-based calorie demand (C_L). The data include a wide range of unprocessed and processed agricultural and fishery products intended for food consumption.

Table 1 Data sources. ItC: Item code, EC: Element code, IC: Indicator code. Download of FAO data from http://faostat.fao.org on 27.10.2010 and from Worldbank from http://data.worldbank.org on 13.09.2011.

Value	Unit	Years	Codes	Sources
C_T	kcal capita^{-1} d^{-1}	1961 – 2007	ItC: 2901, EC: 664	FAOSTAT
C_L	kcal capita^{-1} d^{-1}	1961 – 2007	ItC: 2941, EC: 664	FAOSTAT
I	US$\$_{2005}$ capita^{-1} a^{-1}	1960 – 2007	IC: NY.GDP.PCAP.KD	WDI
P	capita	1960 – 2007	IC: SP.POP.TOTL	WDI
$I_{t,c}^{Proj}$	US$\$_{1990}$ capita^{-1} a^{-1}	1990 – 2100	A1, A2, B1, B2	CIESIN
$P_{t,c}^{Proj}$	capita	1990 – 2100	A1, A2, B1, B2	CIESIN

Historical data on population and per-capita income are taken from the World Bank in the World Development Indicator (WDI) database (Table 1, WORLDBANK, 2011). Population (P_c) is given in capita per country c and income values (I_c) [gdp capita^{-1} year^{-1}] in constant US$\$_{2000}$ from 1961 to 2007, based on market exchange rate (MER). For income (I) and calorie demand (C_T,C_L) there are 5889 complete data pairs from 162 countries.

Projections of per capita income ($I_{t,c}^{Proj}$) and population ($P_{t,c}^{Proj}$) are based on the SRES storylines (Nakicenovic et al., 2000). Population scenarios were estimated with the UN (A2, B2) and the IIASA model (A1, B1); harmonized GDP scenarios were estimated by XX. Four storylines (A1, A2, B1, B2) are distinguished based on diverging assumptions on the societal development concerning economical values as well as regional connectivity. We used the country-specific datasets from CIESIN (2002b,a). Per-capita income projections ($I_{t,c}^{Proj}$) are given in US$\$_{1990}$ capita^{-1} a^{-1} (Fig. C.3), and population projections ($P_{t,c}^{Proj}$) in capita

for the years 1990-2100 in 5-year increments (Fig. C.2). For consistency of historical and projected income values, both were transformed into US$$_{2005}$ using deflators from the World Bank (WORLDBANK, 2011).

2.2 Intake and household waste

FAOSTAT (2011) does not provide information about the partitioning of food demand into intake and household waste. However, given the human metabolic energy requirements, plausible intake estimates are restricted. With a large heterogeneity between individuals, average intakes are mainly dependent on age and sex structure of a society, on the level of physical activity, and on the absence of food shortages (Smil, 2000; Food and Nutrition Board of the Institute of Medicine, 2005). Estimates for global average intake lies probably around 2000 (Smil, 2000) or 2100 (Bender, 1994) kcal capita^{-1} d^{-1} . The most impoverished countries probably have average intakes of 1700-2000 kcal capita^{-1} d^{-1}, while intake in rich countries is about 2000-2300 kcal capita^{-1} d^{-1} (Smil, 2000). These estimates include the incidence of obesity. Changes in the share of the obese population should not influence intake substantially; obese women only consume about 200 kcal capita^{-1} d^{-1}, obese man 400 kcal capita^{-1} d^{-1} more than healthy individuals (Food and Nutrition Board of the Institute of Medicine, 2005). The low spread of intake between rich and poor countries can be partly explained by falling physical labor in developed countries (Smil, 2000), and a more sedentary lifestyle of overweight individuals (Food and Nutrition Board of the Institute of Medicine, 2005).

To estimate intake and household waste, we therefore use a simple set of assumptions: First, we assume that household food waste is at least 15% of food demand. Second, we assume that all calories exceeding 2200 kcal capita^{-1} d^{-1} will be wasted, a value that lies in the upper range of estimates for rich countries. Third, we assume that the percentage of animal-based calories is the same in the wasted as in the total demand. The resulting waste shares match well current estimates (Wirsenius, 2000), but are dynamic for the future when per capita food demand rises.

2.3 Functional relations

Different functional relations can be formulated between food demand, income I and time t. Here, we estimate the parameters of two functional relations for both total per capita demand (2.3.1) and for the share of animal-based calories (2.3.2). These relations can subsequently be used to customize scenarios to their storylines (2.4.1).

Data preparation and analysis is performed using the programming language and statistical software R (R Development Core Team, 2011). To select the necessary predictors, regressions are compared by an analysis of variance and stepwise reduction of the models is performed using the Akaike Information Criterion AIC (function *step* of package stats). The resulting models were tested against observed data with linear regressions using function *lm* of package stats.

2.3.1 Total calorie demand

For total calorie demand C_T, we apply the Engel curve with an under-proportional increase with income I. The relation can be described by the power function g (Eq. 1) with parameter β which can be interpreted as an elasticity (see review in Cirera & Masset, 2010).

$$g(I) = C_T = \alpha \cdot I^{\beta}. \tag{1}$$

We model the parameters α and β to be time-dependent which represents the influence of other drivers than income. In order to provide appropriate functions for the scenario storylines (section 2.1), parameters α and β are determined in two alternative ways. For one parameter set we assume a linear temporal relation, and for the second set a decreasing time increment.

The first parameter set assumes a linear temporal relation and uses the linearized formulation of Eq. (1)

$$\ln(g(I)) = \ln(C_T) = \ln(\alpha) + \beta \cdot \ln(I) \quad . \tag{2}$$

Parameters $\alpha^* = \ln(\alpha)$ and β are estimated by a linear mixed-effect model (using function *lmer* of package lme4, Bates & Maechler, 2010). Hereby, $\ln(C_T)$ is described with the fixed effect $\ln(I)$ and random effect time t. This results in time-dependent relations (exponential for α and linear for β) and is further on referred to as formulation g_A (Eq. 3).

$$g_A(I,t) = exp(\alpha_1 + \alpha_2 \cdot t) \cdot I^{(\beta_1 + \beta_2 \cdot t)} \quad . \tag{3}$$

The second parameter set assumes a decreasing time increment. Parameters α and β are estimated per year by a nonlinear least squares model fitted to Eq. (1) (using function *nls* of package stats, Bates & Watts, 1988). Subsequently, the time series of parameters α and β are fitted to the asymptotic function m of Michaelis-Menten type (Eq. 4) with intercept ω_1, asymptotic value ω_2, half-saturation constant ω_3 and base year $t_1 = 1960$

$$m(t) = \omega_1 + \frac{\omega_2 \cdot (t - t_1)}{(t - t_1 + \omega_3)}. \tag{4}$$

With the resulting functions m_α and m_β, the alternative formulation for g can be written as

$$g_B(I,t) = m_\alpha(t) \cdot I^{m_\beta(t)} \quad . \tag{5}$$

This captures the dynamics with rapidly changing values during the first 20 years of observations and rather less increase in the last period and is referred to as formulation g_B (Eq. 5).

Both functional relations g_A and g_B can be applied to the historical data in a way that assumptions on their future development can be chosen according to the storyline of the respective scenario (see section 2.4.1).

2.3.2 Animal calorie share

To estimate the animal-based calorie demand C_L, the share of animal-based calories C_{LS} (Eq. 6) in total calories C_T is taken.

$$C_{LS} = \frac{C_L}{C_T} \tag{6}$$

Again, two functional relations are used to estimate the demand according to the chosen storylines. First, the linear regression h_A (Eq. 7) on income I and time t is fitted on logarithmic values of C_{LS} as

$$\begin{aligned} h_A(I,t) &= C_{LS} \\ &= \exp(\kappa_1 + \kappa_2 \cdot \ln(I) + \kappa_3 \cdot t + \kappa_4 \cdot \ln(I) \cdot t) \ . \end{aligned} \tag{7}$$

This approach assumes that livestock products are a normal necessity good in accordance with Bennet's law (Bennet, 1941).

Secondly, we assume that animal products are a normal necessity good at low incomes, but turn into inferior goods at high incomes. This can be justified e.g. by the ambition of better educated people for a more balanced, healthy or sustainable diet (Cirera & Masset, 2010). For the derivation of h_B (Eq. 8), the under-proportionally increasing square root function is multiplied with an exponential decline. Thus, the share of animal-based calories declines at high incomes but does not become negative.

$$\begin{aligned} h_B(I,t) &= C_{LS} \\ &= (\rho_1 + \rho_2 \cdot t) \cdot \sqrt{I} \cdot \exp(-(\lambda_1 + \lambda_2 \cdot t) \cdot I) \end{aligned} \tag{8}$$

The functional relations h_A and h_B will be chosen for the projections of C_{LS} according to the storylines (see section 2.4.1).

2.4 Scenario preparation

2.4.1 Choice of functional relations based on storylines of scenarios

The four SRES scenarios were developed to analyze the impacts of global climate change and possible mitigation strategies (Nakicenovic et al., 2000). They have four different underlying storylines. The A1 and A2 scenarios describe worlds emphasizing economic growth with societies that are characterized by materialistic attitudes. In contrast, the B1 and B2 scenarios describe worlds developing towards a sustainable economy, and where the society has a stronger environmental awareness. Common features of the A1 and B1 scenarios are a globalized economy and increased cultural and social interactions whereas A2 and B2 storylines emphasize more self-reliance and local solutions to economic and social problems. A1 and B1 are therefore assumed to have a higher level of education and labor productivity, which also translates into lower reproduction rates and therefore slower population growth or even decreasing population. The higher population growth in A2 and B2 scenarios results in relatively lower per-capita incomes as the available resources have to be shared by a larger population. To represent the differences in the materialistic A1 and A2 storylines and the sustainable B1 and B2 storylines,

we assign different functional relations for A1 and A2 than for B1 and B2. The globalized worlds A1 and B1 and their fragmented counterparts A2 or B2 are only distinguished by different population and income growth paths.

In order to meet the underlying storylines, we use the g_A and h_A functions to estimate food demand in the materialistic A1 and A2 scenarios (Eq. 9). For the storylines with a focus on environmentally sustainable lifestyles (B1 and B2), the formulations g_B and h_B were chosen (Eq. 10). With this parametrization, food demand C_T in the B scenarios is lower than in the A scenarios, even under the same income, interpreting the storyline in the way that higher environmental awareness leads to lower waste of food. Similarily, we assume that people in the B scenarios eat less livestock products (represented by a lower C_{LS}) once they become richer due to the wish for a more sustainable and balanced diet.

$$\mathrm{A}_1, \mathrm{A}_2 : C_T = g_A(I,t),\, C_{LS} = h_A(I,t) \tag{9}$$

$$\mathrm{B}_1, \mathrm{B}_2 : C_T = g_B(I,t),\, C_{LS} = h_B(I,t) \tag{10}$$

2.4.2 Calculating the demand scenarios

The data sources from WORLDBANK (2011), FAOSTAT (2011), CIESIN (2002a), CIESIN (2002b) do not completely cover all countries and the full time-span. For the scenarios (not for the regression analysis), we harmonized the data sources and filled missing data with regional average values. We excluded all countries not listed in FAOSTAT (2011), representing 2.1% of the world population according to WORLDBANK (2011). Historical GDP data was completed with estimates of UNSTATS (2011) for the countries of the Eastern Bloc.

For the 4 SRES scenarios, we projected per-capita vegetal ($C_{t,c}^{V\_Proj}$, Eq. 11) and animal-based calorie demand ($C_{t,c}^{L\_Proj}$, Eq. 12) for time t and countries c based on income $I_{t,c}^{Proj}$.

$$C_{t,c}^{V\_Proj} = g(I_{t,c}^{Proj}) \cdot (1 - h(I_{t,c}^{Proj}))\ \forall\ t,c \tag{11}$$

$$C_{t,c}^{L\_Proj} = \quad g(I_{t,c}^{Proj}) \cdot h(I_{t,c}^{Proj}) \quad \forall\ t,c \tag{12}$$

To meet the historical data for $C_{t,c}^{V\_Proj}$ and $C_{t,c}^{L\_Proj}$, we calibrated our estimates to the observed values $C_{1990,c}^{V\_cal}$ and $C_{1990,c}^{L\_cal}$ in the year 1990. There is evidence that heterogeneous national dietary patterns become more homogeneous through globalization (see e.g., Regmi & Unnevehr, 2005; Herrmann & Röder, 1995; Blandford, 2005). Our regression being a plausible target for a globalized diet, we assume that the deviations of observed values from our regression function gradually disappear until 2100. For this purpose, we applied the function *conv* (Eq. 13), for which calibrated demand X_t meets historical values in an initial year y_s and converges to the regression values until the final year y_e.

$$conv(X_t, C_0, y_s, y_e) = \begin{cases} \frac{t-y_s}{y_e-y_s} X_t + \frac{y_e-t}{y_e-y_s} C_0 & \forall\ t\ \in\ [y_s, y_e] \\ X_t & \forall\ t > y_e \end{cases} \tag{13}$$

$$C_{t,c}^{V\_calib} = conv(C_{t,c}^{V\_Proj}, C_{1990,c}^{V\_cal}, 1990, 2100) \quad \forall\, t, c \tag{14}$$

$$C_{t,c}^{A\_calib} = conv(C_{t,c}^{L\_Proj}, C_{1990,c}^{L\_cal}, 1990, 2100) \quad \forall\, t, c \tag{15}$$

Finally, the calibrated per-capita estimates were multiplied with the population projections $P_{t,c}^{Proj}$ to obtain total vegetal demand $E_{t,c}^{V\_Proj}$ (Eq. 16) and total animal-based demand $E_{t,c}^{L\_Proj}$ (Eq. 17).

$$E_{t,c}^{V\_Proj} = C_{t,c}^{V\_calib} \cdot P_{t,c}^{Proj} \quad \forall\, t, c \tag{16}$$

$$E_{t,c}^{L\_Proj} = C_{t,c}^{A\_calib} \cdot P_{t,c}^{Proj} \quad \forall\, t, c \tag{17}$$

For regional aggregates every country was assigned to one of ten world regions (Fig. B.1).

The validity of the projections was assessed by deriving trends for the transitional period from 1990 to 2007 using the partial Mann-Kendall test according to Hamed & Rao (1998) and Yue & Wang (2004). The trend itself is determined as median value of all differences between the values and their successors. Trends were considered to be significant for $p < 0.05$.

3 Results

For both total calorie demand and animal-based calorie shares, results are presented for the two alternative formulations. Subsequently, features of food demand projections following the SRES storylines are described. We concentrate on regional results, while country-level estimates are given in the Supplement and are visualized in appendix E. An interactive web-application can be used to explore the regional results (given in the Supplement). Own scenarios can be generated using the package "CalorieDemand" in the open source programming language R (R Development Core Team, 2011) which is included in the Supplement.

3.1 Regressions for total calorie demand from historical data

For the formulation g_A for total calorie demand, the linear mixed model of log-transformed data (Eq. 3) determines linearly increasing values for the time-dependent parameter α^* and linearly decreasing values for parameter β (Fig. D.4). The coefficients of determination of the linear regressions for both parameters are higher than 0.95 (Tab. 2). Parameter β, the income elasticity of the demand, is estimated by the mixed model as $0.1002 \pm 4.37 \cdot 10^{-4}$ (mean ± SE) and decreases from 0.101 in 1961 to 0.0995 in 2007. The overall fit of the model to observed data (Fig. 2a) is highly significant with a coefficient of determination of 0.65 and slope 0.66 (Tab. 6). The temporal development of the model (Fig. 2b) at high income $I = 60{,}000$ US$ shows an increase from 3587 kcal capita^{-1} d^{-1} in 2000 to 4289 kcal capita^{-1} d^{-1} in 2100.

For the formulation g_B, we use regression models (Eq. 5) with diminishing time-dependence of the parameters α^* and β of Michaelis-Menten type (Eq. 4). Parameters were determined significantly (all p-values < 0.001, Tab. 3) (Fig. D.4). Parameter β is estimated as $9.88 \cdot 10^{-2} \pm 3.21 \cdot 10^{-3}$ (mean ± SE). The resulting

Table 2 Regression parameters with linear time dependence for total calories for formulation g_A (Eq. 3).

	Intercept	Slope	r^2	p-value	F-statistics
α^*	2.825	$2.131 \cdot 10^{-3}$	0.96	< 0.001 (***)	1057
β	0.162	$-3.124 \cdot 10^{-5}$	0.96	< 0.001 (***)	1057

Table 3 Regression parameters with declining time dependence for total calories for formulation g_B (Eq. 4) (RSE: residual standard error).

	ω_1	ω_2	ω_3	p-value	RSE	Df
α	$9.339 \cdot 10^2$	$3.875 \cdot 10^2$	9.775	< 0.001 (***)	$9.579 \cdot 10^{-7}$	45
β	$8.941 \cdot 10^{-2}$	$8.445 \cdot 10^{-3}$	$-7.557 \cdot 10^{-1}$	< 0.001 (***)	$4.654 \cdot 10^{-3}$	45

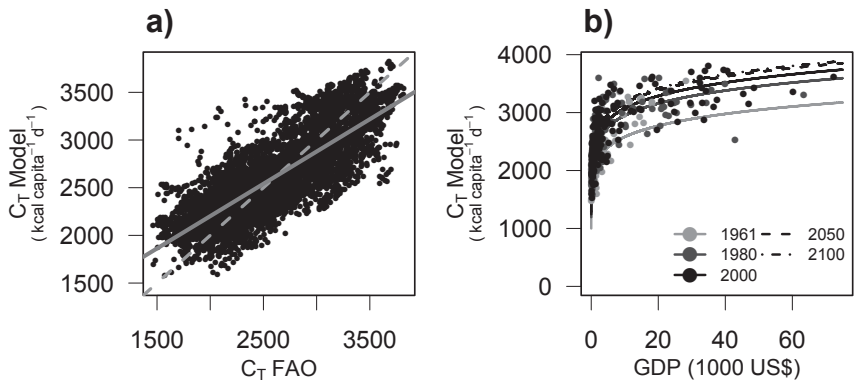

Fig. 1 Same as Fig. 2 for total calorie demand C_T with formulation g_B (Eq. (5), SRES scenarios B1 & B2).

demand values show a high correlation with observed data (Fig. 1a, Tab. 6) with

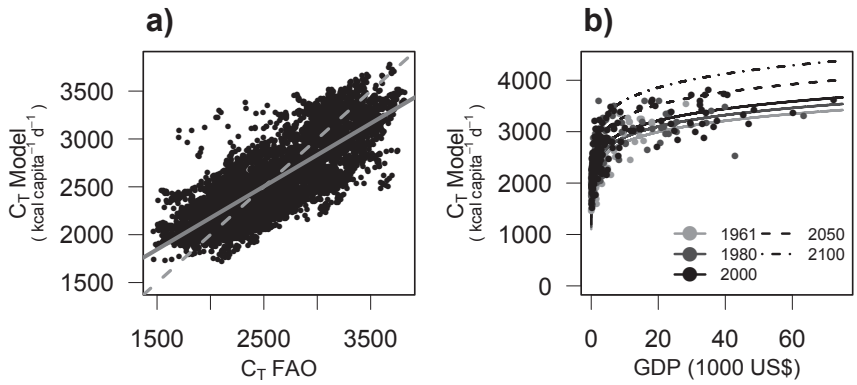

Fig. 2 Model estimation for total calorie demand C_T with formulation g_A (Eq. (3), SRES scenarios A1 & A2). Comparison of reported and modelled total calories (a) with linear regression (solid line, values see Tab. 6) and 1:1 line (dashed line). Predictions (b) for years 1961, 1980, 2000, 2050 and 2100 (lines) as well as reported data for 1961, 1980 and 2000 (dots).

a comparable coefficient of determination as g_A and a slightly higher slope of 0.68. The temporal change is similar to g_A for the historical time span but nearly disappears for the projected years (Fig. 1b). At income values of $I = 60{,}000$ US$, total calorie demand after g_B changes from 3661 kcal capita^{-1} d^{-1} in 2000 to 3805 kcal capita^{-1} d^{-1} in 2100.

3.2 Regressions for the animal-based calorie share from historical data

The animal-based calorie demand is determined by using the proportion of animal-based to total calories C_{LS} and fitting the linear regression model h_A (Eq. 7) and the inverted U-shaped curve h_B (Eq. 8).

For the monotonically increasing function h_A, anova results show that $\ln(I)$ and time t are significant predictors for $\ln(C_{LS})$ as well as the interaction of both. The linear regression model (Eq. 7, parameters in Tab. 4) is highly significant (Tab. 6) with a slope of 0.63 and intercept 0.04 (Fig. 3a). Predictions with this model (Fig. 3b) decrease dramatically at higher incomes of 60,000 US$ from 0.42 in 2000 to 0.2 in 2100. Although the coefficients of income and time are positive, the negative coefficient for the interaction produces this decline (Tab. 4).

Table 4 Regression parameters of the multiple linear regression for animal-based calorie share h_A (Eq. 7).

Parameter	Value	p-value
κ_1	$-3.673 \cdot 10^{1}$	< 0.001 (***)
κ_2	$4.497 \cdot 10^{0}$	< 0.001 (***)
κ_3	$1.604 \cdot 10^{-2}$	< 0.001 (***)
κ_4	$-2.077 \cdot 10^{-3}$	< 0.001 (***)

Table 5 Regression parameters for animal-based calorie share h_B (Eq. 8).

Parameter	Value	p-value	
ρ_1	$1.372 \cdot 10^{-2}$	< 0.001	(***)
ρ_2	$-5.295 \cdot 10^{-6}$	< 0.010	(**)
λ_1	$-1.102 \cdot 10^{-4}$	0.062	(.)
λ_2	$6.404 \cdot 10^{-8}$	0.031	(*)

The nonlinear function h_B (Eq. 8) is fitted for time-dependent parameters ρ and λ (Tab. 5). With the estimated values, parameter $\rho = \rho_1 + \rho_2 \cdot t$ is decreasing and $\lambda = \lambda_1 + \lambda_2 \cdot t$ is increasing over time (Fig. D.5). Thus, for a given income level both parameters affect the values over time in a declining way. This model is highly significant (Fig. 4a) with a slightly higher coefficient of determination of 0.62 (Tab. 6) and less overestimated values than with the multiple regression (compare Fig. 3a). Predictions in lower income ranges up to 20,000 US$ are comparable to the multiple regression model h_A but give much lower values at higher incomes (Fig. 4b). At 60,000 US$, the animal-based calorie share drops from 0.28 in 2000 to 0.14 in 2100.

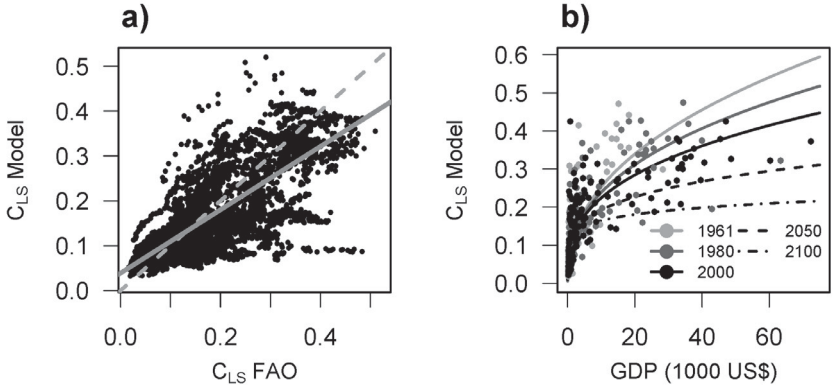

Fig. 3 Same as Fig. 2 for animal-based calorie share C_{LS} with formulation h_A (Eq. (7), SRES scenarios A1 & A2).

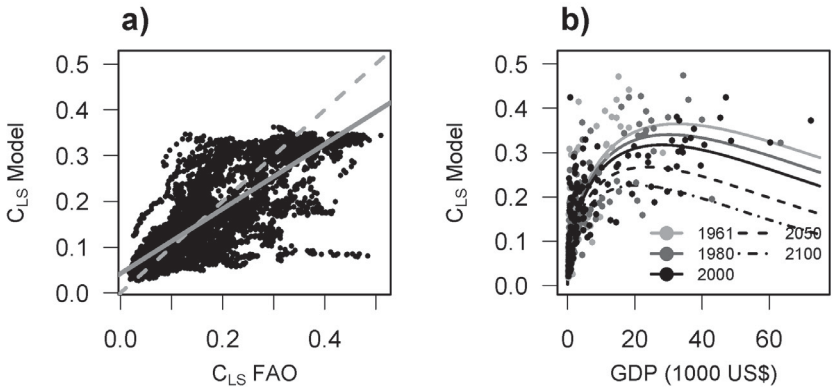

Fig. 4 Same as Fig. 2 for animal-based calorie share C_{LS} with formulation h_B (Eq. (8), SRES scenarios B1 & B2).

Table 6 Statistical properties of regression models on total calorie demand (C_T) and animal-based calorie share (C_{LS}).

Model	Intercept	Slope	r^2	p-value	F-statistics
g_A on C_T	857.9880	0.658	0.65	< 0.001 (***)	11060
g_B on C_T	846.4430	0.678	0.64	< 0.001 (***)	10551
h_A on C_{LS}	0.0389	0.705	0.63	< 0.001 (***)	9913
h_B on C_{LS}	0.0423	0.706	0.62	< 0.001 (***)	9685

3.3 Projections for the total calorie demand

Total calorie demand in all country-based projections (see appendix E) increases but distinct differences appear between scenarios A1 and A2 (Figs. E.6 and E.7) and those of B1 and B2 (Figs. E.8 and E.9), especially in Sub-Saharan Africa. Also for industrial countries in Europe or Northern America, the level as well as the proportion of demand values to those in transitional countries in Asia and southern America differ.

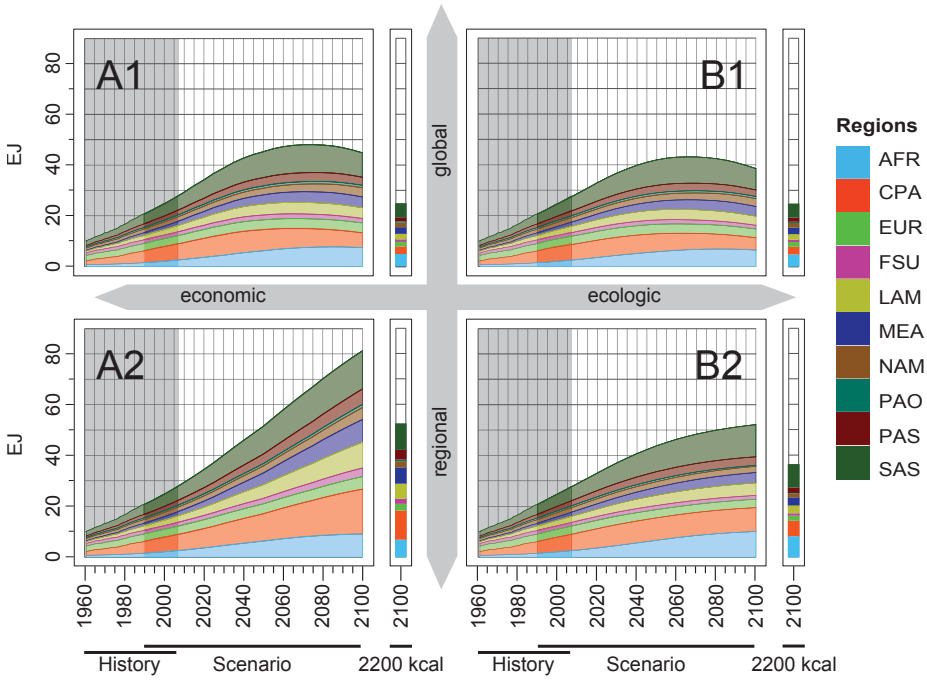

Fig. 5 Total food energy demand projections per region over time for 4 SRES scenarios (10^{18} Joule a^{-1}). Figures consist of historical data by FAO (1961–2007, "History"), respective projections (1990–2100, "Scenario"). The bars to the right ("2200 kcal") set total demand (intake and waste) in relation to a food intake of 2200 kcal $capita^{-1}$ d^{-1}in the year 2100.

Regionally aggregated values (population-weighted averages) show a strong increase in total calorie demand up to 2050 in all scenarios (Fig. 5). In the second half of the century, the scenarios diverge with either a continuous increase (A2), a stagnation (B2) or a slight decrease (A1, B1). Population growth (Fig. C.2), especially in developing countries, is the major driver for food demand throughout all scenarios. World population grows by 65% from 1990 to 2050 in the A1 and B1 scenarios, by 78% in the B2 scenario and by 114% in the A2 scenario. Compared to population growth, the increase in per-capita demand has a lower, yet substantial influence on total demand. It grows by 18% (A2), 21% (B2), 26% (B1) and 36% (A1) from 1990 to 2050. Here, the regions with the highest growth rates in per-capita demand are Sub-Saharan Africa, South, Central as well as South-East Asia.

Interpreting the surplus demand above 2200 kcal $capita^{-1}$ as probable food waste (Fig. 7), the global waste share increases from 17% in 1990 to 29% (A2), 31% (B2), 33% (B1) and 38% (A1) in 2050. The A1 scenario thereby reaches an average waste share globally of approximately the level of Northern America in 1990 (36%), with high-income regions approaching waste shares of almost 50% in 2050.

In order to compare our results with other studies (Valin et al., 2010; Kruse, 2010; Dorin et al., 2011; Alexandratos & Bruinsma, 2012), we aggregated the country-based values of the respective scenario to the same regions as given in the

literature. These regional population-weighted means as well as global averages compare well (Fig. 8, left and tables 9, 10 and 11).

3.4 Projections for the animal-based calorie demand

Animal-based calorie demand (see appendix E) increases for all countries and scenarios only until the middle of the 21st century (Figs. E.6 to E.9, panels b and d). Here, some countries with a traditionally high share of meat in the diet (e.g. Mongolia or Hungary) reach a per-capita animal-based calorie demand of more than 2000 kcal capita^{-1} d^{-1}. Projections for the end of the century (Figs. E.6 to E.9, panel f) show similarities between A1 and A2 as well as between B1 and B2 caused by the application of the same function but are not identical because of differing income projections. The highest overall deviations between scenarios are projected for countries with traditionally high meat consumption (like Russia, eastern Europe or Northern America) with dramatic decreases of demand values for animal-based products. Also for countries with very low historical values (like most countries in Africa), the differences for 2100 are high both due to the income projections and the applied regression functions.

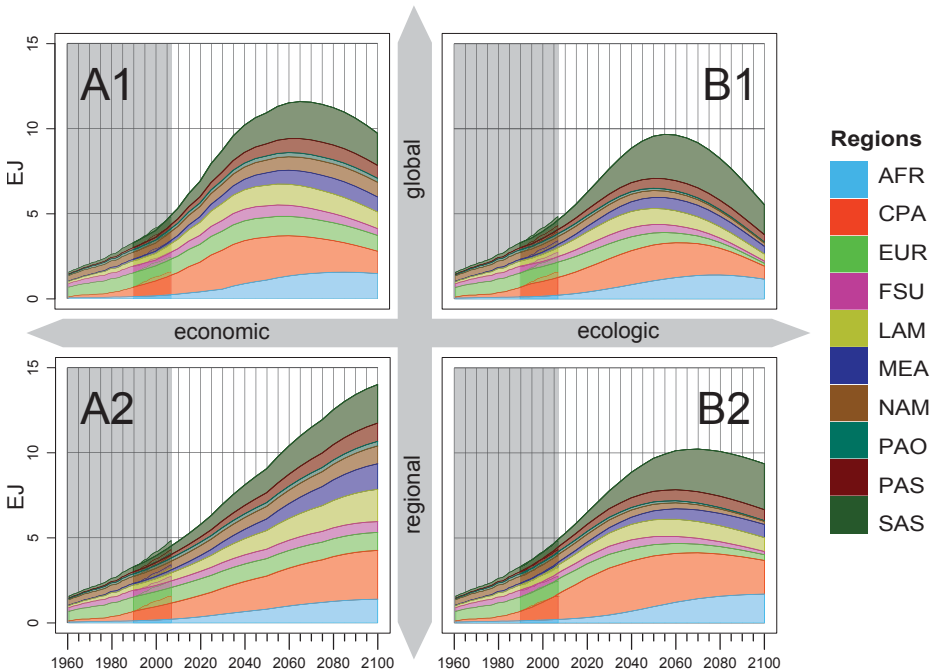

Fig. 6 Same as Fig. 5 for animal-based food energy demand projections per region over time for 4 SRES scenarios (10^{18} Joule a^{-1}).

The regionally aggregated demand for animal-based products increases in all four scenarios (Fig. 6). With global growth rates of 176% (A2), 191% (B1), 194% (B2) and 233% (A1), it outpaces the average increase in total food consumption.

The increase is strongest in Sub-Saharan Africa where total animal-based demand in 2050 reaches 7 to 9 times the value of 1990. In South Asia, demand also increases by a factor of 5 to 9. In contrast, animal-based demand stagnates in the OECD regions in the A scenarios or even declines in the environmentally oriented B scenarios.

For animal-based demand in the A1 and B1 scenarios, the increase in per-capita demand by 106% (A1) and 80% (B1) becomes the dominant driver, compared to the increase in population. In the regionalized A2 and B2 scenarios, population growth still remains the dominant driver, but also per-capita demand grows by 31% (A2) and 68% (B2). Throughout all four animal-based demand projections, the increase in the share of animal-based calories is more decisive than the increase in total calorie demand per capita.

In the second half of the century, most regions already have a decreasing share of animal-based calories. Only in the least developed regions like Sub-Saharan Africa, South and South-East Asia, the animal-based share in some scenarios still slowly increases. At the same time, global demand for animal-based calories decreases in all but the A2 scenario (Fig. 6). In the B scenarios this can be explained by a declining share and stagnating per-capita calorie consumption. In the globalized scenarios A1 and B1, a decreasing global population also contributes to the decline in the demand for animal-based products.

Comparing our results to those studies with projections for animal-based calories (Valin et al., 2010; Dorin et al., 2011), the regional population-weighted means as well as global averages are also in good agreement (Fig. 8, right and tables 9 and 11).

4 Discussion

4.1 Interpretation of functional relations

Driving processes behind food demand dynamics are most likely different for the short-term and for the long-term. Major drivers of short-term dynamics include harvest failure, food storage capacities, expectation failures, policy interventions like trade regulations or biofuel targets, or business cycles and speculation. In contrast, major drivers of long-term dynamics include population, income, urbanization, demographic composition, market access, lifestyles, or the extent of physical labor. Long-term models should therefore consider other drivers and dynamics than short-term models. We decided to reduce the complexity to three major indicators that are representatives for these drivers, namely population, income and time.

Both sets of functions for per-capita consumption (g_A and g_B) and the share of animal-based products (h_A and h_B) give satisfactory results for the reproduction of historical data but assumptions on the form of the temporal development have a decisive effect on the projections. Especially the functional types for the share of animal-based products strongly diverge for high-income countries. As only 10 countries reach an average income above 40,000 US$ before 2005, the few observations do not allow to favor one of these functional relations.

Our statistical analysis cannot reveal the underlying causal relationship between income, time and food demand, which is open for debate. The strong in-

Fig. 7 Per capita demand for plant and animal-based calories, distinguished into intake (plates) and household waste (waste bins). For 1990, we used waste shares from Wirsenius (2000), for future scenarios we assumed all demand above 2200 kcal capita^{-1} d^{-1} as waste, based on Smil (2000). We assume that plant and animal calories are wasted at equal shares.

crease of both total food demand and the share of animal-based products for low and medium income countries may be attributed amongst others to a higher purchasing power of consumers or a stronger economic development of food producers leading to falling prices for staple goods (Popkin, 1993, 2001). The positive non income-related time trend of total food demand could stem from a higher social alienation from the food production process due to urbanization, more single member households and increased openness to global markets (Parfitt et al., 2010; Popkin, 1993). Finally, the negative time-trend for the share of animal calories

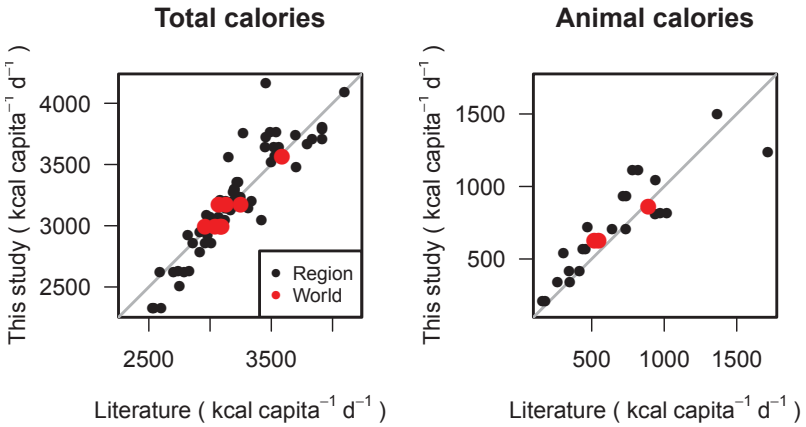

Fig. 8 Comparison of estimates of total (left) and animal-based (right) calorie demand between values from the literature (Valin et al., 2010; Kruse, 2010; Dorin et al., 2011; Alexandratos & Bruinsma, 2012) (x-axis) and own calculations (y-axis). Values are given for 2030 and 2050 as regional means (black dots) which are derived as population weighted averages of countries in the same regions as given in the literature and global means (red dots). Compare tables (9, 10, 11) and discussion in section G.2.

and the eventual negative connection to income could be attributed to a higher health consciousness among better-educated people (Cirera & Masset, 2010) or to alternative lifestyles like vegetarianism becoming a status symbol.

At the same time, our empirical findings are not in line with the underlying assumptions of simple economic demand models that are based on a positive elasticity constant over time and income. The significant time trends in all our representations indicate that either elasticities should be adjusted over time, or that demand models should include other drivers than income. Additionally, our U-shaped regression even opens up the possibility to question a positive elasticity as such.

Our demand projections are influenced by our assumption, that national peculiarities, represented by the divergence of current demand from the regression results, disappear until the end of the 21st century and that all values converge to the regression results (Eq. 13). A contrasting hypothesis could be that these peculiarities are kept over time, such that the projections for 2100 deviate by the same percentage from the regression results as in the inital year (here 1990). Changing our assumption (Eq. 13) accordingly leads globally to an increase of the total calorie demand of about 3.3 % (2050) and 6 % (2100). The alteration of the assumption would have larger consequences on the heterogeneity across countries. Whereas under the convergence assumption, the standard deviation across countries is about 704 kcal capita^{-1} d^{-1}; this value increases to 833 kcal capita^{-1} d^{-1} when the peculiarities in national diets are maintained. Total demand is much higher than in the globalized estimates for the former Soviet Union members like Uzbekistan, Kosovo, Bosnia and Herzegovina, Serbia, Montenegro, or Ukraine, while it is lower for African countries like Angola, Chad, Eritrea, or Ethiopia.

Nevertheless, the analysis of the FAO statistics of the past 50 years already reveals a convergence of food baskets (Khoury et al., 2014) so that the assumption of convergent diets is most likely.

4.2 Interpretation of projection results

For long-term projections, data quality is still low and the understanding of socio-economic dynamics is weak. Future analysis has therefore to rely on the evaluation of multiple scenarios instead of best-guess forecasts. The benefit of such scenario analysis is to consistently deliberate the implications of a set of assumptions. We provide a tool that can be used for such scenario analysis. Even though the set of adjustable assumptions is limited (adjustable population and per-capita income, two sets of functional relations for per capita caloric demand and for the share of livestock-based calories, two types of calibration, exogenous values for long-term targets), this provides already considerable freedom to create own scenarios that can be adapted to a storyline consistently.

Exemplarily, we applied our method to the SRES storylines, which are characterized by the contrasting concepts of economic (A) versus ecological values (B) and increasing globalization (1) versus regionalization (2). Their combinations result in different developments regarding population increases (Fig. C.2) as well as income projections (Fig. C.3).

The resulting regional per-capita demand values remain in reasonable ranges given the strong income growth in the scenarios. The highest value for total calorie demand of 4700 kcal capita^{-1} d^{-1} for Northern America (Fig. F.10) has to be seen in the context of the materialistic storyline of scenario A1 and an per-capita income which quadruples compared to current levels. For the sustainable scenarios B1 and B2, the projected per-capita demand (Fig. F.10) exceeds only slightly the observed range of the past despite more than doubling per-capita income. Animal-based calorie demand on the regional level (Fig. F.11) remains completely in the observed range. The increase of vegetal demand beyond observed ranges (Fig. F.12) in scenarios B1 and B2 stems from the reduction of animal-based calorie demand due to the assumption of a more healthy diet in higher income ranges.

Some of our results seem surprising given the SRES storylines, but are consistent at a closer look.

First, one would not expect an increase of per-capita demand in high-income countries, as metabolic requirements are constant or even decline due to less physical labour in high-income groups (Smil, 2000). Nevertheless, demand in these regions still increases, which has to be attributed to an increase in food waste at the household level.

Second, it seems plausible that food waste in the environmentally oriented scenarios (B) is lower than in the economically oriented scenarios (A) (Fig. 7). A closer look reveals that a highly populated A2 world goes along with lower per-capita incomes in many developing regions. This leads to lower total per-capita demand values and, hence, people simply cannot afford to waste food.

Third, the animal-based share in developing countries increases faster in the environmental than in the materialistic scenarios. This occurs only for medium income countries with an income range that increases from 1,000 - 26,000 US$ in 1990 to 3,600 - 39,000 US$ in 2100. In this income range, the projected livestock

share is smaller for function h_A than for h_B which are applied for the scenarios A and B, respectively (Eqs. 9, 10). For the same income and year, results from function h_A are up to 20 % lower than values from h_B. Especially for the regionalized scenario B2, income in many countries stays below 30,000 US$ so that their livestock share is higher than in scenarios A1 and A2. This peculiarity comes from the requirement that both models have to be consistent with past data. If the share of animal-based calories is assumed to fall with high incomes, the medium incomes need to have an even higher share in order to fit observations (compare Figs. 4b and 3b). We did not find a functional type fitting the same observation data that produces lower results for every possible combination of drivers.

Although using a single global approach for the estimation of food demand on the country level, trends of the projections in the overlapping period of data and projections compare well to those from observed data (see appendix G.1). Moreover, our estimates are well within the range of published projections by other authors (see detailed discussion in appendix G.2).

4.3 Food security prospects

The share of population which is undernourished declines with increasing per-capita demand at the country-level (FAO, 2004; von Grebmer et al., 2012). According to FAO (2004), in all countries with a per capita-demand above 2500 kcal capita^{-1} d^{-1}, less than 20% of the population are undernourished. Comparing von Grebmer et al. (2012) with FAOSTAT (2011), countries with a per-capita demand of 2000-2500 kcal capita^{-1} d^{-1} have an average global hunger index (GHI) of 18 (range 6-30), categorized as serious hunger. Countries with a per-capita demand below 2000 kcal capita^{-1} d^{-1}have an average GHI of 30 (range 18-42), categorized as extremely alarming hunger. In 2005, 56 countries with a total population of 2 billion people had a per-capita demand of 2000-2500 kcal capita^{-1} d^{-1}, and 11 countries with 0.2 billion people had a per-capita demand below 2000 kcal capita^{-1} d^{-1}.

According to our projections, serious hunger will persist in some countries also in the coming decades (see E.6, E.7, E.8, E.9). While there is no country in 2050 that has a per-capita demand below 2000 kcal capita^{-1} d^{-1}, many people still live in countries with an average demand of 2000-2500 kcal capita^{-1} d^{-1}, ranging from 1.9 billion people in the A2 scenario to 0.3 billion people in the A1 scenario.

The major geographical areas of concern are clustered in the tropics. Most severely affected are the world regions Sub-Saharan Africa (<2400 kcal capita^{-1} d^{-1} and 8% animal-based calories in the year 2005) and South Asia (<2300 kcal capita^{-1} d^{-1} and 9% animal-based calories in 1990). According to our scenarios, per-capita demand will grow slowly in both regions. Comparing the demand in these least developed regions to the demand of a transition country like China in 1990 ($\approx$ 2600 kcal capita^{-1} d^{-1}, 12% animal-based calories), Sub-Saharan Africa will reach a similar level only in 2030 (A1) or 2045 (B2). Also South Asia will reach this level only between 2020 (A1) and 2030 (A2). This time lag of 30 to 55 years to China suggests that wide spread hunger and malnutrition in these regions will be a persistent issue also for the next decades.

While undernourishment can be expected to decline, obesity and related health problems are currently strongly rising. Already today, obesity is responsible for a

similar extent of pre-mature deaths as hunger (WHO, 2009). It is of large interest to include the dynamics that determine overnourishment into future assessments.

5 Conclusion

This paper provides a simple method to create customized food-demand scenarios. It requires only information on GDP and population which are the key indicators of many future scenarios. Despite this slim approach, the results of our projections are in good accordance with other studies.

Many investigations use scenarios that are adapted ex post to meet a certain storyline, e.g. by reducing the demand of livestock products by a certain share (Popp et al., 2010; Stehfest et al., 2009; Wirsenius et al., 2010). Our method is exceptional by adapting ex ante the core assumptions of the functional relations instead of changing the results ex post. In so doing, projections remain consistent with historical data. The results of our ex-ante customization sometimes seem unintuitive at the first glance, but at a closer look they are logical and more consistent with the overall storyline and history than an ex-post approach.

Our method has been applied exemplarily for constructing a set of food demand scenarios consistent with the SRES storylines, but could also be easily used for upcoming scenarios like the Shared Socio-economic Pathways (SSP, Kriegler et al., 2012). Applying our method or using our results may be of interest for different scientific communities. For investigations of food security aspects, our long-term scenarios affirm that in the tropics, especially Sub-Saharan Africa, massive undernourishment is likely to persist in the coming decades and requires a long-term engagement. For health studies, the dynamics of excessive consumption of animal-based calories may be interesting, as it can be clearly linked to a number of heart and vascular health problems and colon cancer (Cross et al., 2007; Corpet, 2011; Fraser, 1999; Kelemen et al., 2005; Sinha et al., 2009). Finally, our long-term food demand scenarios can be valuable for the environmental science community. As our results suggest a doubling of food demand in all scenarios and an over-proportional increase of animal-based calorie demand, the pressure on environmental systems can be expected to increase strongly.

6 Online Supporting Material

Datasets are available on country and regional resolution (URL). An interactive web-application (http://bodirskys.de/demand/) can be used to explore visually the regional demand estimates from the SRES scenarios. Own scenarios can be created using our package "CalorieDemand" of the open source programming language R (R Development Core Team, 2011). The package contains functions and input data of our approach which are described in help files and a manual.

List of Figures

A Discussion on data sources

Beside the uncertainty connected to our assumptions and methodology, the uncertainty underlying the data we use is also passed on to our estimates. While some random errors might cancel out in our method, systematic biases in a specific direction will persist. In the following we will discuss the quality of each data source.

A.1 FAOSTAT

FAOSTAT provides the most comprehensive statistical database on global agricultural production and use. According to Smil (2000), data is mostly collected based on interviews, surveys or secondary sources. Only about 10 percent of data in European countries is based on direct physical measurements. In developing countries, data availability is even worse and data gaps have often been filled using extrapolation techniques. Smil (2000) estimates, that errors in country data might be as much as 20-25%, and global phytomass may diverge by 5-10 %. The subsequent use of produced crops and livestock products is even more uncertain than their production. Errors accumulate over the estimation of the shares used for feed, food, seed or processing, the postharvest and processing losses and nutritional contents of the final products. A large part of the underlying parameters are not obtained from national statistics but have to be estimated in the FAO headquarters in Rome (Smil, 2000).

Most errors probably do not represent a systematic bias in one direction. They may thus partly cancel out within the regressions. However, any systematic bias of our input data will remain in our regressions and thus in our scenario projections. One example for a systematic bias is that a number of food items hunted or gathered from wildlife (e.g. wild meat, insects or gathered plants, seeds, nuts and fruits) are not represented in these statistics (Smil, 2000). As poor countries tend to have higher shares of products from wildlife origin, we systematically underestimate their food availability. Yet, as incomes increase in the future, the projections are more influenced by the data points with high income that tend to have a better data quality.

A.2 Worldbank Data

Also GDP estimates are subject to uncertainty. As GDP is usually estimated based on tax data, shadow markets and smuggled goods are not covered. Subsistence production, making up a major part of poor economies, is also not covered. In general, this leads to a systematic bias, underestimating the GDP of less developed countries. As some goods and services cannot be traded, the purchase power of a currency can also be different within the country to its purchase power abroad so that the use of GDP in purchase power parity (PPP) would be better comparable in the international context (Nordhaus, 2007). Nevertheless, we based our regressions on GDP in market exchange rates (MER) because of two reasons. Firstly, available Worldbank data for MER have a longer coverage (since 1960) than PPP values (since 1980). Secondly, the SRES GDP scenarios (Nakicenovic et al., 2000) are only available in MER.

Population estimates are also rather uncertain. Due to high administrative efforts population censuses are carried out only in larger time-intervals. 17% of all countries in the UN 1998 Revision had no census data from after 1985 available (Zlotnik, 1999). Revised population estimates may therefore show substantial correction. For example, a census in Nigeria in 1991 came up with 35 million people less than expected (Keilman, 2001).

A.3 SRES projections

The SRES population and income scenarios (CIESIN, 2002b,a) are calibrated to the year 1990 and therefore inherit the uncertainty of the base year estimates mentioned above. For the subsequent years 1990-2010, which lie now in the past, the SRES scenarios are already projections and therefore diverge substantially from recorded values (WORLDBANK, 2011). The SRES projections overestimate global average GDP per capita in 2010 by 5% (B1) to 9%

(A2) compared to WORLDBANK (2011). Regional divergences are even higher. The per capita GDP in Centrally Planned Asia (CPA) was estimated to be 22% (A1), 58%(A2), 44%(B1) lower or 14% (B2) higher. For Middle East and North Africa (MEA), scenario values were too high by 22% to 85%. Errors are much lower in developed regions than in developing countries. Also the population scenarios diverge from recorded values (WORLDBANK, 2011). In 2010, the projections for A1, B1 and B2 were approximately 1% higher, while A2 was almost 5% higher. Again, regional estimates for our 10 world regions are much higher with a spread of -3 to +11%. There is a clear tendency in all scenarios to overestimate population growth in developed countries and to underestimate the growth in developing countries.

The originally regional scenarios were disaggregated by CIESIN (2002b,a) to country level by applying the regional growth rates to country-specific starting values. This method creates unrealistic values for countries with relatively small population that diverge in their state of development from the rest of their region. Examples include Mauritius, The Bahamas, Singapore, or Macao, which reach extremely high values for per-capita income in 2100.

B World regions

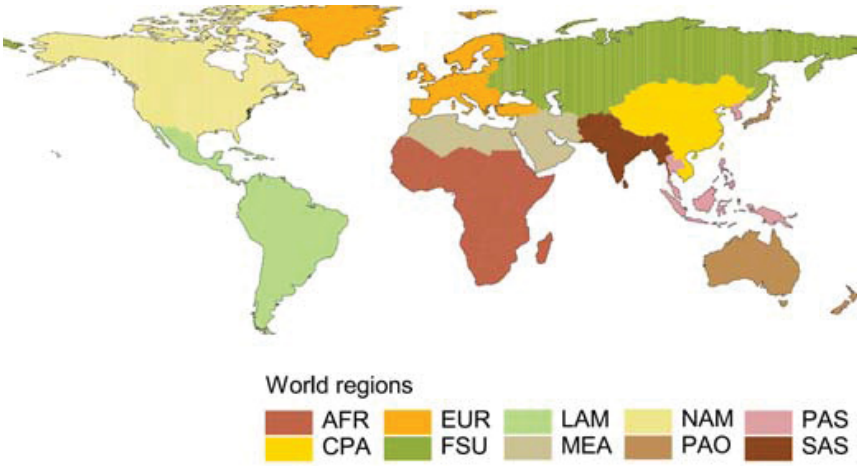

Fig. B.1 World regions with similar socio-economic conditions according to Lotze-Campen et al. (2008): AFR = Sub-Sahara Africa, CPA = Centrally Planned Asia (incl. China), EUR = Europe (incl. Turkey), FSU = Former Soviet Union, LAM = Latin America, MEA = Middle East, NAM = North America, PAO = Pacific OECD (Australia, Japan and New Zealand), PAS = Pacific Asia, SAS = South Asia (incl. India).

C Regional demand projections for SRES scenarios

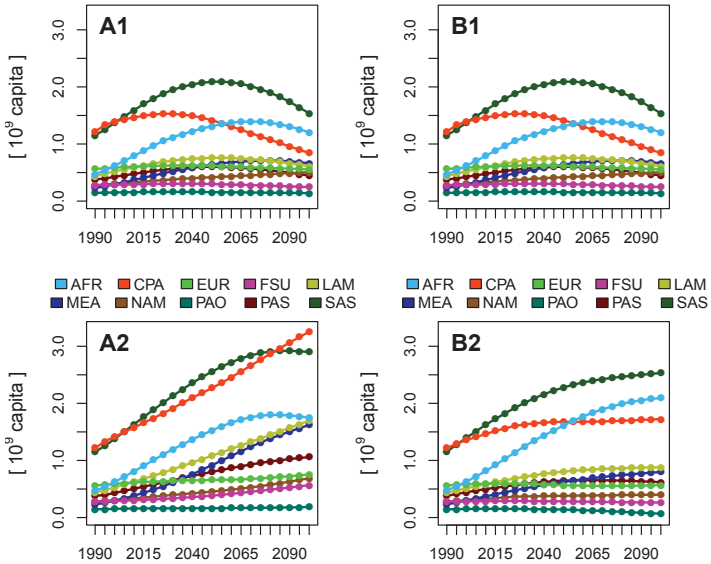

Fig. C.2 Population projections of the four SRES scenarios aggregated to 10 world regions (Fig. B.1) in 10^9 capita.

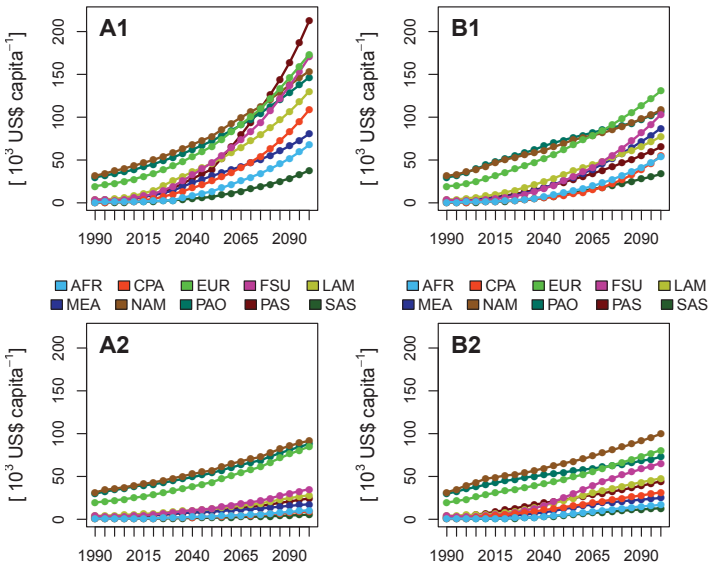

Fig. C.3 Per-capita income projections of the four SRES scenarios aggregated to 10 world regions (Fig. B.1) in 10^3 US$\$_{2005}$ capita^{-1}.

D Regression parameter values

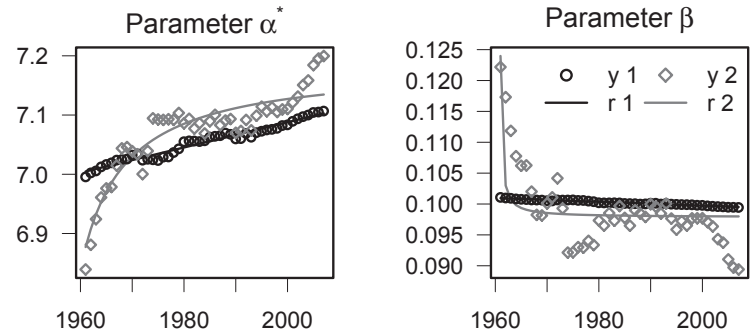

Fig. D.4 Regression parameter values for total calories with functions g_A and g_B for parameter α^* (left) and β (right). Yearly values of the linear mixed effect model (y1: black dots) and the linear regression (Table 2) (r1: black line) are given as well as yearly fitted values for the non-linear fit (y2: grey diamonds) and the Michaelis-Menten-type functions (Table 3) (r2: grey line).

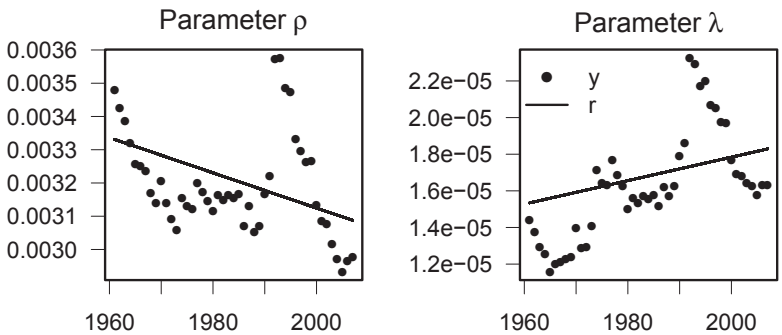

Fig. D.5 Regression parameter values for total calories with function h_B for parameter ρ (left) and λ (right). Yearly fitted values for the non-linear function (dots) and the linear regression (Table 5) (line) are given.

E Country-specific projections of SRES scenarios

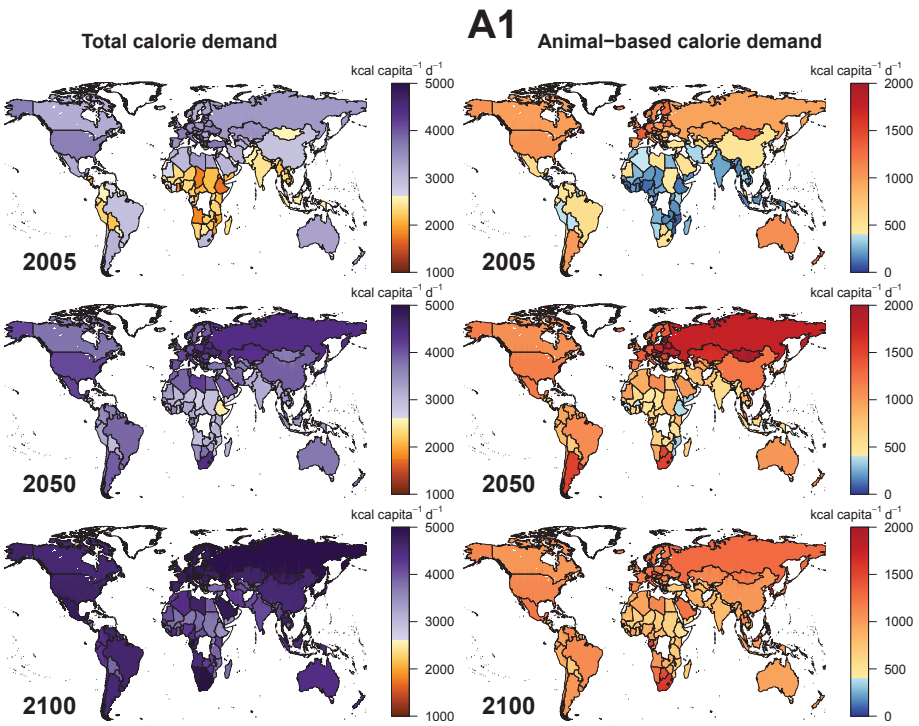

Fig. E.6 Projections for scenario A1 for total (left) and animal-based (right) calories per capita for the years 2005 (top), 2050 (middle) and 2100 (bottom).

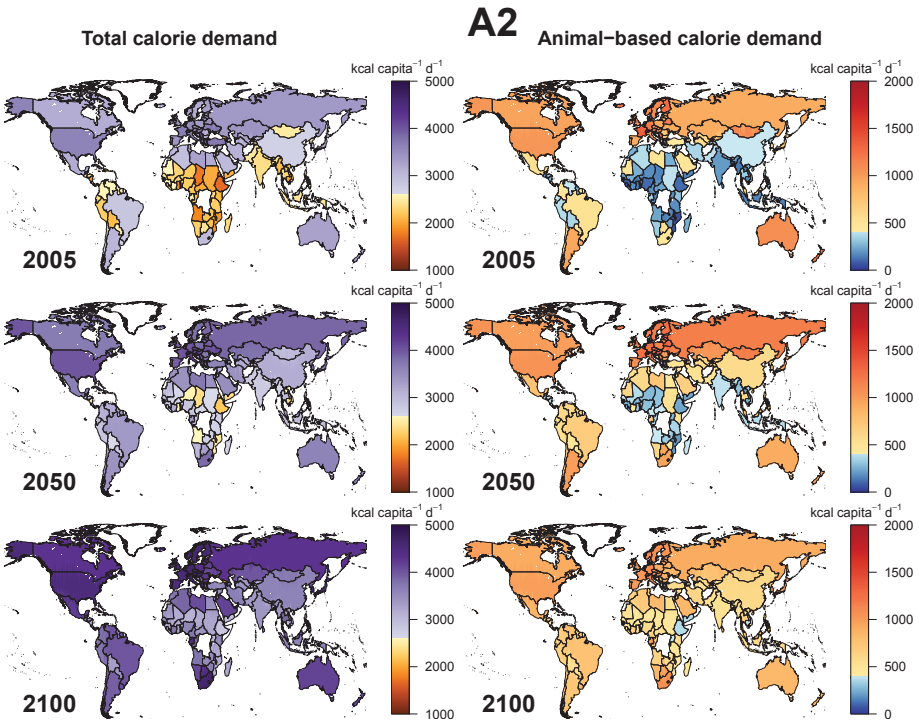

Fig. E.7 Projections for scenario A2 for total (left) and animal-based (right) calories per capita for the years 2005 (top), 2050 (middle) and 2100 (bottom).

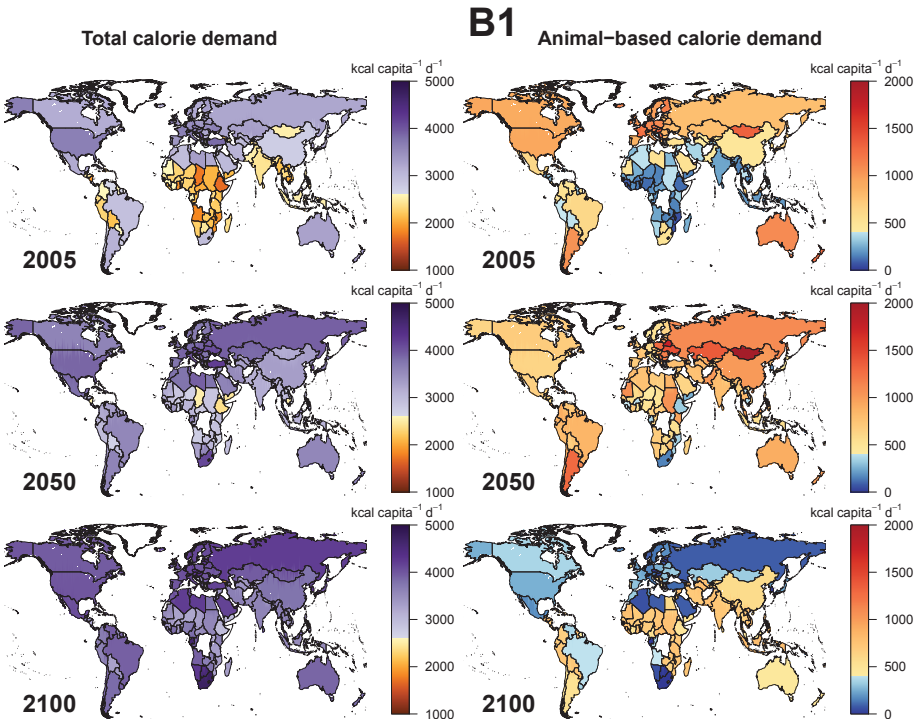

Fig. E.8 Projections for scenario B1 for total (left) and animal-based (right) calories per capita for the years 2005 (top), 2050 (middle) and 2100 (bottom).

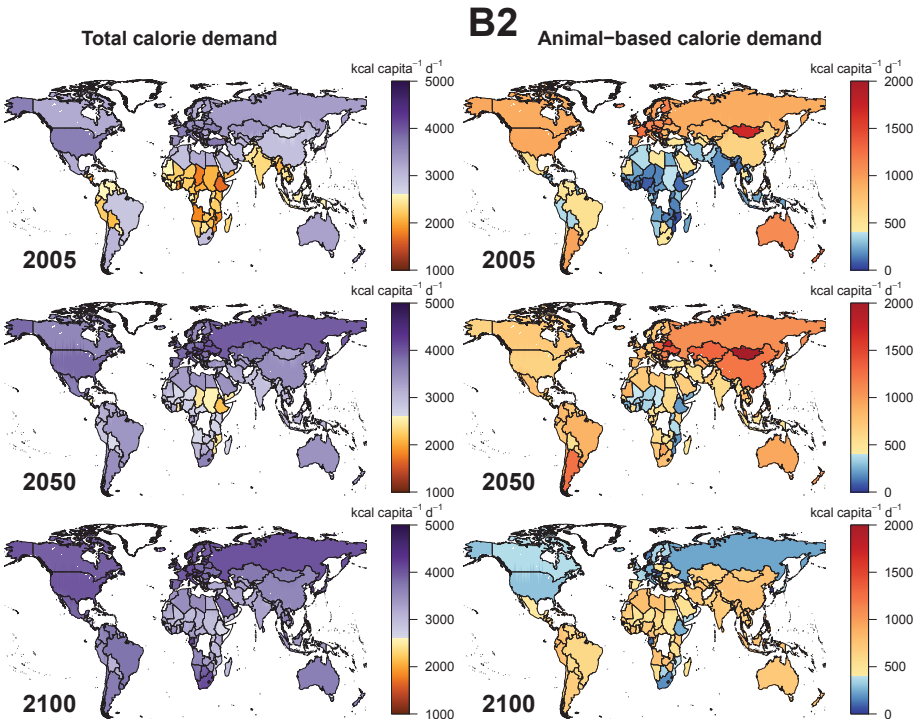

Fig. E.9 Projections for scenario B2 for total (left) and animal-based (right) calories per capita for the years 2005 (top), 2050 (middle) and 2100 (bottom).

F Regional projections of demand scenarios

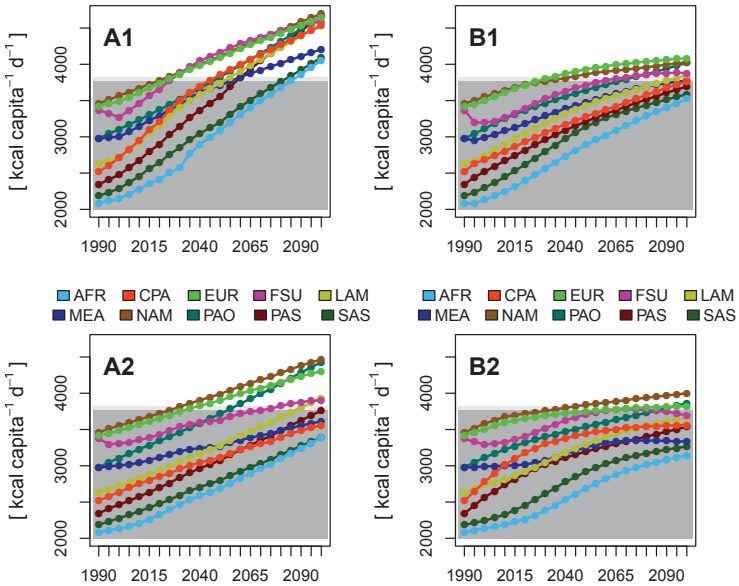

Fig. F.10 Total calorie demand projections for 4 SRES scenarios as population-weighted means for each world region. Shaded areas indicate observed value ranges from historical FAO data on the country (light shading) and on the regional scale (dark shading).

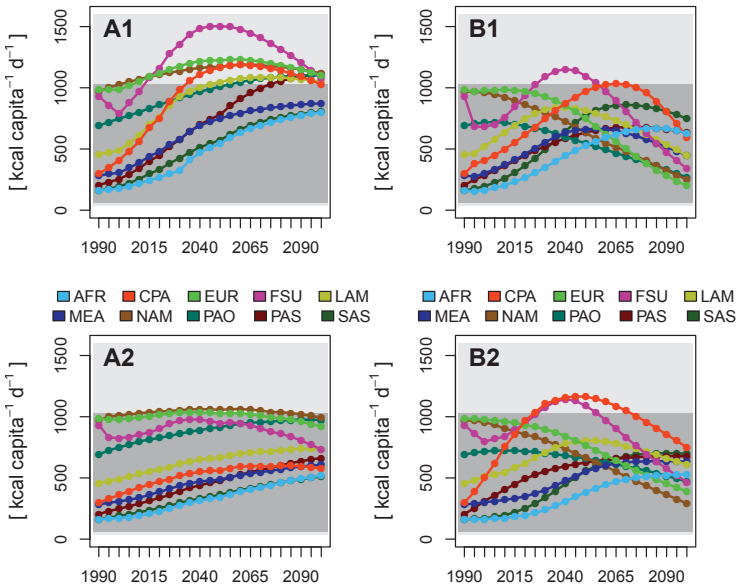

Fig. F.11 Animal-based calorie demand projections for 4 SRES scenarios as population-weighted means for each world region. Shaded areas indicate observed value ranges from historical FAO data on the country (light shading) and on the regional scale (dark shading).

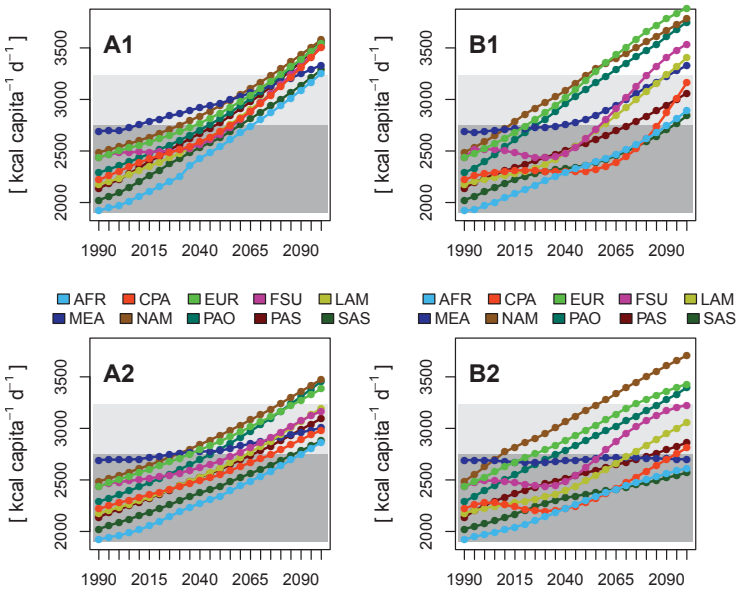

Fig. F.12 Vegetal-based calorie demand projections for 4 SRES scenarios as population-weighted means for each world region. Shaded areas indicate observed value ranges from historical FAO data on the country (light shading) and on the regional scale (dark shading).

G Literature and data comparison

G.1 Comparison of trends in overlapping period

The SRES projections start in 1990 and have an overlapping period of approximately 20 years with historical data. Within this period, population and income projections already diverge substantially from historical records (see A.3). Nevertheless, we decided to keep the base year of 1990 in order to make our results consistent with other SRES-based studies. Moreover, this allows for evaluating the dynamics of our projections during the overlapping period. We used the last 20 years of the reported data (1988-2007) as well as the first 20 years of the projection values (1990-2010) to derive regional temporal trends. For most of the regions, the reported data show significant trends as calculated by the Mann-Kendall test. For the total calorie demand (Tab. 7), regions with highest trends are best met by globally oriented scenarios (A1 or B1), whereas medium increases are mostly met by the regional scenarios (A2 or B2). Trends for the animal-based share have much lower values (Tab. 8) and are in most cases well represented by regional scenarios. Here, the higher positive trends follow mostly an economic storyline (A2) and the lower and negative trends are better met by an ecologic scenario (B2).

Table 7 Significant trends (values only shown for $p < 0.05$) for total calorie demand (kcal capita^{-1} d^{-1} a^{-1}) between 1988 and 2007 for FAO data and between 1990 and 2020 for all scenarios estimated using the Mann-Kendall test.

Region	FAO	A1	A2	B1	B2
AFR	17.39	11.99	7.50	11.76	5.64
CPA	26.79	23.31	11.22	12.16	23.10
EUR	5.04	11.28	8.03	10.72	7.89
FSU		13.46	4.40		
LAM	14.65	19.07	10.11	15.08	9.10
MEA	10.61	11.74	4.17	8.74	1.36
NAM	23.10	10.60	8.54	8.78	8.99
PAO	-5.58	13.05	11.91	13.23	11.19
PAS	14.19	19.39	11.66	14.85	17.64
SAS	4.82	16.33	9.69	14.82	8.37

Table 8 Significant centennial trends (values only shown for $p < 0.05$) for animal-based demand shares (kcal kcal^{-1} capita^{-1} day^{-1} 0.01 a^{-1}) between 1988 and 2007 for FAO data and between 1990 and 2020 for all scenarios estimated using the Mann-Kendall test.

Region	FAO	A1	A2	B1	B2
AFR		0.12	0.06	0.14	0.02
CPA	0.58	0.40	0.16	0.30	0.63
EUR	-0.11	0.06	-0.02	-0.08	-0.11
FSU	-0.33	0.24			
LAM	0.18	0.26	0.07	0.27	0.13
MEA	0.09	0.19	0.09	0.19	0.07
NAM	-0.07	0.04		-0.17	-0.18
PAO	-0.03	0.07	0.06	-0.10	-0.05
PAS	0.10	0.24	0.12	0.23	0.27
SAS	0.13	0.18	0.09	0.22	0.08

G.2 Comparison of calorie demand projections to other studies

In the following, we compare the projections of per-capita calorie demand to published estimates. Since the respective studies use world regions which partly differ from our geographical units, we aggregate our national values in order to make them comparable. For the comparison to Alexandratos (1999), Valin et al. (2010), Alexandratos & Bruinsma (2012), and Kruse (2010), we choose calorie projections of the B2 scenario (Table 9 and 10). For Valin et al. (2010) this is a natural choice, since their assumptions on income and population are also based on the B2 scenario of Nakicenovic et al. (2000). The B2 scenario is also the most appropriate "middle of the road" for the comparison with Alexandratos (1999), Alexandratos & Bruinsma (2012) and Kruse (2010), which have only one projection. To compare our calorie demand values to Dorin et al. (2011)(Tab. 11), we choose the A1 scenario, which fits best to the Agrimonde Global Orchestration Scenario based on the Millenium Assessment report.

Table 9 Comparison of projections in kcal capita^{-1} d^{-1}. Transforming the projections of Alexandratos et al. (2006) for animal products into calorie values has been done by Valin et al. (2010). We have aggregated our national values in order to match the FAO regions below. In comparison with values from FAOSTAT (2011), we presume that Valin et al. (Tab. 2, 2010) refers to total animal-based calories. The B2-Regional scenario of Valin et al. (2010) takes population and income projections of the B2-scenario of Nakicenovic et al. (2000) and assumes that food demand evolves according to strong regional specifications.
DC=Developing Countries, IC=Industrial Countries (including Australia, Japan and New Zewland), TC=Transition Countries. The following subregions belong to DC: SSA=Sub-Sahara Africa (excluding South Africa), MEA=Middle East and North Africa (including Turkey and Afghanistan), LAM=Latin America, SAS=South Asia, EAS=East Asia.

	Alexandratos et al. (2006)		**Valin et al. (2010)**		**Alexandratos & Bruinsma (2012)**	**This study**	
Scenario	Default		B2-Regional		Default	B2	
	Total	Animal	Total	Animal	Total	Total	Animal
Year 2030							
World	**3040**	**548**	**3095**	**518**	**2960**	**2990**	**626**
DC	**2960**	**458**	**3010**	**440**	**2860**	**2859**	**568**
SSA	2600	180	2541	160	2530	2326	211
MEA	3130	416	3311	345	3130	3146	414
LAM	3120	735	3420	640	3090	3046	710
SAS	2790	350	2701	263	2590	2621	344
EAS	3190	738	3339	716	3130	3202	937
IC	**3520**	**1019**	**3562**	**973**	**3450**	**3646**	**814**
TC	**3150**	**821**	**3529**	**781**		**3565**	**1111**
Year 2050							
World	**3130**				**3070**	**3177**	**702**
DC	**3070**				**3000**	**3073**	**683**
SSA	2830				2740	2632	363
MEA	3190				3200	3274	552
LAM	3200				3200	3296	805
SAS	2980				2820	2924	563
EAS	3230				3220	3354	994
IC	**3540**				**3490**	**3762**	**697**
TC	**3270**					**3753**	**1077**

Table 10 Comparison of projections in kcal capita^{-1} d^{-1}. Values extracted from Kruse (2010). We aggregated our national values in order to be comparable.
ME=Middle East (including Turkey), LAM=Latin America, Asia (including Afghanistan), FSU=Former Soviet Union, EUR=Europe, NAM=North America.

Scenario	**Kruse (2010)** Default Total	**This study** B2 Total
Year 2030		
World	**3083**	**2990**
Africa	2750	2505
ME	3167	3124
LAM	2958	2955
Asia	2917	2951
FSU	3500	3516
EUR	3833	3704
NAM	3792	3668
Year 2050		
World	**3250**	**3177**
Africa	2917	2783
ME	3250	3231
LAM	3083	3209
Asia	3208	3160
FSU	3917	3708
EUR	3917	3808
NAM	3917	3788

Table 11 Comparison of projections in kcal capita^{-1} d^{-1}. We have aggregated our national values in order to match the regional values. The AGO scenario (Agrimonde Global Orchestration) is based on the Global Orchestration scenario of the Millenium Assessment report.
SSA=Sub-Sahara Africa (including South Africa), MEA=Middle East and North Africa (including Turkey and Afghanistan), LAM=Latin America, Asia (including Afghanistan), FSU=Former Soviet Union, OECD (including Turkey).

Scenario	**Dorin et al. (2011)** AGO Total	Animal	**This study** A1 Total	Animal
Year 2050				
World	**3588**	**890**	**3566**	**861**
SSA	2972	305	3084	544
MEA	3457	470	3721	724
LAM	3698	940	3741	1047
Asia	3703	937	3477	807
FSU	3457	1366	4166	1503
OECD	4099	1714	4090	1235

For global average per-capita calorie demand in the year 2000, other studies give values of 2789 kcal capita^{-1} d^{-1} Alexandratos (1999), 2731 kcal capita^{-1} d^{-1} Valin et al. (2010) and 2708 kcal capita^{-1} d^{-1} Kruse (2010). As these studies have the same base year, namely 2000, the difference between the studies is probably caused by different download dates from FAO food balance sheets on which their values are based. The base year of our study is 1990, values for 2000 are therefore projections, which at 2736 kcal capita^{-1} d^{-1} meet the global values of the other studies quite well.

Our global estimate for per-capita calorie demand in 2030 is within a range of 100 kcal capita^{-1} d^{-1} of Alexandratos (1999), Valin et al. (2010) and Kruse (2010) for the B2 scenario. Valin et al. (2010) also report global estimates for other scenarios which are comparable to our values. While they project 2995, 3095 and 3135 kcal capita^{-1} d^{-1} for the scenarios A1, B2 and B1, we predict 2949, 2990 and 3062 kcal capita^{-1} d^{-1}, respectively. In 2050, our scenario B2

lies at 3177 kcal capita^{-1} d^{-1} between the results of Kruse (2010) at 3259 kcal capita^{-1} d^{-1} and Alexandratos (1999) at 3130 kcal capita^{-1} d^{-1}. Our A1 scenario estimate for 2050 is only 20 kcal capita^{-1} d^{-1} lower than Dorin et al. (2011). On a regional level, the results of our B2 scenario for per-capita calorie demand are in between the estimates of Alexandratos (1999), Kruse (2010) and Valin et al. (2010) for most regions. Only for Sub-Saharan Africa our estimates are lower, and for the transition countries of the Former Soviet Union we have slightly higher estimates.

Turning to animal-based demand, our global B2 estimate for 2030 is about 100 kcal capita^{-1} d^{-1} higher than in Valin et al. (2010) and Alexandratos (1999). This is due to the temporal increase in animal-based demand in developing countries. The livestock share is 20.9% of the total demand in our study, while it is only 15.5% and 14.6% for Alexandratos (1999) and Valin et al. (2010).

On a regional level, we find that Valin et al. (2010) and Alexandratos (1999) project an increase in animal-based calories up to 2030 for the OECD regions, while our study projects a decrease. When comparing our results for animal-based calorie demand in 2050 to Dorin et al. (2011), we have considerably lower estimates for the OECD countries. Therewith, our projection is able to reproduce the historical trend in North America, Europe and Pacific OECD, where the share of animal-based calories fell since the 1960s, 1980s and 1990s, respectively, and where the absolute amount of livestock calories stagnates or decreases. This pattern is most predominant in our sustainable B scenarios, while e.g. in our global and materialistic A1 scenario animal-based calories in OECD regions increase to 1181 kcal capita^{-1} d^{-1} in 2030. Only after reaching a peak value of 1235 kcal capita^{-1} d^{-1} in 2050 they do start to decline.

For countries in transition we project quite a steep increase in animal-based calorie demand until the year 2030 compared to other studies. Traditionally, countries in Central and Eastern Europe and Former Soviet Union had very high livestock demand in relation to their income (Liefert & Liefert, 2012). Due to our calibration to the base year 1990, this regional peculiarity persists in our approach into the near future. Furthermore, for the B2 scenario we assume that the animal-based calorie share is highest for medium incomes (function h_B) which applies to Former Soviet Union countries.

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Chapter III

The future of food demand: understanding differences in global economic models

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The future of food demand: understanding differences in global economic models

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Abstract

Understanding the capacity of agricultural systems to feed the world population under climate change requires projecting future food demand. This article reviews demand modeling approaches from 10 global economic models participating in the Agricultural Model Intercomparison and Improvement Project (AgMIP). We compare food demand projections in 2050 for various regions and agricultural products under harmonized scenarios of socioeconomic development, climate change, and bioenergy expansion. In the reference scenario (SSP2), food demand increases by 59–98% between 2005 and 2050, slightly higher than the most recent FAO projection of 54% from 2005/2007. The range of results is large, in particular for animal calories (between 61% and 144%), caused by differences in demand systems specifications, and in income and price elasticities. The results are more sensitive to socioeconomic assumptions than to climate change or bioenergy scenarios. When considering a world with higher population and lower economic growth (SSP3), consumption per capita drops on average by 9% for crops and 18% for livestock. The maximum effect of climate change on calorie availability is −6% at the global level, and the effect of biofuel production on calorie availability is even smaller.

JEL classifications: C63, C68, Q11, Q54

Keywords: World food demand; Socioeconomic pathways; Climate change; Computable general equilibrium; Partial equilibrium

1. Introduction

Agriculture has succeeded so far to respond globally to increased food demand from population growth. Food supply has more than tripled since the 1960s and continues to rise everywhere (FAO, 2011). But prospects for the future are uncertain

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Data Appendix Available Online

A data appendix to replicate main results is available in the online version of this article.

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as climate change and natural resource depletion threaten the capacity of agriculture to continue these trends in the long term. Simulating possible agricultural futures requires analytical tools that can represent world agriculture in a comprehensive way and reproduce the main structural drivers of demand and supply.

An important component of such quantitative analysis is modeling of consumer demand. Modelers have different perspectives, captured in their choice of behavioral parameters, on how future food consumption might evolve. But quantitative models also rely on specific mathematical functions to represent consumer behavior. This article, part of a series comparing results from the initial phase of the global economic model intercomparison of the Agricultural Model Intercomparison and Improvement Project (AgMIP) (von Lampe et al., 2014), examines how model demand specifications influence results. It provides a comparison of food demand projections across eight scenarios that vary by socioeconomic assumptions (GDP and population growth), crop productivity, and climate change for major agricultural commodity groups in 13 world regions through to year 2050. We consider the use of agricultural products as human food, ignoring crop use for feed and bioenergy.[1]

Demand for food is driven mainly by population growth, but also by income growth. Using food expenditure data across countries, Muhammad et al. (2011) find that the marginal share of income spent on food declines with countries ranked from low to high per-capita income. Income growth also leads to a change in consumption to a more diverse diet that includes a larger share of animal protein and fats and oils (a phenomenon known as Bennett's Law, see Bennett, 1941). China is an interesting example of diet transition, with a very rapid growth in per-capita income over the past two decades. Chinese per-capita consumption of livestock products has grown rapidly, while per-capita consumption of rice has declined slightly since the late 1990s, a pattern of consumption change following that of Japan in the latter part of the 20th century.

Commodity affordability also conditions access to food. Therefore, real prices constitute another important driver of food demand. High commodity prices not only directly impact food consumption in developing regions (Headey and Fan, 2008) but also consumption choices for final products in more advanced countries (Green et al., 2013).

Food demand is influenced by many other drivers, such as education, local traditions, degree of urbanization, trade liberalization, and development of downstream services such as supermarket chains and dining out (Kearney, 2010). Population, age, and gender structure as well as physical activity lead to different metabolic requirements and determine patterns of over- or underconsumption. The share of products wasted also increases food demand, especially in industrial countries (Gustavsson et al., 2011). FAO food balance sheets (FBSs) generally overestimate the amount of food actually consumed when compared to dietary surveys (Kearney, 2010). Most of the food demand systems used by models in this study explicitly consider income and price effects, with limited representation of other drivers.

[1] We follow the Food and Agricultural Organization of the United Nations (FAO) standard of reporting. Food demand then corresponds to food supply to households, including both actual food intake and household waste.

Table 1
Main characteristics of PE and CGE demand systems analyzed in this study

Demand system	PE	CGE
Theoretical representation	Reduced form	Utility based
Degrees of freedom	Unconstrained	Constrained by functional form
Commodity representation	Commodity	Final consumption good
Consumption metric	Quantity (FAO)	Volume in USD (GTAP)
Welfare approach	Consumer surplus kilocalorie intake	Compensating or equivalent variation of utility

2. Modeling demand in global models

Two classes of models have traditionally been used in developing forward looking scenarios for food at the world level: partial equilibrium (PE) and computable general equilibrium (CGE) models. These are reflected in the sample of models studied here (see von Lampe et al., 2014 for an overview of models characteristics). The main characteristics of PE and CGE demand systems are summarized in Table 1. Demand in all CGE models starts with a theoretically consistent utility function from which it is possible to derive demand functions, and functional forms for income and price elasticities. Demand functions are based on final consumption of household goods. All consumption items are included, exhausting the budget constraint. Because consumer utility is explicitly modeled, it is possible to calculate a change in welfare between scenarios as either compensating or equivalent variation.

PE models typically use reduced-form demand functions, which can be thought of as a local approximation of the full demand system, and are limited to a narrower set of goods, in their primary product form. They can therefore only compute a partial index of household welfare, such as consumer surplus or simply household food intake. PE models usually have a much greater level of detail at the commodity level (e.g., 26 crops in the IMPACT model), sometimes with a few stages of processing following the traditional supply utilization account structure (e.g., bioenergy in GLOBIOM).

Due to their origins in input–output models, CGE models typically include relatively few commodities but have a more detailed representation of the supply chain through various processing and intermediate activities between the producer and the consumer—restaurants, hotels, tourism, or even business or other service expenses such as school and hospital meals. As incomes rise, a growing share of household food consumption no longer relies on commodity consumption and is spent on

Table 2
Final demand specifications of the 10 global economic models compared in this paper

Model	Type	Demand regions	Food goods			Demand system	Price response	Inc. elast. dyn. adj.†	Data source	Parameter or data sources for calibration	
			Crop	Livestock	Processed					Price	Income†††
AIM	CGE	17	6	2	1	LES	Y	Y	GTAP/FAO	Implicit‡‡	FAO proj.
ENVISAGE	CGE	20	7	3	5	LES	Y	Y	GTAP	Implicit‡‡	FAO proj.
EPPA	CGE	16	1	1	1	Nested CES	Y	Y	GTAP	GTAP	GTAP
FARM	CGE	13	8	4	8	LES	Y	Y	GTAP/FAO	Implicit‡‡	USDA
GCAM	PE	16‡	13	5	–	Double-log	Y/N††	Y	FAO	USDA§§	FAO proj.
GLOBIOM	PE	30§	18	6	–	Double-log	Y	Y	FAO	USDA§§	FAO FBS & FAO proj.
GTEM	CGE	13	6	1	7	CDE	Y	Y	GTAP/FAO	GTAP	GTAP
IMPACT	PE	115‡	25	6	8	Double-log¶	Y	Y	FAO	USDA¶¶	USDA¶¶
MAGNET	CGE	45	8	3	8	CDE	Y	Y	GTAP	GTAP	GTAP & FAO proj.
MAgPIE	PE	10§	16	5	–	Econometric	N	Y	FAO	–	WB & FAO FBS

Notes: †Income elasticity dynamic adjustment.
‡More regions on supply side: GCAM (151 AEZ) and IMPACT (251 FPU).
§Gridded models for biophysical parameters at 0.5° resolution.
¶With cross-price effects.
††Price response for livestock products only.
‡‡In an LES calibrated on income elasticities, price elasticities are endogenously determined.
§§USDA estimates from Seale et al. (2003) and Muhammad et al. (2011).
¶¶Sourced from the USDA literature review database (USDA, 1998).
†††For USDA, income elasticities estimate are directly used. GTAP provides calibration parameters for the CDE. For other sources, modelers use their own estimation on past time series of FAO (2011), World Bank (2011) or future projections from FAO (Alexandratos and Bruinsma, 2012).

processed foods and beverages and food consumed outside of the house.

The AgMIP comparison includes six CGE models and four PE models. The characteristics of the food demand systems are summarized in Table 2. All models compute demand using a representative household for each of the modeled regions.[2] The behavior of each demand system is then driven by the choice of functional form and its parameterization.

2.1. Food demand systems in PE models

Two approaches to modeling food demand in the PE models in this article are used. The standard approach has per-capita food demand expressed as a function of per-capita income and a vector of all prices (in the IMPACT, GLOBIOM, and GCAM models). Food demand is calculated as follows:

$$D_{r,c,t} = Pop_{r,t}\left(\frac{Y_{r,t}}{Pop_{r,t}}\right)^{\eta_{r,c,t}} \prod_{c'} \left(P_{r,c',t}\right)^{\varepsilon_{r,c,c',t}}, \quad (1)$$

where D is food demand for commodity c in region r in year t, Pop is population, Y is total income, P is the vector of commodity prices, η are the commodity specific income elasticities, and ε is the matrix of own-price and, for IMPACT, also cross-price elasticities.[3] This functional form is based on a representative household model, but it can be readily adjusted to allow for household heterogeneity. In these models, population and income growth are exogenous and prices are endogenous. When looking forward to 2030 or 2050, one of the key questions is how income and price elasticities evolve. For example, there is overwhelming evidence that income elasticities for most food commodities decline as consumer income increases, a relationship known as Engel's Law. Models should therefore have varying income elasticity values depending on the level of development of their countries. This is done by having an exogenous trend depending on time (IMPACT, GCAM) or on income evolution (GLOBIOM).

The second variant, used by MAgPIE is to compute the demand for calories *ex ante* and to use it as a constraint for the model optimization (Bodirsky et al., in review). Total calorie demand D in MAgPIE is estimated on the basis of population Pop, income Y, and time t for each region r according to the relation:

$$D_{r,t} = Pop_{r,t} F\left(\frac{Y_{r,t}}{Pop_{r,t}}, t\right) = Pop_{r,t}\alpha(t)\left(\frac{Y_{r,t}}{Pop_{r,t}}\right)^{\beta(t)}, \quad (2)$$

[2] A region can be a subregional part of a country, a country, or an aggregation of countries, depending on the model and the part of the world. The aggregated regions used in this paper are reported in the Data Appendix, Section 4.2.

[3] Neither GCAM nor GLOBIOM have cross-price effects in their standard versions. Further, GCAM has only own-price effects in the livestock sectors with limited cross substitution across crops, using a logit choice function. Some cross-price substitution effects can be obtained in GLOBIOM using an extended version of the model with a hard-link to a nonlinear demand module (Valin et al., 2010).

where the $\alpha(t)$ and $\beta(t)$ parameters are econometrically estimated based on a panel data set of per-capita demand (FAO, 2011) and per-capita income (World Bank, 2011). Nonincome-related processes that shape demand are represented by the time-dependent parameters $\alpha(t)$ and $\beta(t)$, leading to a positive time-trend that declines over time. The share of calories from animal-based products in total food demand (*LS*) is estimated separately based on per-capita income and time t. Historical developments of food demand (FAO, 2011) show that the share of animal-based products has increased in low- and medium-income countries, but declined in high-income countries. Therefore, the following functional relationship was chosen:

$$LS = G\left(\frac{Y_{r,t}}{Pop_{r,t}}, t\right) = \rho(t)\sqrt{\frac{Y_{r,t}}{Pop_{r,t}}}e^{-\frac{Y_{r,t}}{Pop_{r,t}}\sigma(t)}, \tag{3}$$

where the positive parameters $\rho(t)$ and $\sigma(t)$ were also estimated econometrically based on the above panel data set. The term $\rho(t)\sqrt{\frac{Y_{r,t}}{Pop_{r,t}}}$ increases strongly for low per-capita incomes, but stagnates for high incomes, while the term $e^{-\frac{Y_{r,t}}{Pop_{r,t}}\sigma(t)}$ approaches zero for very high per-capita incomes. The combination leads to a function resembling an inverted U-shape. Prices do not enter Eqs. (2) and (3). Hence, this modeling approach focuses more on where and how food is produced and does not consider the effects of supply shocks such as climate change or biofuel policies on food demand.

Models using explicit income elasticities can also use their own estimates of these parameters. For example, income elasticities in GLOBIOM are defined as rational functions that are calibrated to the following constraints: (i) the base year value should match USDA econometric estimates on past data (Muhammad et al., 2011), (ii) the level of total calorie consumption per capita should converge to advanced countries intake when income per capita reaches similar levels of development, and (iii) composition in product consumption should correspond to future diet preferences such as defined for a given scenario (level of red or white meat consumption, share of sugar, fat, etc.). This approach results in the production of a set of Engel curves for each region and associated elasticities for each good, depending on assumptions about future preferences.

2.2. Food demand systems in CGE models

As noted, food demand in the CGE models of this study is based on utility functions that are consistent with microeconomic theory and thus are consistent with an overall budget constraint (adding up properties). However, without some form of dynamic recalibration process, none are consistent with the stylized facts of consumer behavior over multiple decades with high per-capita growth where the food budget share declines.

The workhorse utility function for CGE models has been the Linear Expenditure System (LES), also referred to as the Stone–Geary utility function.[4] It is used by three of the AgMIP CGE models (AIM, ENVISAGE, and FARM). The starting point for the LES is the following expression for utility:

$$u_{r,t} = \prod_c \left(d_{r,c,t} - \gamma_{r,c,t}\right)^{\mu_{r,c,t}}, \tag{4}$$

where u is utility, d is per-capita consumption, and μ and γ are parameters of the utility function. The γ parameters are often referred to as the subsistence minima, or floor consumption. Maximizing u subject to the standard budget constraint leads to the following demand function:

$$d_{r,c,t} = \gamma_{r,c,t} + \frac{\mu_{r,c,t}}{P_{r,c,t}}\left[y_{r,t} - \sum_{c'} P_{r,c',t}\gamma_{r,c',t}\right],$$
$$\Downarrow$$
$$D_{r,c,t} = Pop_{r,t}\gamma_{r,c,t} + \frac{\mu_{r,c,t}}{P_{r,c,t}}\left[Y_{r,t} - Pop_{r,t}\sum_{c'} P_{r,c',t}\gamma_{r,c',t}\right], \tag{5}$$

where y is per-capita expenditure on goods and services.[5] Demand is the sum of two components—the subsistence minima γ, and a share μ of residual expenditures after aggregate expenditures on the subsistence minima, often referred to as supernumerary income.

Though the LES is widely used, it has dynamic properties that are clearly contradicted by empirical evidence. As the above equation shows, with constant γ parameters, the LES converges toward a Cobb–Douglas utility function with unitary income elasticities. The LES also has minimal flexibility in determining price elasticities, so even for comparative static exercises it might be less than ideal.

As an alternative, the EPPA model uses a nested constant-elasticity-of-substitution (CES) structure to describe consumption preferences, that is, a combination of several functions of the form:

$$u_{r,i,t} = \left(\sum_j A_{r,i,j}.d_{r,i,j,t}^{\rho_i}\right)^{1/\rho_i}, \tag{6}$$

where for a given region r, $u_{r,i,t}$ is the utility associated to the food bundle i, $d_{r,i,j,t}$ is the demand of good j in the bundle i, and $A_{r,i,j}$ are calibration parameters. The price elasticity of food demand is therefore determined by the different elasticities of substitution in the nesting, defined as $\sigma_i = 1/(1 - \rho_i)$ in each nest i. The elasticity of substitution between food and nonfood goods, and the food consumption share are updated as a function of per-capita income growth between periods to reproduce the dietary changes as indicated by Bennett's Law (see Lahiri et al., 2000, for a detailed discussion).

Another commonly used utility function—popularized by the wide use of the GTAP model—is the Constant Differences in

[4] See, for example, Sadoulet and de Janvry (1995).

[5] Abstracting from savings.

Elasticities (CDE) utility function.[6] Starting from an indirect utility function, the CDE demand function takes the following form:

$$d_{r,c,t} = \frac{\alpha_{r,c,t} b_{r,c,t} u_{r,t}^{e_{r,c,t} b_{r,c,t}} \left(\frac{P_{r,c,t}}{y_{r.t}}\right)^{b_{r,c,t}-1}}{\sum_{c'} \alpha_{r,c',t} b_{r,c',t} u_{r,t}^{e_{r,c',t} b_{r,c',t}} \left(\frac{P_{r,c',t}}{y_{r.t}}\right)^{b_{r,c,t}}},$$

where d, P, y, and u have the same interpretation as mentioned above and the key response parameters are represented by b and e. The b coefficients are linked to own- and cross-price substitution effects; the e parameters govern the responsiveness of demand with respect to income. It is still a relatively parsimonious functional form, although it allows for significantly more realistic price elasticity response than the LES. Nonetheless, in the absence of a dynamic recalibration of the price and income parameters, CDE income responsiveness is limited and final-year income elasticities are close to their initial levels. This functional form is used by several of the GTAP-based models participating in AgMIP (MAGNET, GTEM). Preckel et al. (2005) provide extensions to the CDE class of expenditure functions that introduce minimum quantities in the utility function, as in the LES.

Although rarely adopted for modeling demand in CGE models, other more flexible functional forms are also used in the modeling literature. These functions are not directly derived from a utility function but they allow for a broad range of price and income response. For instance, Jorgenson and associates have made extensive use of the translog functional form and have provided econometric estimates of the various parameters of this function (Jorgenson et al., 2013).[7]

Lewbel (1991) has characterized utility functions by their rank number. A utility function of rank 1 has expenditure shares for each good invariant with the level of income. The utility functions described are all of rank 2, that is, their expenditure shares vary but Engel curves remain linear. In the absence of adjustments, a utility function of rank 3, that is, having nonlinear Engel curves, is needed to appropriately deal with plausible dynamic behavior.

Relatively minor adjustments to the functions mentioned above have led to rank 3 demand systems. One, known as An Implicit Directly Additive Demand System (AIDADS) and first developed by Rimmer and Powell (1996), makes the marginal consumption parameter of the LES a function of utility and no longer constant.

$$d_{r,c,t} = \gamma_{r,c,t} + \frac{\mu_{r,c,t}}{P_{r,c,t}} \left[y_{r,t} - \sum_{c'} P_{r,c',t} \gamma_{r,c',t} \right],$$

where

$$\mu_{r,c,t} = \frac{\alpha_{r,c,t} + \beta_{r,c,t} e^{u_{r,t}}}{1 + e^{u_{r,t}}}. \tag{7}$$

AIDADS collapses to the familiar LES when $\alpha = \beta$. Although it has better dynamic characteristics, AIDADS still suffers from the constrained own- and cross-price elasticities compared with other alternative functional forms.

Addition of quadratic terms in per capita incomes to the translog functional form also provides the extra "curvature" needed to make these rank 3 demand systems—though the additional terms do not solve the problem of the domain of applicability, that is, the budget shares can wander outside the 0–1 range.[8]

Despite recent literature introducing new utility functions with more desirable dynamic properties, their use to date has been relatively limited in empirical models. One of the limiting factors has been the sparseness of available price and income elasticities—needed for the calibration of these functional forms—particularly at the level of disaggregation typically used by a GTAP-style model, both in terms of regional and commodity coverage (see Yu et al., 2004 for an application). A major challenge for the CGE modeling community is to address this gap by providing utility functions that can better track demand behavior over a wide spectrum of income changes, and empirically estimate income and price elasticities to provide a basis for calibration.

Faced with the deficient dynamic behavior of the standard utility-derived demand functions—even if meeting the regularity conditions—most modelers resort to pragmatic approaches that focus on using these simpler utility functions, but shifting the functions' parameters over time to reflect best judgment on the evolution of either budget shares or income elasticities. What this typically involves is some notion of the evolution of income elasticities over time, and then recalibration of functional parameters between solution periods, based on the model solution for the just solved-for period, to line up with a desired path for the income elasticities.

2.3. Commodity supply chain challenges

PE models have a simplified structure of food supply, as they only represent the supply chain through the primary products processing of the FAO supply utilization accounts (oilseed crushing, sugar refining, etc.). In CGE models, the full supply chain representation creates additional challenges when income increases and consumption move to more complex products. For example, in most high-income countries there is very little or no direct consumption of cereal grains—wheat consumption is in the form of bread and other bakery items such as breakfast cereals, pasta, and pizza. Reproducing the growth of household demand for wheat from an external projection—such as from the FAO's long-term scenario reports—would require tracking the indirect consumption of wheat through the input–output

[6] See, for example, Hertel (1997).

[7] Another often-used flexible functional form is the Almost Ideal Demand System (or AIDS), typically used in econometric estimation of price and income responses (Deaton and Muellbauer, 1980).

[8] See Jorgenson et al. (2013) and Cranfield et al. (2003).

table. Moreover, processed goods tend to have higher income elasticities than raw commodities and thus the derived demand for the raw commodities may lead to higher trend growth than warranted. The problem is exacerbated by the food demand embodied in services demand, where income elasticities are typically greater than 1.

Two solutions can be implemented in CGE models to deal with these problems. The first is to introduce a trend on the input–output coefficient of the relevant commodity. For example, even if the income elasticity for processed foods is relatively high, over time one would expect that a greater portion of the value of that demand will represent the added value to the raw commodity (labor, capital, transport, packaging, advertising, etc.). Thus over time, the input of wheat in the value of baked goods or other wheat-based products declines. A second solution is to reconfigure the base data—either partially or wholly. A simple partial reconfiguration is to move the food items of the input–output table to household consumption (and reduce the relevant part of household consumption by the same amount).

The ENVISAGE model, for example, incorporates a mix of these strategies. The database reconfiguration moves all food consumed in the service sectors to household demand—and adjusts downwards by the same amount household demand for services. Food demand on a regional basis (as expressed in raw agricultural commodities) is constrained in the reference scenario to line up with the FAO long-term projections. Two sets of parameters are adjusted to achieve these constraints. Both the LES parameters (the subsistence minima and the marginal budget share parameters) and the agricultural input–output coefficient of processed foods are adjusted. The allocation of the adjustment across these two sets of parameters is driven by an *ad hoc* assumption about the share of direct household consumption of raw agriculture relative to the processed share, with typically the processed share increasing with income.

3. Demand system parameters

Each demand system contains a certain number of parameters that need to be initialized in a calibration phase. The specific functional form chosen for the utility function (and therefore the demand functions) dictates to some extent the requirements needed to calibrate the demand system under the usual assumption that the functional form is able to reproduce some base year data set. Thus each demand system has a different degree of freedom for its calibration that can also limit the range of potential behavior. The demand systems can also differ by the type of parameters they are calibrated on. For example, the CES only allows one calibration parameter (the elasticity of substitution), and therefore a nested CES structure such as the one in EPPA will have the same degrees of freedom as the number of nests. The LES has degrees of freedom equal to the number of products included in the demand function. Therefore, when calibrating an LES, a trade-off must be done between fitting a set of price elasticities or income elasticities. The most flexible form in terms of parameterization is the specification found in IMPACT, that allows each product to have unique income, own-price and cross-price elasticities. This approach has the advantage of its large degree of freedom (for n products, $n + n\,(n+1)/2$ parameters), although substitution patterns can be affected for large shocks if cross-price elasticities are kept fixed.[9] This is also the case with the flexible functional forms found in some CGE models such as the translog and AIDS functions.

In order to calibrate their systems, the different modeling teams have relied on different sources, an overview of which is provided in Table 1. For income elasticities, five out of ten models (two PEs and three CGEs) use the FAO food demand projections (Alexandratos and Bruinsma, 2012) to calculate their income elasticities; hence these models should have relatively similar food demand trends. Among the five other models, FARM uses income and price elasticities from a USDA data set (Muhammad et al., 2011). IMPACT elasticities rely on a more empirically grounded database from USDA (1998) that compiles a large number of regional econometric studies. Finally, MAgPIE also produces its own elasticities drawing on both the FAO FBS database and historical consumption data reported by the World Bank.

Interestingly, only three sources have been used for price elasticities. For PEs, two models used USDA (Muhammad et al., 2011) to target their price responses, although only for livestock in the case of GCAM. IMPACT again relied on the USDA literature survey (USDA, 1998); for CGEs, models with a CDE system have used the GTAP parameterization but adjusted it only with respect to income behavior. In the case of LES-based CGEs, price elasticities are derived endogenously once income elasticities are determined. The two elasticities are therefore structurally correlated and a commodity with a high income elasticity will necessarily have a high price elasticity.

Income and price elasticity magnitudes are reported in Fig. 1 for food commodities.[10] EPPA is the model with the largest spread in income elasticities and some of the highest values, in spite of its aggregated product representation. ENVISAGE and MAGNET show less dispersion but have the highest mean value in their elasticity distribution, closely followed by IMPACT. GCAM, FARM, and AIM display lower values and dispersion when compared with others. An interesting pattern is that only four of the ten models report some negative income elasticities, mostly PEs and CGEs based on the CDE functional form, as well as EPPA. As expected, price elasticities are correlated in magnitude with income elasticities. This comes as a direct effect of the functional form constraints in degrees of freedom (CES, LES), or from the data used for calibration.

[9] Cross-price elasticities are calculated with respect to an initial structure of food consumption to represent a certain degree of substitution patterns when relative prices are changing. Therefore, these elasticities need to be recalculated when shares of good in final consumption become different.

[10] For CGEs, these elasticities correspond to direct demand elasticity of commodity products and do not account for the indirect food consumption through the processing chain.

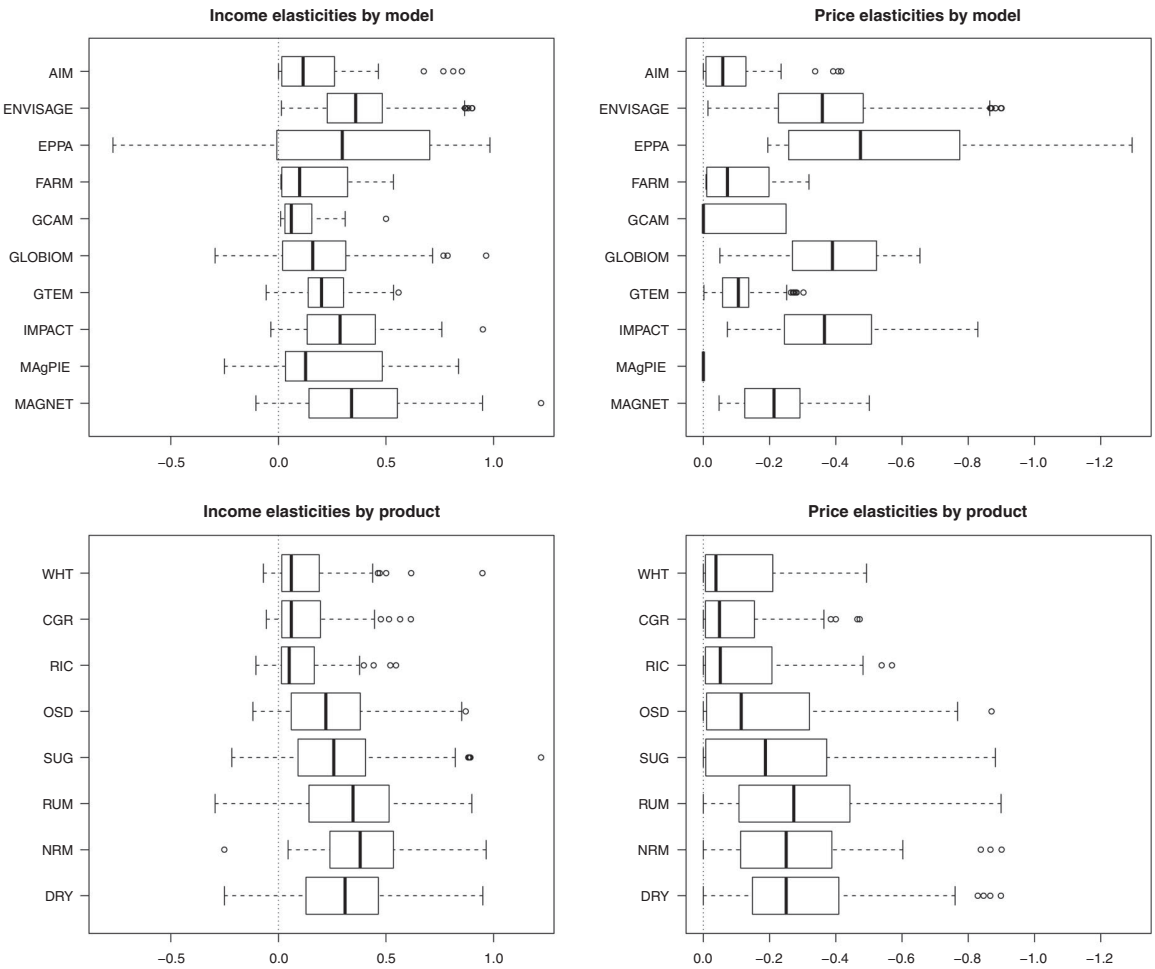

Fig. 1. Base year income and price elasticities by model and by product as reported by modeling teams. For PE models, elasticities are reported as they are exogenously fed in the model; for CGEs, elasticities are inferred from formulas based on calibration parameters, and estimations can be less accurate. Elasticities of EPPA are not represented for the representation by product because only two aggregates are available for this model. Boxes represent the first to third quartile range and the plain line indicates the median; dotted lines delineate the first and fourth quartile points up to 1.5 times the interquartile range of the box and bullets represent outliers.

Some interesting counterexamples are found in the group of PE models. Two of them have no sensitivity to prices (MAgPIE and, for crops, GCAM), whereas two others have the highest average values after EPPA (IMPACT, GLOBIOM). We discuss how these patterns can explain differences in projections in the next section.

4. Comparison of food demand projections from AgMIP agroeconomic models

We can now compare the food demand results from the 10 AgMIP models for the period 2005–2050. Our analysis follows the three dimensions of the AgMIP scenarios: socioeconomics, climate change, and bioenergy (see von Lampe et al., 2014). Note that climate change and bioenergy results are more extensively explored in separate papers (Nelson et al., 2014, for climate and Lotze-Campen et al., 2014, for bioenergy).

4.1. Food projections toward 2050 for a "Middle of the Road" scenario (SSP2)

The reference scenario, S1, uses the GDP and population pathways of the "Middle of the Road" Shared Socio-economic Pathway (SSP2) developed by the climate change impacts research community (O'Neill et al., 2012). This scenario, quantified by OECD and IIASA, leads to a world population of 9.3 billion by 2050 (42% higher than the 2005 level) and more than a doubling in average income per capita globally, from 6,700 USD in 2005 to 16,000 USD in 2050.

Global food projections by 2050 associated with SSP2 can be seen in the first column of Table 3. The average demand increase for all models is 74%, ranging from 62% to 98%. All model projections are higher than the value of 54% projected by FAO (labeled "AT2050"). This difference cannot be explained by population growth as both FAO and this exercise have

Table 3
Decomposition of food demand change by 2050 in the SSP2 scenario between population, price, and income effects (percent change, except for price index)

Model	All	Crops					Livestock				
	Total food change (1)	Total food change (2a)	Food per cap change (3a)	World price index (4a)	Price effect (5a)	Income effect (6a)	Total food change (2b)	Food per cap change (3b)	World price index (4b)	Price effect (5b)	Income effect (6b)
AIM	66	62	13	1.21	−7	22	88	32	1.12	−17	59
ENVISAGE	70	65	15	0.93	6	9	94	36	0.90	15	19
EPPA	79	82	28	0.80	14	12	62	14	0.86	18	−3
FARM	98	97	38	0.85	0	38	102	41	0.97	0	41
GCAM	59	55	8	0.93	0	8	79	25	1.04	0	25
GLOBIOM	62	57	10	1.00	0	11	84	29	1.06	−2	31
GTEM	94	84	29	1.04	0	29	144	71	0.80	1	69
IMPACT	65	63	14	1.31	−7	23	78	25	1.03	−5	31
MAgPIE	83	55	8	1.54	0	8	242	140	1.04	0	140
MAGNET	65	66	16	0.93	1	15	61	12	0.85	5	7
AT2050†	54	50	8	NA	NA	NA	76	27	NA	NA	NA

Notes:†"Agriculture Towards 2050" (Alexandratos and Bruinsma, 2012).
Calculation method:

(1), (2a), (3a), (2b), (3b): Aggregated on a calorie basis for the five crop categories considered or the three livestock products.
(4a), (4b): Based on model reported values. For CGEs, the world price index is deflated by the world consumer price index.
(5a), (5b): Calculated at the product level using the price index and the price elasticities reported by models.
(6a), (6b): Obtained by subtracting the price effect from (5a) and (5b) from the change per capita (3a) and (3b).

similar population growth assumptions.[11] The differences in food consumption are driven in large part by the differences in economic growth assumptions. The SSP2 scenario assumes that income per capita in 2050 will be 50% higher than does FAO (16,000 USD per capita as a world average vs. 11,000 USD per capita in the FAO scenario, based on World Bank projections). For example, in SSP2 China and India per-capita GDP increase 13 and 11 times, respectively, whereas FAO assumes 7 times and 4 times increase.

But the greater income effect in SSP2 only explains a part of the differences observed. To better analyze the source of differences, we can decompose the results between contribution of income effects and price effects for crop and livestock products by adjusting the projections by the price responses and the price demand elasticities reported by the different models. Results are displayed in Table 3. First, for some models the overall high growth in food demand is related to demand for livestock products (+144% for GTEM, +136% for MAgPIE). Only FARM and EPPA have strong expansion of food consumption in both crops and livestock products. The second interesting source of difference comes from the role of prices. As observed in von Lampe et al. (2014), the models have different price trends for the SSP2 baseline. Some price changes can compensate or exacerbate the effect of income response. For instance, the IMPACT food consumption of crops per capita increases only by 14% by 2050, whereas it would have risen by 23% without the price effect. ENVISAGE crop and livestock demands reach levels similar to those of AIM, whereas their income responses are initially very different (for livestock, +59% for AIM vs. +19% for ENVISAGE). This is the result of decreasing prices in ENVISAGE and increasing crop prices in AIM. Some models have very little sensitivity to price changes. GCAM and MAgPIE are, by assumption, price inelastic for some or all of their products.

4.2. Product-specific and regional differences across models

We have so far looked at the differences in results between crop and livestock categories. In fact, five crop aggregates (wheat, rice, other coarse grains, oilseeds, sugar crops) and three livestock sectors (ruminant meat, nonruminant meat, and dairy) were included in the analysis (see the Data Appendix for each model mapping). The product-specific results at the world level are presented in Fig. 2.

Cereals constitute a strong part of global crop consumption and we observe again the pattern mentioned above of greater consumption increase under SSP2 than in the FAO baseline. Food demand for the AgMIP models increases for wheat, corn, and rice by an average 53%, 106%, and 47%, respectively, between 2005 and 2050 in SSP2, whereas it only rises by 34%, 68%, and 30%, respectively, for FAO.[12] Oilseeds and sugar consumption growth (83% and 75%, respectively) are much higher than for wheat or rice, and this holds for most of the models, as well as for the FAO projection. This is consistent with the values of income elasticities in demand systems for these commodities, which are on average higher than for cereals (see Fig. 1). The case of coarse grains is interesting as it has the highest growth rate globally of all crop categories

[11] The FAO scenario from AT2050 has a population of 9,150 million by 2050 whereas the IIASA SSP2 scenario projects 9,287 million.

[12] For FAO results, we report the figures obtained by processing FAO detailed projections at product level using the same methodology as for other models. Therefore, results can vary slightly from the published version of FAO projections (Alexandratos and Bruinsma, 2012).

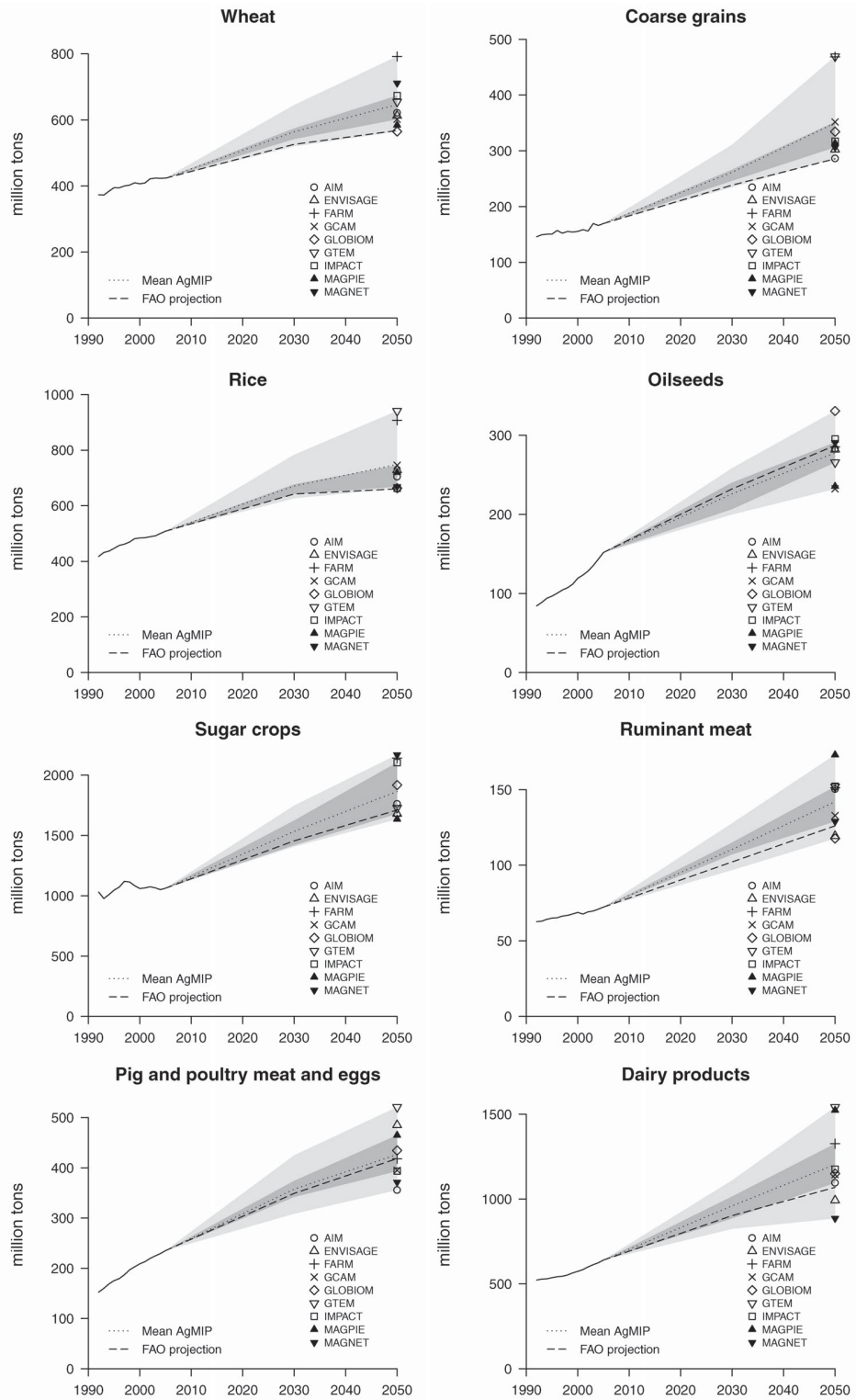

Fig. 2. World food demand projection for SSP2 scenario by 2050 for the different models, by product category, in raw primary equivalent. Black plain line corresponds to historical data in FAOSTAT. Dashed line corresponds to FAO projections (Alexandratos and Bruinsma, 2012). Dotted line corresponds to mean of model results. Light gray indicates the span of results and dark gray the first to third quartile range.

but a low average income elasticity. This large global growth is determined in large part by the high share of global maize food consumption (33%) in Africa and Middle East and the importance of sorghum and millet in this part of the world. This region has the fastest population growth in the SSP2 scenario (+130% increase between 2005 and 2050).

Livestock products have the largest average income elasticities, which explains the trends observed in Table 3. But the range of values is large across models and associated food projections differ greatly. This is reflected by the large range of income elasticities associated with these different products, for which consumers in many countries have a preference when their budget allows. Model differences reflect this uncertainty with a 50% higher dispersion around the mean than for crops (Fig. 2).

As we have just seen in the specific case of coarse grains, the diversity of results across products is also related to differences in results across regions. Figure 3 presents the results for diet evolution in the different regions, expressed in kcal per capita, aggregated for the crop and livestock categories. We also calculate the implicit income elasticities associated to these estimates for each of the aggregates (Fig. 4).[13]

As expected, high-income OECD countries and regions in transition (Former Soviet Union, FSU) start from the highest level of intake, which does not change much over time. Two notable exceptions are EPPA where crop consumption increases sharply (+34%), and for MAgPIE livestock, where a significant decrease of consumption per capita (−13%) takes place. MAgPIE's meat decline is the result of the econometric estimates (as mentioned above) that lead to implicit income elasticities of −0.37 for Europe and −0.79 for North America. EPPA also has strongly negative values for these regions for both livestock and crops (−0.4 in Europe and −0.5 in North America), resulting from the nested CES recalibration technique. However, on the crop side, the negative income effect in EPPA is counteracted by an implicit income elasticity of 0.75 for FSU, which overall leads to a consumption increase in crops.

For developing regions, the dispersion in results is much wider. This is in part the result of larger income per capita changes for these regions that exacerbate divergence due to income elasticities. Some models reach daily kcal consumption levels in some regions of close to 3,000 kcal per capita from crop consumption alone. This level is equivalent to 80% of the current caloric intake in the United States. The range of results is the largest for livestock products. This does not come as a surprise considering the large uncertainty about future demand in China, where meat and dairy products are experiencing rapid consumption growth driven by fast economic growth; or in India, where very low consumption of meat and dairy due to traditional culture gives uncertain prospects on the effect of western influence in the future (Alexandratos and Bruinsma, 2012). For instance, for developing regions, average implicit elasticities for livestock products range in Fig. 4 from 0.1 (EPPA) to 0.6 (MAgPIE). This range seems plausible when compared with values corresponding to Alexandratos and Bruinsma (0.3–0.7 for the first to third quartile). As average income per capita in developing countries increases by a factor of 4 on average under SSP2, corresponding uncertainty for livestock consumption ranges from +15% to +130%. And the choice of elasticity is even more important for countries such as China or India where income per-capita growth is faster (multiplied by 13 and 11, respectively, between 2005 and 2050).

4.3. The role of socioeconomic assumptions

Results in the previous section were based on GDP and population pathways from SSP2. To illustrate the uncertainty in future demand related to macroeconomic drivers, we can contrast these results with those obtained in the SSP3 scenario ("Fragmented world"). The SSP3 scenario generally has greater population growth and slower income growth. The population level in 2050 is 10.2 billion in SSP3 instead of 9.3 in SSP2, with the greatest increases in Africa, India, and South-East Asia. World GDP in SSP3 is only two-thirds of that in SSP2 by 2050. The greatest differences in income are in China, India, Africa, and South-East Asia where growth rates in SSP3 are roughly half of those in SSP2. OECD countries also experience lower income growth in SSP3 but their populations grow more slowly, which results in smaller differences in per-capita income than for the non-OECD countries. For example, per capita income in 2050 is 5% smaller in SSP3 for the USA but 46% smaller in China, 50% smaller in India, and 52% smaller in Sub-Saharan Africa.

Changing the socioeconomic scenario has different effects across models. The population increase is similar for all models but the demand response to changes in income per capita depends on income elasticities that vary across the models. Additionally, differences in prices responses can also influence the level of consumption.

The results reflect the differentiated effect of income per-capita shocks between developed and developing countries (see Fig. 5). Three different patterns are observed for almost all models. In the developed countries, total consumption declines in SSP3 relative to SSP2 (−14%), because income and price elasticities are lower and the effect of population decrease dominates.[14] In developing countries, total demand for crops generally increases, but not for all models. Population growth is indeed larger, and remains the dominating effect, except for some models where the income effect is stronger: FARM, for

[13] For the model results, this is calculated by adjusting the projection to remove the effect of price changes. For the FAO results, price projections are not known so the price effects cannot be removed.

[14] The GTEM crop results for OECD and FSU are outliers. This is due to a different response to the decline in total factor productivity in SSP3 relative to SSP2. The productivity decrease is distributed to uses of intermediate inputs by industries, including the food processing sector. As a consequence, increased inefficiencies in indirect food use of crops and livestock products lead to additional demand for commodity products under SSP3 that offsets the reduction in household use of crops and livestock products.

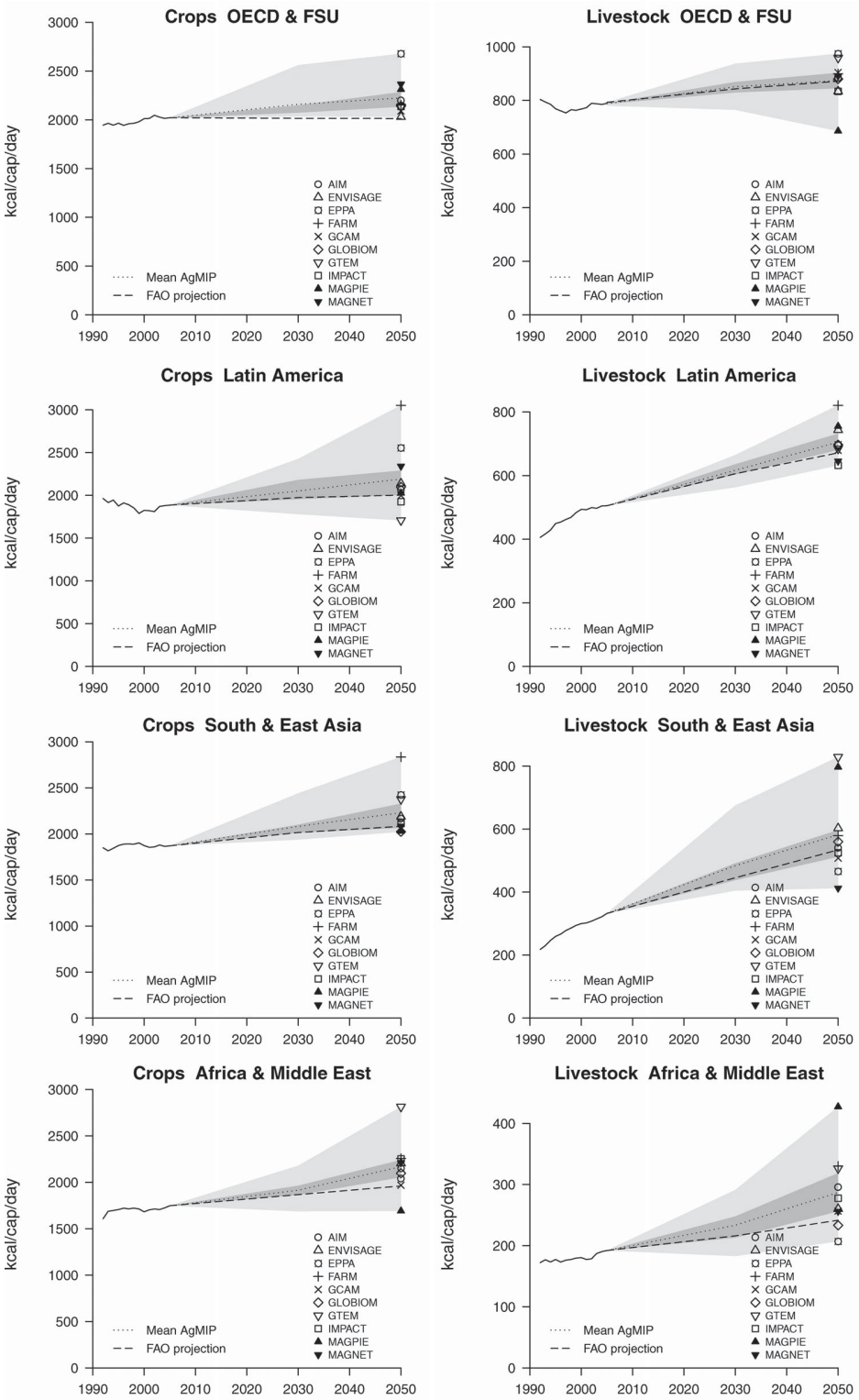

Fig. 3. World food demand per capita projection for SSP2 by 2050 for the different models, by region. The black plain line corresponds to historical data in FAOSTAT. The dashed line corresponds to FAO projections (Alexandratos and Bruinsma, 2012). The dotted line corresponds to the mean of model results. Light gray indicates the span of results and dark gray the first to third quartile range.

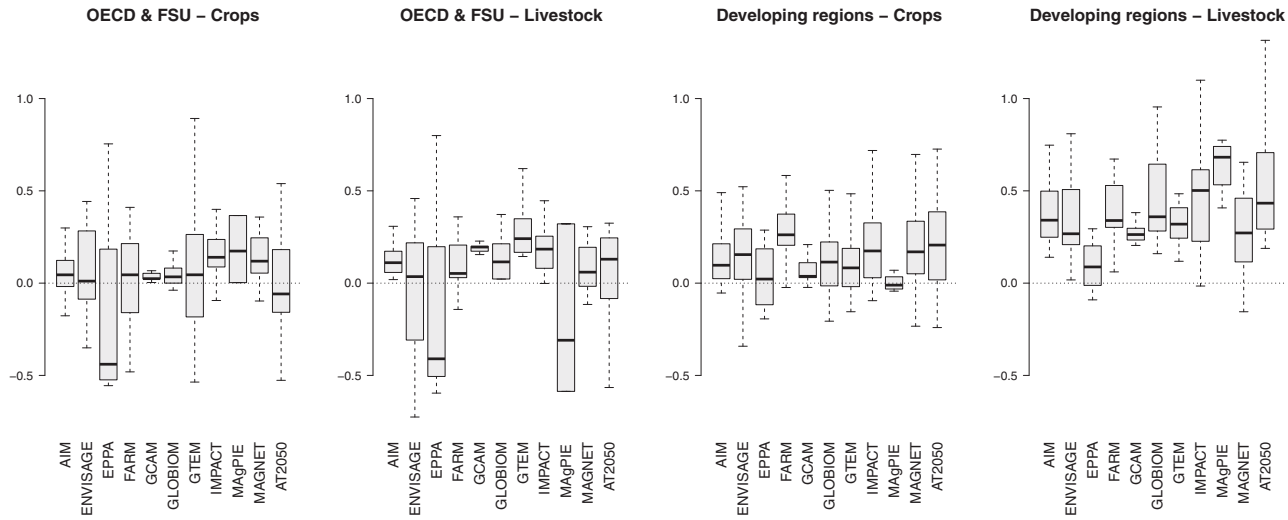

Fig. 4. Implicit income elasticities in the SSP2 scenario for the period 2005–2030. Implicit elasticities are defined as the log of food demand divided by the log of change in income per capita, after correction for price effect. For each region group, implicit elasticities are plotted by model for the five crops or the three livestock products. Boxes represent the first to third quartile range and the plain line indicates the median; dotted lines delineate the first and fourth quartile points up to 1.5 times the interquartile range of the box. For clarity, outliers are not represented in this figure.

example, has higher income elasticities on crops in developing countries than other models (Fig. 1). Income elasticities in ENVISAGE and EPPA are lower in SSP3 than in SSP2 (see Fig. S4 in the Data Appendix). The third pattern observed is a general decrease of total demand for livestock in developing regions, due to the higher income elasticities for these products (Fig. 1). The effect of lower economic growth in these regions generally dominates the population effect, and meat and dairy consumption decrease by 11% in Asia and by 8% in Africa.

Projection uncertainty therefore depends both on the models and the macroeconomic scenario considered. The comparison between these two dimensions of uncertainty can be observed in Fig. 6 that shows the standard deviation across models observed at the world level, decomposed by its regional contribution. This information is displayed for each of the food categories and for the two macro-economic scenarios. The span of projections appears rather limited for wheat and rice. The standard deviation is small for these products and contribution to uncertainty is evenly distributed across all regions for wheat and developing regions for rice. Much more uncertainty is associated with coarse grain projections, and this is highly linked to demand in Africa which accounts for more than half the standard deviation in world consumption. For all crops, the uncertainty across models is much higher than the uncertainty across socioeconomic scenarios, which is consistent with the previous observation that, for most models, the population and income effects in developing regions tend to compensate for crops between SSP2 and SSP3. FAO projections always appear at the lower bound of projections, except for oilseeds. For livestock products, the standard deviation across models is higher than for crops and the macro-economic assumptions play a more significant role. Most of the uncertainty comes in particular from China and India for nonruminant meat consumption as well as for dairy products in India due to their high income growth. Scenario and model-driven uncertainty are now both comparable for livestock products, and the range of outcomes is a wide band above and below the FAO projections.

4.4. Sensitivity of demand projection to climate change

Using the climate change scenarios from AgMIP, we can illustrate the effect of a pure supply shock on the food demand response for nine of the models.[15] We look at the impact of RCP 8.5, the scenario with the largest greenhouse gas emissions, with outputs from two different general circulation models (IPSL-CM5A-LR and HadGEM2-ES) as inputs into two crop models providing their impact on crop yields (LPJmL and DSSAT). No incremental CO_2 fertilization effect is considered for the crop model runs (see Nelson et al., 2014, for more details).

The climate change shock affects the production side and therefore prices faced by the consumer. For CGE models, they also can impact the representative agent through an income effect but, except in regions where agriculture represents a large share of value added, this feedback is second order.[16] Crop demand and price responses to climate shocks are illustrated in Fig. 7 for the four climate scenarios. The diversity of demand responses is striking. First, the price-inelastic response

[15] Climate change scenario results were not available for the EPPA model.

[16] All climate change scenarios are run under SSP2 that assumes a certain level of GDP growth. For PEs, the income level is exogenous and related to GDP growth. However, the usual practice when running a climate change scenario in a CGE is to set the GDP as endogenous and to set partial or total factor productivity exogenously. Climate change therefore impacts GDP and associated income in each region. This can explain in particular outliers observed in Fig. 7 in the NE quadrant for some CGEs, because prices are not the only variable interacting with demand.

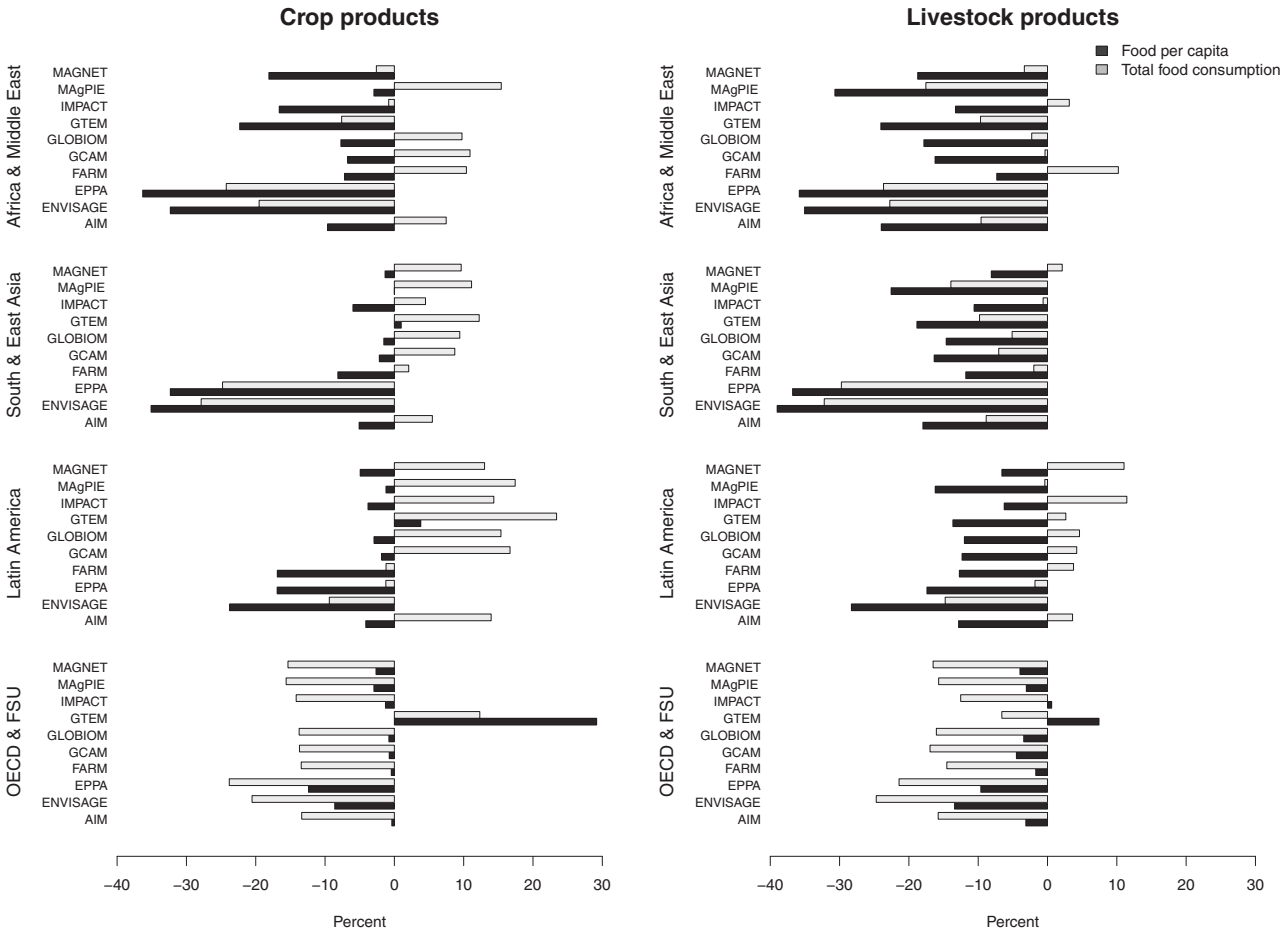

Fig. 5. Change in crop and livestock calorie consumption in scenario SSP3 in 2050 when compared with SSP2, by model and region.

of MAgPIE and GCAM is expected because price is not included in their demand functions. GTEM, MAGNET, and to some extent FARM, appear as relatively price-inelastic models, with a large number of implied elasticities in the −0.1/−0.2 range. This is consistent with their reported elasticities (Fig. 1), except for MAGNET that appears lower than initially expected. ENVISAGE, IMPACT, AIM, and GLOBIOM are much more price responsive models, with most elasticities in the −0.5/−0.1 range.

The implications of model sensitivity to prices appear when looking at the impact on food consumption (see Fig. 8; GCAM and MAgPIE, not price responsive for crops, are not considered here). The range of magnitude of climate impact varies significantly depending on input data, in particular on the crop model, but for a given input, economic model results also differ widely. On average, we observe a range of impact from −50 kcal to −88 kcal, that is, −1.6% to −2.9% of the average food consumption, if we compare with FAO trends to 2050 that do not consider climate change. However, if we consider the results from the most responsive models (AIM, GLOBIOM, and IMPACT), we obtain a notably larger impact at −6%. When we compare with the magnitude of future food demand projections, this would for instance be equivalent to a shift downward in the FAO projection from +54% to +45%. This result therefore confirms that integrating climate change in future projections affects the magnitude of future demand. But it also highlights that climate change-related uncertainty is much lower than uncertainty from macroeconomic scenarios and from model responses. This observation is in line with other studies looking at this question (Easterling et al., 2007; Nelson et al., 2013).

4.5. Sensitivity of model projections to bioenergy policy

The response of agricultural markets to bioenergy policy is the third dimension of AgMIP scenarios (Lotze-Campen et al., 2014). This scenario explores some degree of second generation bioenergy development relying on perennial crops and possibly with limited encroachment from newly cultivated areas on current cropland. Five models out of ten implemented this scenario, but only three are food demand responsive and provide some food consumption results. These three models find the impacts to range from −1.5% to no change of the average consumption in 2050 (maximum value for AIM with −33 kcal/cap/day;

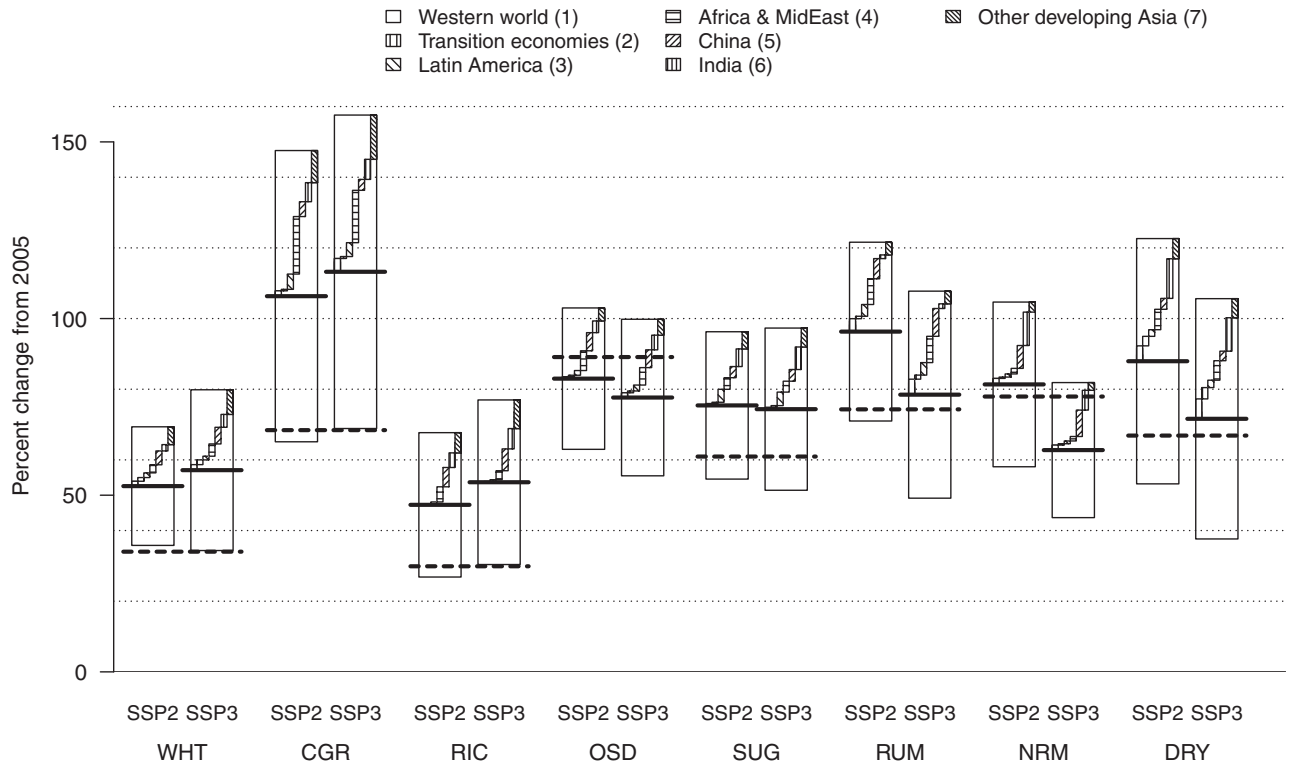

Fig. 6. Standard deviation around mean 2050 value of model projections, decomposed by region. The black plain line represents the mean of model results, the dashed line the FAO projection. White boxes indicate the standard deviation observed across models for variation at the world level. The stacked thin bars represent the decomposition of the world standard deviation. This is performed by splitting the world standard deviation across regions proportionally to standard deviation observed in each region.

see detailed results in the Data Appendix). Unfortunately, no scenario on first generation biofuels was explored in this exercise to compare with the impact of feedstocks competing more directly with food consumption.

5. Conclusion

Many of the underlying drivers of food demand are subject to uncertainty. Demographics are not easily predictable beyond a few decades, and economic growth is even more unpredictable. In this article, we compared 10 global economic models providing projections of future agricultural market conditions under common scenarios. The models used harmonized socioeconomic assumptions, providing a convenient platform to investigate other sources of divergence across model projections.

The range of results obtained in our comparison reflects the diversity of demand system specifications and calibration choices. Regional projections and sectoral evolution vary significantly from one model to another for a similar baseline. Overall for the middle of the road scenario (SSP2), the average increase in calories in our sample of models is 74% (st. dev. 14%) and all models are higher than the value of +54% projected by FAO. This finding would probably call into question the usual expectations of a 60% increase in world agriculture production by 2050. Even without the leverage effect on feed crops needed for more livestock products, an increase by 20–30% higher magnitude seems plausible following our analysis, which is in line with some other works (Tilman et al., 2011).

This projection is however dependent on macroeconomic assumptions and was obtained under a middle range assumption scenario (SSP2), which supposes higher economic growth than FAO does for its most recent projections. When considering a variant with lower economic growth but higher population increase (SSP3), we find decreased per-capita consumption relative to SSP2 (−12%). However, overall food demand remains similar because population is higher, but it is more sourced from crop than from animal products. Demand for livestock products indeed appears very sensitive to change in projected GDP per capita, and large uncertainty is associated with the development of these markets, in particular in developing regions.

We also tested the sensitivity of these projections to future climate change and to a second generation bioenergy scenario. We find that magnitude of change in per-capita demand for the most dramatic climate change scenario would be in the range of −1.5% to −3% using model averages, with −6% for the most reactive model, whereas the bioenergy option would have at most a 1.5% effect on world demand. The related uncertainties are therefore lower than those associated with model

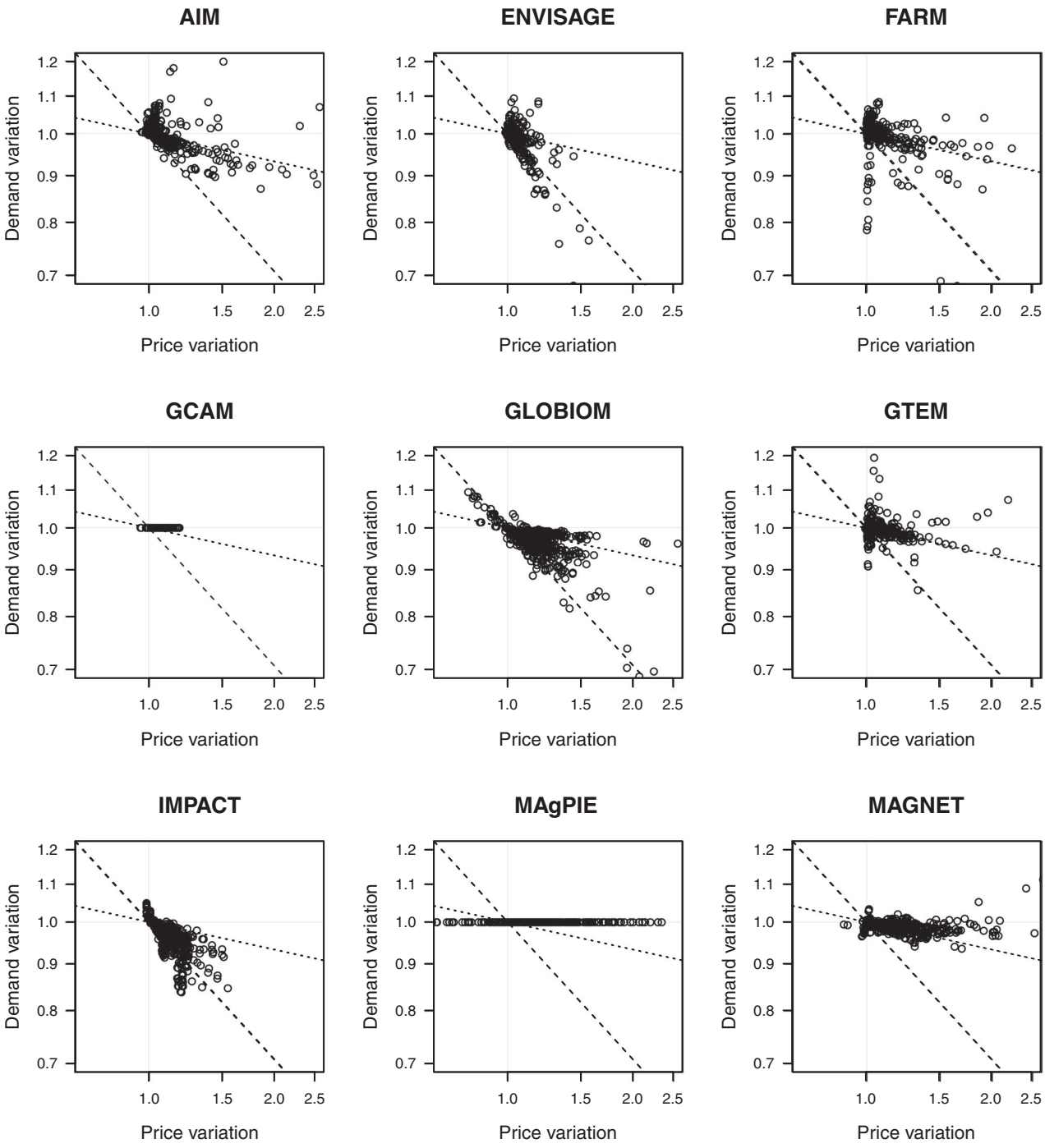

Fig. 7. Food demand response of nine of the models to a change in producer prices across the AgMIP climate scenarios for single commodity products. Graphs are plotted in log scale. Each dot corresponds to a product × region × scenario result; dot superposition areas appear in black. The dashed line indicates the response of a price elasticity of −0.5; the dotted line locates is the response of an elasticity of −0.1. Both responses for the year 2030 and the year 2050 are displayed, as variation from the baseline, for the four AgMIP climate scenarios. Results on climate change scenarios were not available for the EPPA model.

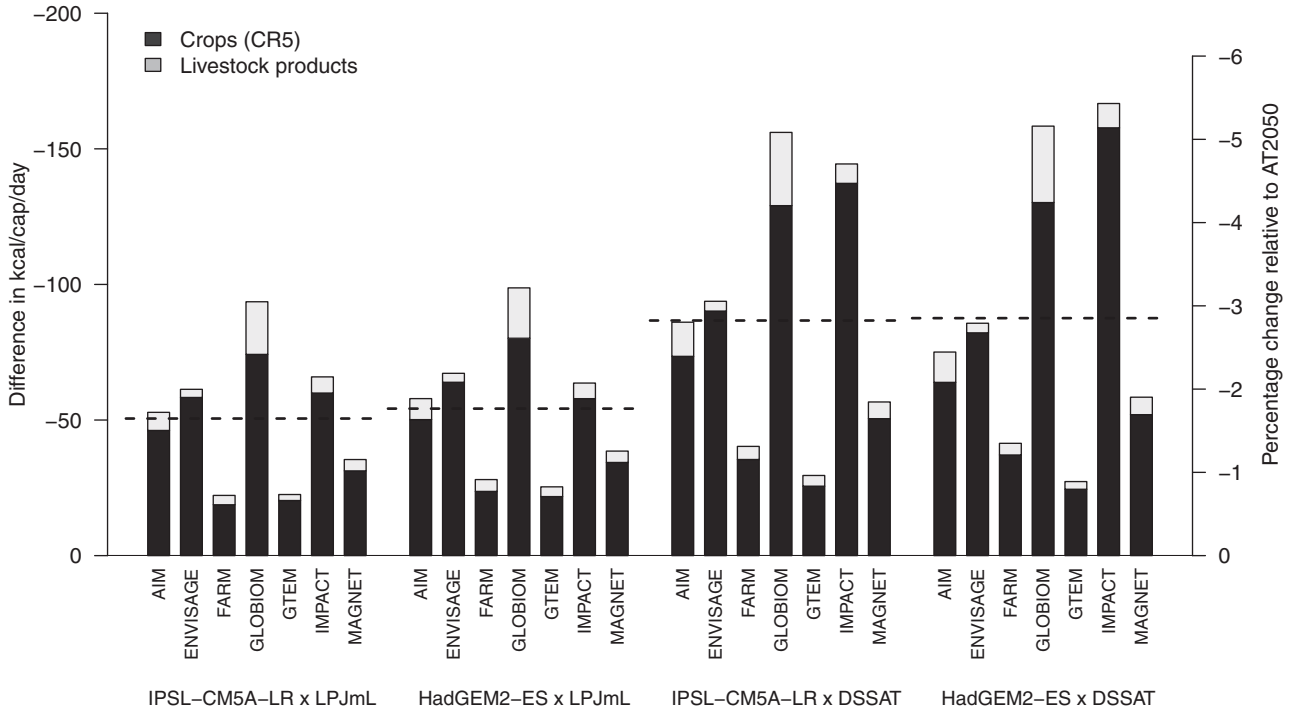

Fig. 8. Food consumption change for the seven models showing demand response to climate change. Left axis indicates the loss in kcal per capita per day, whereas right axis indicates the results expressed as a percentage of total calorie intake according to 2050 FAO estimate in AT2050 (Alexandratos and Bruinsma, 2012). Results are calculated for two general circulation models (IPSL-CM5A-LR and HadGEM2-ES) using the RCP 8.5 greenhouse gas emissions pathway and two crop and vegetation models (LPJmL and DSSAT) with no CO_2 fertilization (see Nelson et al., 2014 for more details). Dashed lines indicate the mean of the scenario considered across economic models. Models with no price responses on crops (MAgPIE, GCAM) are not displayed in this figure. EPPA results on climate change scenarios were not available.

specifications and much lower than the usual uncertainty on population and GDP projection drivers.

From a technical point of view, this comparison exercise also provides a first attempt to open the model "black boxes" in order to analyze the origin of their differences. A first important source of divergence comes from the choice of behavioral parameters. Implicit elasticities measured in this exercise often show wide and sometimes fuzzy distributions. The process allowed for significant improvement of assumptions and specifications, and exchange of good practices across modelers. In particular, before reaching the final results presented earlier, modelers sometimes had to revise some of their specifications (for instance when no dynamic recalibration was present), when not switching to a different demand system more suitable for long-term projections. Discussions on most recent estimates of elasticities also allowed some teams to update their parameter choices, in particular with respect to negative income elasticities, often absent in a standard calibration.

However, the limitations from some demand systems and the lack of empirical estimates at a global level pose problems. More serious, the risk of endogeneity is also present as five of ten models calibrated their income elasticities using FAO projections. Only two models used other data sources than FAO: IMPACT relying on expert information for specific countries, and MAgPIE using their own panel estimation. This being said, the analysis of their projected patterns also raises questions, and the need for more precise parameter estimates is clearly demonstrated by this analysis. Also usually lacking in the technical debate is the influence of price trends for estimation of projections. FAO reports on the future of agricultural demand remain silent on this point, and models relying on econometric estimations should also acknowledge the role of this parameter for more transparent projections. In an era of economic uncertainty and highly turbulent prices, both drivers could considerably influence the future conditions for food access and food demand.

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Pathways (http://www.iiasa.ac.at/web-apps/tnt/RcpDb), the Coupled Model Intercomparison Project Phase 5 (http://cmip-pcmdi.llnl.gov/cmip5), the Shared Socioeconomic Pathways (https://secure.iiasa.ac.at/web-apps/ene/SspDb), and the climate impacts on agricultural crop yields from the Inter-Sectoral Impact Model Intercomparison Project (http://www.isi-mip.org).

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Chapter IV

N_2O emissions from the global agricultural nitrogen cycle - current state and future scenarios

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N_2O emissions from the global agricultural nitrogen cycle – current state and future scenarios

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Abstract. Reactive nitrogen (N_r) is not only an important nutrient for plant growth, thereby safeguarding human alimentation, but it also heavily disturbs natural systems. To mitigate air, land, aquatic, and atmospheric pollution caused by the excessive availability of N_r, it is crucial to understand the long-term development of the global agricultural N_r cycle.

For our analysis, we combine a material flow model with a land-use optimization model. In a first step we estimate the state of the N_r cycle in 1995. In a second step we create four scenarios for the 21st century in line with the SRES storylines.

Our results indicate that in 1995 only half of the N_r applied to croplands was incorporated into plant biomass. Moreover, less than 10 per cent of all N_r in cropland plant biomass and grazed pasture was consumed by humans. In our scenarios a strong surge of the N_r cycle occurs in the first half of the 21st century, even in the environmentally oriented scenarios. Nitrous oxide (N_2O) emissions rise from 3 Tg N_2O-N in 1995 to 7–9 in 2045 and 5–12 Tg in 2095. Reinforced N_r pollution mitigation efforts are therefore required.

1 Introduction

More than half of the reactive nitrogen (N_r) fixed every year is driven by human activity (Boyer et al., 2004). The main driver of the nitrogen cycle remains agricultural production, whose ongoing growth will require ever larger amounts of N_r to provide sufficient nutrients for plant and livestock production in the future.

The industrial fixation of the once scarce nutrient contributed to an unrivaled green revolution of production in the second half of the 20th century. Yet, only 35 to 65 % of the N_r applied to global croplands is taken up by plants (Smil, 1999). The remaining share may interfere with natural systems: The affluent availability of N_r leads to biodiversity losses and to the destruction of balanced ecosystems (Vitousek et al., 1997). In the form of nitrous oxide (N_2O), N_r contributes to global warming (Forster et al., 2007) and is the single most important ozone depleting substance (Ravishankara et al., 2009). Finally, it contributes to soil (Velthof et al., 2011), water (Grizzetti et al., 2011), and air pollution (Moldanova et al., 2011). Brink et al. (2011) estimate that the damage caused by nitrogen pollution adds up to 70–320 billion Euro in Europe alone, equivalent to 1–4 % of total income.

Therefore, much effort has been dedicated to improving our knowledge about the global agricultural N_r cycle. Smil (1999) pioneered the creation of the first comprehensive global N_r budget, and determined the key N_r flows in agriculture, most importantly fertilizer application, biological nitrogen fixation, manure application, crop residue management, leaching, and volatilisation. Sheldrick et al. (2002) extended the nutrient budgets to phosphorus and potash. Galloway et al. (2004) included natural terrestrial and aquatic systems in the N_r cycle. Liu et al. (2010a) broke up the global agricultural nutrient flows to a spatially explicit level. Bouwman et al. (2005, 2009, 2011) were the first, and so far the only, to have simulated the future development of the N_r cycle with detailed regional N_r flows.

However, the description of the current state of the N_r cycle was often incomprehensive. Belowground residues were so far not considered explicitly by other global studies, even though they withdraw large amounts of N_r from soils, and their decay on fields contributes to N_r losses and emissions. Similarly, not all past studies included fodder crops in their

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budgets, although they make up a considerable share of total cropland production. Furthermore, no bottom-up estimate for N_r release by the loss of soil organic matter exists so far. Regarding future projections, substitution effects between different N_r inputs are usually not considered.

In this paper, we create new estimates for the state of the agricultural N_r cycle in 1995 and four future scenarios until 2095 based on the SRES storylines. Our study presents a comprehensive description of the N_r cycle and covers N_r flows that have not been regarded by other studies so far. We create detailed cropland N_r budgets, but also track N_r flows upstream towards the processing sector, the livestock system and final consumption. This unmasks the low N_r efficiency in agricultural production. We use an independent parametrisation of the relevant N_r flows, concerning for example N_r in crop residues or biological N_r fixation. This allows for the identification of uncertainties in current estimates. For future projections we use a closed budget approach that allows for substitution between cropland N_r inputs (like fertilizer, manure or crop residues) and for an endogenous calculation of livestock N_r excretion. The budget approach is also used to estimate total nitrogen losses from fertilization and manure management (the sum of N_2, NO_x, NH_y and N_2O volatilisation as well as N_r leaching). As N_2O emissions play a crucial role in a global context, our model estimates them explicitly. For this purpose, our study uses the emission parameters of the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (Eggleston et al., 2006).

The paper is set up as follows: In the methods section, we first describe the Model of Agricultural Production and its Impact on the Environment (MAgPIE) that delivers the framework for our analysis. Then we give an overview on the implementation of crop residues, conversion byproducts and manure in the model. The description of all major N_r flows is followed by a summary of the scenario designs. In the results section, we present our simulation outputs for the state of the N_r cycle in 1995 and our projections for inorganic fertilizer consumption, N_2O emissions and other important N_r flows. In the discussion section, we compare our estimates to other studies and integrate the findings to a comprehensive cropland N_r budget for 1995, highlighting the largest uncertainties. We also compare our scenarios for the rise of the N_r cycle in the 21st century to estimates of other studies. As it is a key driver of the N_r cycle, we examine the livestock sector in more detail. Finally, the implications of our findings on the threat of N_r pollution are followed by our conclusions and an outlook on the opportunities for mitigation.

2 Materials and methods

2.1 General model description

MAgPIE (Lotze-Campen et al., 2008; Popp et al., 2010, 2012; Schmitz et al., 2012) is a model well suited to performing assessments of agriculture on a global scale and to simulating long-term scenarios. It is comprehensive concerning the spatial dimension and covers all major crop and livestock sectors. Moreover, it features the major dynamics of the agricultural sector, like trade, technological progress or land allocation according to the scarcity of suitable soil, water and financial resources. As it treats agricultural production not only as economic value but also as physical good, it can easily perform analysis of material flows.

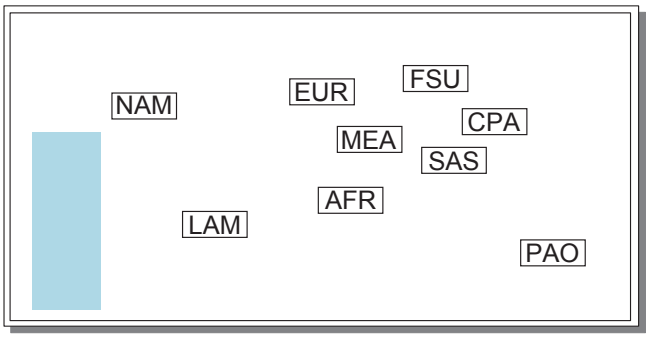

Fig. 1. The ten MAgPIE world regions. Sub-Sahara Africa (AFR), Centrally Planned Asia (CPA), Europe (including Turkey) (EUR), Former Soviet Union (FSU), Latin America (LAM), Middle East and North Africa (MEA), North America (NAM), Pacific OECD (Australia, Japan and New Zealand) (PAO), Pacific Asia (PAS), and South Asia (SAS).

MAgPIE optimizes global land-use patterns to settle a global food demand at minimal production costs. Food demand is exogenous to the model and differentiated into 18 crop groups and 5 livestock production types. The demand for feed depends on the livestock production quantity with individual feed baskets for each livestock category (Weindl et al., 2010). The demand for material consumption and the production waste are assumed to grow in proportion to food demand, while the production for seed is a fixed share of crop production. All demand categories are estimated separately for 10 world regions (Fig. 1) 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). The production of crops requires financial resources as well as land and irrigation water. Production costs per area are derived from GTAP cost-of-firm data (Schmitz et al., 2010). Land requirements depend on the yield-level of the region, which are calibrated to meet 1995 FAO data. Higher production can either be reached by land expansion or by the purchase of yield-increasing technological change (Dietrich, 2011; Popp et al., 2011). Water availability and water requirements per crop are derived from the LPJmL model (Bondeau et al., 2007; Gerten et al., 2004). MAgPIE is solved for each 10-yr timestep between 1995 and 2095, whereby the cropland area and the level of technology are passed on from one timestep as input data to the consecutive timestep.

The existing model (as described in the Supplement) has been extended by a number of features in order to describe the dynamics of the N_r cycle. Crop residues and conversion byproducts from crop processing make up a major share of total biomass and were therefore integrated into the model (Sect. 2.2). Moreover, all dry matter flows were transformed into N_r flows. N_r flows in manure management, cropland fertilization and the transformation of N_r losses into emissions were included (Sect. 2.3). Finally, the scenario setup is described in Sect. 2.4. Detailed documentation as well as a mathematical description of all model-extensions can be found in Appendix A.

2.2 Crop residues and conversion byproducts

As official global statistics exist only for crop production and not for crop residue production, we obtain the biomass of residues by using crop-type specific plant growth functions based on crop production and area harvested. Plant biomass is divided into three components: the harvested organ as listed in FAO, the aboveground (AG) and the belowground (BG) residues. For AG residues of cereals, leguminous crops, potatoes and grasses, we use linear growth functions (Eggleston et al., 2006) with a positive intercept which accounts for the decreasing harvest index with increasing yield. For crops without a good matching to the categories of Eggleston et al. (2006), we use constant harvest indices (Wirsenius, 2000; Lal, 2005; Feller et al., 2007).

Based on Smil (1999), we assume that 15 % of AG crop residues in developed and 25 % in developing regions are burned in the field. Furthermore, developing regions use 10 % of the residues to settle their demand for building materials and household fuel. The demand for crop residues for feed is calculated based on crop residues in regional livestock specific feed baskets from Weindl et al. (2010). The remaining residues are assumed to be left on the field. We estimate BG residue production by multiplying total AG biomass (harvest + residue) with a crop-specific AG to BG ratio (Eggleston et al., 2006; Khalid et al., 2000; Mauney et al., 1994). All BG crop residues are assumed to be left on the field.

Conversion byproducts like brans, molasses or oil cakes occur during the processing of crops into refined food. We link the production of conversion byproducts to the domestic supply of the associated crops using a fixed regional conversion ratio. Feed demand for conversion byproducts is based on feed baskets from Weindl et al. (2010) and rises with livestock production in the region. All values are calibrated to meet the production and demand for conversion byproducts of FAO in 1995 (FAOSTAT, 2011). In case the future demand for feed residues or crop byproducts exceeds the production, they can be replaced by feedstock crops of the same nutritional value.

2.3 N_r flows

2.3.1 N_r content of plant biomass, conversion byproducts and food

The biomass flows of the MAgPIE model are transformed into N_r flows, using product-specific N_r contents. We compile the values for harvested crops, conversion byproducts, AG and BG residues from Wirsenius (2000); Fritsch (2007); FAO (2004); Roy et al. (2006); Eggleston et al. (2006) and Khalid et al. (2000). The N_r in vegetal food supply is estimated by subtracting the N_r in conversion byproducts from N_r in harvest dedicated for food. N_r in livestock food supply is calculated by multiplying the regional protein supply from each commodity group of FAOSTAT (2011) with protein to N_r ratios of Sosulski and Imafidon (1990) and Heidelbaugh et al. (1975). As food supply does not account for waste on the household-level, we use regional intake to supply shares from Wirsenius (2000).

2.3.2 Manure management

The quantity of N_r in livestock excreta is calculated endogenously from N_r in feed intake (consisting of feedstock crops, conversion byproducts, crop residues and pasture) and livestock productivity. The N_r in feed minus the amount of N_r in the slaughtered animals, milk and eggs equals the amount of N_r in manure. To estimate the mass of slaughtered animals, we multiply the FAO meat production with livestock-specific carcass to whole body weight ratios from Wirsenius (2000). N_r contents of slaughtered animals, milk and eggs are obtained from Poulsen and Kristensen (1998).

Manure from grazing animals on pasture is assumed to be returned to pasture soils except a fraction of manure being collected for household fuel in some developing regions (Eggleston et al., 2006). Manure from feedstock crops and conversion byproducts are assumed to be excreted in animal houses. We estimate that one quarter of the N_r in crop residues used as feed in developing regions stems from stubble grazing on croplands, while the rest is assigned to animal houses. Finally, we distribute all manure in animal houses between 9 different animal waste management systems according to regional and livestock-type specific shares in Eggleston et al. (2006).

2.3.3 Cropland N_r inputs

In our model, cropland N_r inputs include manure, crop residues left in the field, biological N_r fixation, soil organic matter loss, atmospheric deposition, seed and inorganic fertilizer.

For the manure managed in animal houses, recycling shares for each animal waste management system are adopted from Eggleston et al. (2006). The manure collected for recycling in developing regions is assigned fully to cropland soils, while it is split between cropland and pasture soils

in developed regions. Additionally, all N_r excreted during stubble grazing is returned to cropland soils.

For crop residues left in the field, we assume that all N_r is recycled to the soils, while 80–90 % of the residues burned in the field are lost in combustion (Eggleston et al., 2006).

N_r fixation by free living bacteria in cropland soils and rice paddies is taken into account by assuming fixation rates of 5 kg per ha for non-legumes and 33 kg per ha for rice (Smil, 1999). The N_r fixed by leguminous crops and sugar cane is estimated by multiplying N_r in plant biomass (harvested organ, AG and BG residue) with regional plant-specific percentages of plant N_r derived from N_2 fixation (Herridge et al., 2008).

N_r release by the loss of soil organic matter after the conversion of pasture land or natural vegetation to cropland is estimated based on the methodology of Eggleston et al. (2006). Our estimates for 1995 use a dataset of soil carbon under natural vegetation from the LPJmL model (Sitch et al., 2003; Gerten et al., 2004; Bondeau et al., 2007). For 1995, we use historical land expansion from the HYDE-database (Klein Goldewijk et al., 2011a), while the land expansion in the future is estimated endogenously by MAgPIE.

The regional amount of atmospheric deposition on croplands for 1995 is taken from Dentener (2006). For future scenarios, we assume that the atmospheric deposition per cropland area grows with the same growth rate as the average regional agricultural NO_x and NH_y emissions.

The amount of harvest used for seed is obtained from FAOSTAT (2011). We multiply the seed with the N_r share of the harvested organ to estimate N_r in seed returned to the field.

Regional inorganic fertilizer consumption in 1995 is obtained from IFADATA (2011). For the scenarios, we use a closed budget approach. For this purpose, we define cropland soil N_r uptake efficiency (SNUpE) as the share of N_r inputs to soils (fertilizer, manure, residues, atmospheric deposition, soil organic matter loss and free-living N_r fixers) that is withdrawn from the soil by the plant. These withdrawals from the soil are calculated by subtracting N_r derived not from the soil (seed and internal biological fixation by legumes and sugarcane) from N_r in plant biomass. SNUpE is calculated on a regional level for the year 1995 and becomes an exogenous scenario parameter for future estimates. Its future development is determined by the scenario storyline (see Sect. 2.4).

In future scenarios, the soil withdrawals and the exogenous SNUpE determine the requirements for soil N_r inputs. If the amount of organic fertilizers is not sufficient, the model has to apply as much nitrogen fertilizer as it requires to balance out the budget. In our model, the N_r inputs to crops have no influence on the yield. We assume in reverse that a given crop yield can only be reached with sufficient N_r inputs. An eventual N_r limitation is already reflected in the height of the crop yield.

2.3.4 Emissions

Emission calculations are in line with the 2006 IPCC Guidelines of National Greenhouse Gas Emissions (Eggleston et al., 2006), accounting for NO_x, NH_y as well as direct and indirect N_2O emissions from managed soils, grazed soils and animal waste. Our estimates neither cover agricultural N_2O emissions from savannah fires, agricultural waste burning or cultivation of histosols, nor emissions from waste disposal, forestry or fertilizer production. Emission factors are connected directly to the corresponding N_r flows of inorganic fertilizer application, as well as residue burning and decay on field, manure management, manure application, direct excretion during grazing, and soil organic matter loss. We use a Monte Carlo analysis to estimate the effect of the uncertainty of the IPCC emission parameters on global N_2O emissions.

2.4 Future scenarios

For future projections, we analyse four scenarios based on the SRES storylines (Nakicenovic et al., 2000), varying in two dimensions: economy versus ecology and globalisation versus heterogeneous development of the world regions. The parametrisation of these scenarios differs in several aspects, which try to cover the largest uncertainties for the future development of the N_r cycle (Table 1). In the following, the scenario settings are shortly described, while a detailed description and an explanation of the model implementation is provided in Appendix A4.

Food demand projections and the share of calories from livestock products are calculated based on regressions between income and per-capita calorie demand (intake and household waste), as well as regressions between income and the share of livestock calories in total demand. The regressions are based on a panel dataset (5889 data points) from FAOSTAT (2011) and WORLDBANK (2011) for 162 countries from 1961 to 2007. In the environmentally oriented scenarios, we used different functional forms for the regressions that result in lower values for plant and livestock demand. The future projections are driven by population and GDP scenarios from the SRES marker scenarios (CIESIN, 2002a,b).

Trade in MAgPIE is oriented along historical trade patterns, fixing the share of products a region has imported or exported in the year 1995. To account for trade liberalisation, an increasing share of products can be traded according to comparative advantages in production costs instead of historical patterns. We use two different trade scenarios based on Schmitz et al. (2012), assuming faster trade liberalisation in the globalised scenarios.

The livestock production systems in the 10 MAgPIE regions differ in 1995 both regarding their productivity and the animal feed baskets. To account for the increasing industrialisation of livestock production, we assume an increasing convergence of the livestock systems from the current mix towards the industrialised European system. This highly

Table 1. Scenario definitions, based on the IPCC SRES scenarios.

	1995	2045				2095			
		A1	A2	B1	B2	A1	A2	B1	B2
GDP (10^{12} US$)	34	222	106	170	138	674	314	453	319
Population (10^9 heads)	5.7	8.6	10.8	8.6	9.2	7.4	14.8	7.4	10.4
Food demand (10^{18} J)	23	46	50	42	43	47	81	41	53
– Thereof livestock products	16 %	24 %	17 %	22 %	22 %	22 %	17 %	16 %	18 %
Trade patterns									
– Historical	100 %	60 %	88 %	60 %	88 %	37 %	78 %	37 %	78 %
– Comparative advantage	0 %	40 %	12 %	40 %	12 %	65 %	22 %	65 %	22 %
Livestock systems									
– Current mix	100 %	20 %	50 %	20 %	50 %	0 %	20 %	0 %	20 %
– Industrialised	0 %	80 %	50 %	80 %	50 %	100 %	80 %	100 %	80 %
Animal waste[1]									
– Current mix	100 %	30 %	80 %	40 %	80 %	0 %	50 %	20 %	50 %
– Daily spread	0 %	0 %	0 %	30 %	20 %	0 %	0 %	40 %	50 %
– Anaerobic digester	0 %	70 %	20 %	30 %	0 %	100 %	50 %	40 %	0 %
Soil N_r uptake efficiency (SNUpE)	51 %[2]	60 %	55 %	65 %	65 %	60 %	60 %	70 %	70 %
Intact and frontier forest protection		no	no	yes	yes	no	no	yes	yes

[1]Only for waste in animal houses.
[2]Global average.

productive system has a large proportion of feedstock crops and conversion byproducts in the feed baskets. In the globalised scenarios, convergence is assumed to be faster than in the regionalised scenarios.

Currently, regional animal waste management systems are diverse and their future development is highly uncertain. We assume two major future trends. Firstly, due to the scarcity of fossil fuels and the transformation of the energy system towards renewables, the use of animal manure as fuel for bioenergy will become increasingly important. Secondly, in the environmental scenarios, we also assume that an increasing share of manure is spread to soils in a timely manner. We therefore shift the current mix of animal waste management systems gradually towards anaerobic digesters and daily spread.

Improvements in the cropland soil N_r uptake efficiency may occur in the future due to increasing environmental awareness or to save input costs. The regional efficiencies have been calculated for 1995, and we assume that they gradually increase in all scenarios, with the environmental scenarios reaching the highest efficiencies.

Finally, the expansion of agricultural area into unprotected intact and frontier forests is restricted gradually until 2045 in the environmental oriented scenarios, as described in Schmitz (2012).

The scenarios start in the calibration year 1995 and continue until 2095. The base year 1995 facilitates the comparison with other studies (Smil, 1999; Sheldrick et al., 2002; Liu et al., 2010a) and allows for a consistency check and benchmarking between the scenarios and the real development since 1995.

3 Results

Detailed global and regional results of the current state of the agricultural N_r cycle and the four scenarios can be found in the Supplement. In the following, the most important results are summarised.

3.1 Global nitrogen cycle

3.1.1 State in 1995

According to our calculations for the year 1995, 205 Tg N_r are applied to or fixed on global cropland, of which 115 is taken up by cropland plant biomass. Thereof, 50 Tg are fed to animals in the form of feedstock crops, crop residues, or conversion byproducts, plus an additional 72 Tg from grazed pasture, to produce animal products which contain 8 Tg N_r. In total, plant and animal food at whole market level contains 24 Tg N_r, of which finally only 17 Tg N_r are consumed. Figure 2 shows an in-depth analysis of N_r flows in 1995 on a global level.

3.1.2 Scenarios

In our four scenarios, the throughput of the N_r cycle rises considerably within the 21st century. Total N_r in cropland plant biomass reaches 244 (B2)–323 (A1) Tg N_r in 2045 and 251 (B1)–434 (A2) Tg N_r in 2095. Also, the range of soil inputs increases throughout the century, starting with 185 Tg in 1995 to 286 (B2)–412 (A1) Tg N_r in 2045 and 286 (B1)–553 (A2) Tg N_r in 2095. Inorganic fertilizer consumption in the B scenarios show a modest increase to 121 (B2) and 145

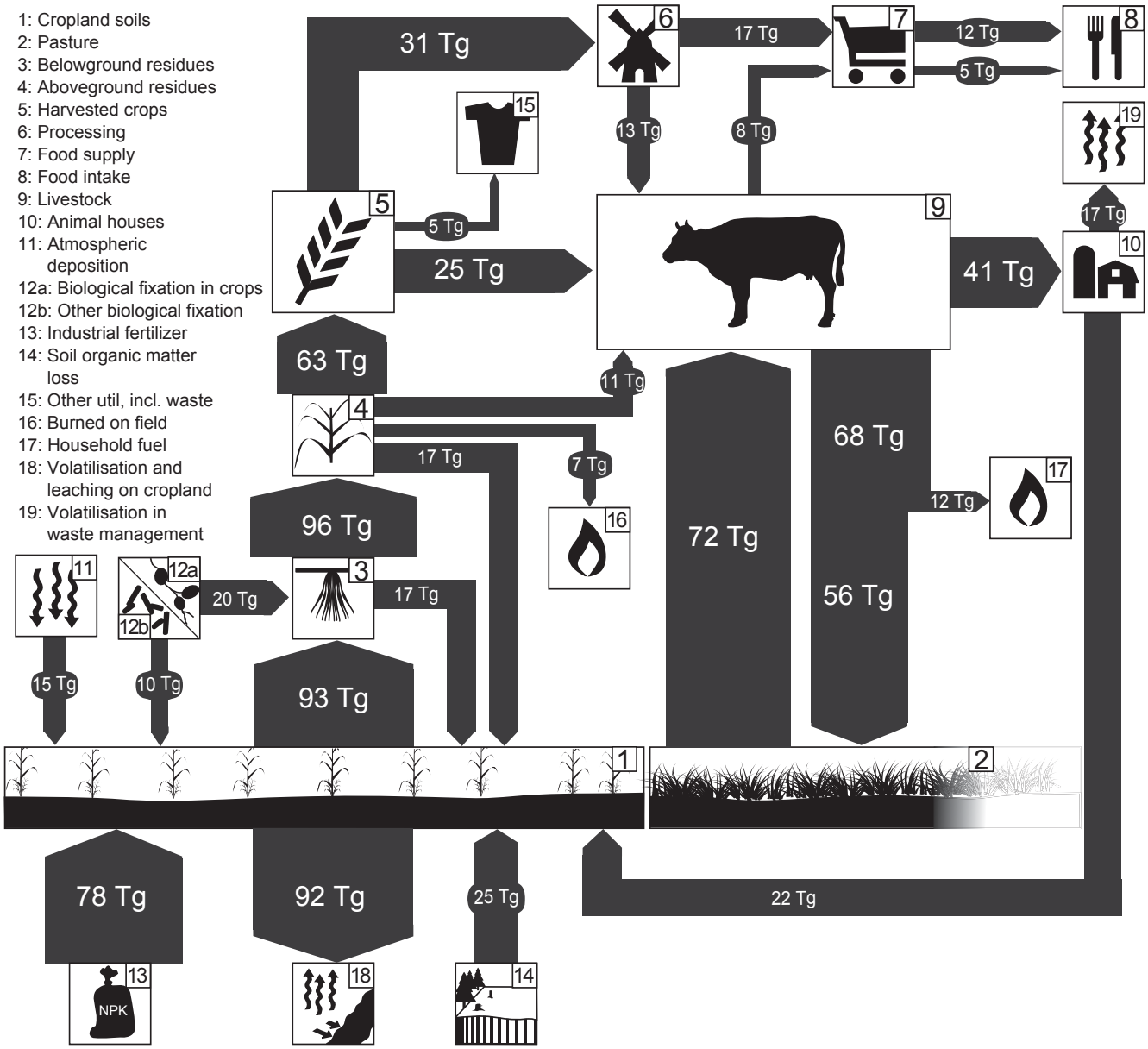

Fig. 2. Agricultural N_r cycle in Tg N_r in the year 1995. Flows below 5 Tg N_r are not depicted. No estimates were made for N_r inputs to pasture soils by atmospheric deposition and biological fixation.

(A1) Tg N_r until 2045 and a stagnating or even declining consumption thereafter, while the A scenarios exhibit a much stronger and continuous increase to 173 (A1) and 177 (A2) Tg N_r in 2045, and 214 (A1) and 260 (A2) Tg N_r in 2095 (Fig. 3). Despite these wide ranges, the differences of N_2O emissions between the scenarios is in the first half of the century rather narrow. They start with 3.9 Tg N_2O-N in 1995, with a range of 3.0 to 4.9 Tg N_2O-N being the 90 % confidence interval for uncertainty of the underlying emission parameters of Eggleston et al. (2006). Up to 2045, they rise to 7.2 (5.4 to 9.0) Tg N_2O-N in the B1 scenario and 8.6 (6.6 to 10.5) Tg N_2O-N in the A2 scenario, and widen towards the end of the century to 4.9 (3.5 to 6.4) Tg N_2O-N in the B1 scenario and 11.6 (8.8 to 14.2) Tg N_2O-N in the A2 scenario (Fig. 4).

3.2 Regional budgets

While the surge of the N_r cycle can be observed in all regions, the speed and characteristics are very different between regions (Table 2). Sub-Saharan Africa (AFR), South Asia (SAS), and Australia and Japan (PAO) show the strongest relative increases in harvested N_r, while in Europe (EUR) and North America (NAM) the increases are more modest. The

Table 2. Regional estimates of N_r flows for the state in 1995 and for the four scenarios $\frac{A1|B1}{A2|B2}$ in Tg N_r per year. Losses consist of losses from cropland soils and animal waste management.

N_r flow	Year	World		Regions																					
				AFR		CPA		EUR		FSU		LAM		MEA		NAM		PAO		PAS		SAS	
Harvest	**1995**	**63**		**3**		**12**		**10**		**5**		**6**		**2**		**13**		**2**		**3**		**7**	
	2045	182	160	15	14	30	28	15	14	10	9	29	21	10	10	20	19	17	11	6	5	30	29
		153	143	12	12	26	28	15	14	9	9	22	19	8	7	23	20	10	7	6	5	21	22
	2095	196	137	20	9	33	27	16	13	11	8	26	13	14	12	21	17	18	7	5	3	33	29
		260	169	24	19	38	30	19	15	13	11	50	22	13	9	32	21	25	9	10	6	35	29
Residues	**1995**	**35**		**3**		**6**		**4**		**3**		**4**		**1**		**6**		**1**		**2**		**5**	
	2045	94	85	10	9	15	15	7	7	7	7	16	13	4	4	10	9	9	6	4	4	12	12
		73	67	8	7	12	13	6	6	4	4	11	9	3	2	10	8	5	3	4	3	11	10
	2095	98	76	11	7	17	15	7	6	8	7	15	9	5	5	11	9	8	3	4	3	13	12
		114	76	12	10	19	14	8	6	5	4	21	9	5	3	13	9	11	4	6	3	15	12
Fertilizer	**1995**	**78**		**1**		**24**		**13**		**2**		**4**		**3**		**13**		**1**		**4**		**13**	
	2045	173	145	9	7	40	36	13	13	11	9	6	7	15	14	23	21	33	19	5	3	20	15
		177	122	14	8	41	30	21	16	8	5	7	10	11	8	30	20	18	9	6	4	22	11
	2095	214	128	0	0	50	39	21	16	12	8	23	0	23	17	19	15	32	12	4	4	24	17
		260	131	19	10	59	35	22	15	10	7	20	5	12	9	37	20	46	12	7	5	27	12
Manure	**1995**	**111**		**15**		**12**		**13**		**7**		**21**		**3**		**10**		**4**		**3**		**22**	
	2045	241	217	65	60	28	22	20	15	8	7	63	55	7	7	9	6	3	2	6	5	32	39
		250	262	51	56	26	37	17	13	10	9	58	52	11	8	14	9	5	3	9	9	49	65
	2095	205	131	105	44	16	12	6	2	7	5	23	36	5	3	17	8	2	1	4	2	19	18
		332	240	69	69	34	26	21	10	11	5	92	51	20	11	17	5	5	1	12	7	50	55
Biol. N_r	**1995**	**27**		**2**		**4**		**2**		**2**		**4**		**0**		**5**		**1**		**2**		**4**	
	2045	72	61	8	7	8	7	5	4	4	4	17	11	1	1	8	7	2	2	4	2	17	16
		57	56	6	6	6	8	4	3	4	4	13	11	1	1	8	8	2	2	3	2	10	11
	2095	75	46	11	4	9	5	4	3	5	3	15	6	1	1	7	6	3	0	1	1	20	17
		95	64	12	8	7	7	4	4	5	6	30	12	3	2	11	8	3	2	4	2	17	14
Trade	**1995**	**0**		**0**		**-1**		**-2**		**-1**		**2**		**-2**		**4**		**0**		**-1**		**0**	
	2045	0	0	-8	-8	-1	3	-6	-3	1	1	-11	-14	-2	-1	10	11	14	8	-3	-2	9	6
		0	0	-3	-6	-4	-7	-1	1	1	2	1	3	-7	-4	10	11	7	4	-4	-4	1	0
	2095	0	0	-51	-21	16	14	6	7	1	0	4	-21	0	1	0	6	14	5	-3	-3	14	11
		0	0	-5	-15	-6	1	-2	4	1	6	-3	-8	-19	-6	15	14	20	8	-6	-3	4	-2
Losses	**1995**	**109**		**5**		**27**		**15**		**9**		**8**		**3**		**18**		**3**		**7**		**15**	
	2045	180	146	17	16	32	27	15	13	11	10	28	23	10	9	18	14	19	10	7	6	21	19
		201	137	18	14	37	31	21	14	11	8	27	16	10	7	27	16	13	6	10	7	25	18
	2095	197	103	39	11	31	20	14	8	12	7	23	14	14	8	19	11	18	5	6	3	21	15
		257	131	25	19	45	25	21	11	12	6	43	19	14	8	30	12	26	6	12	5	29	19
N_2O	**1995**	**3.9**		**0.4**		**0.7**		**0.5**		**0.3**		**0.7**		**0.1**		**0.6**		**0.1**		**0.2**		**0.4**	
	2045	8.1	7.2	1.4	1.3	1.1	1	0.6	0.5	0.4	0.3	1.8	1.6	0.4	0.4	0.6	0.5	0.6	0.4	0.3	0.2	0.9	0.9
		8.6	7.5	1.3	1.3	1.2	1.3	0.7	0.5	0.4	0.3	2	1.6	0.4	0.3	0.9	0.6	0.4	0.2	0.3	0.3	1	1
	2095	7.2	4.9	1.8	0.8	1	0.8	0.5	0.3	0.4	0.3	0.8	0.8	0.5	0.4	0.7	0.5	0.6	0.2	0.2	0.1	0.7	0.6
		11.6	7.2	1.7	1.5	1.5	1	0.8	0.5	0.4	0.3	2.9	1.5	0.6	0.4	1.1	0.5	0.9	0.3	0.4	0.2	1.2	1.1

increase in production in AFR is not sufficient to settle domestic demand, such that large amounts of N_r have to be imported from other regions. Also the Middle East and Northern Africa (MEA) have to import large amounts of N_r due to the unsuitable production conditions and high population growth. At the same time, AFR requires only low amounts of inorganic fertilizer, as the domestic livestock production fed with imported N_r provides sufficient nutrients for production. In the globalised scenarios A1 and B1, the overspill of manure even reduces the actual soil nutrient uptake efficiency (SNUpE) in 2095 with 0.41 (A1) and 0.67 (B1), below the potential scenario value of 0.6 or 0.7.

Despite its large increase in consumption, SAS does not require large imports, as it can also settle its N_r requirements with a balanced mix of biological fixation, manure, crop residues and inorganic fertilizer. Similarly, Latin America can cover large parts of its N_r demand with biological fixation and manure. In comparison with this, the large exporters North America (NAM) and Pacific OECD (PAO) have a much stronger focus on fertilization with inorganic fertilizers.

In the globalised scenarios, these characteristics tend to be more pronounced than in the regionalised scenarios, as each region specialises in its relative advantages. The structural differences between the economical and ecological oriented scenarios are less distinct, yet it can be observed that the reduced livestock consumption in developed regions leads to a lower importance of manure and a generally lower harvest of N_r in these regions.

4 Discussion

This study aims to create new estimates for the current state and the future development of the agricultural N_r cycle. For this purpose, we adapted the land-use model MAgPIE to calculate major agricultural N_r flows. As will be discussed in the following, the current size of the N_r cycle is much higher than previously estimated. The future development of the N_r cycle depends largely on the scenario assumptions, which we based on the SRES storylines (Nakicenovic et al., 2000). We expect the future rise of the N_r cycle to be higher than suggested by most other studies. Thereby, the livestock sector dominates both the current state and future developments. The surge of the N_r cycle will most likely be accompanied by higher N_r pollution.

4.1 The current state of the agricultural N_r cycle

Data availability for N_r flows is poor. Beside the consumption of inorganic fertilizer, no N_r flow occurs in official statistics. Even the underlying material flows, like production and use of crop residues or animal manure are usually not recorded in international statistics. Therefore, independent model assessments are required, using different methodologies and parametrisations to identify major uncertainties. In the following we compare our results mainly with estimates of Smil (1999), Sheldrick et al. (2002) and Liu et al. (2010a), as summarised in Table 3.

The estimates for N_r withdrawals by crops and aboveground residues are relatively certain. They have now been estimated by several studies using different parametrisations. The scope between the studies is still large with 50 to 63 Tg N_r for harvested crops and 25 to 38 Tg N_r for residues, whereby the estimate of Sheldrick et al. (2002) may be too high due to the missing correction for dry matter when estimating nitrogen contents (Liu et al., 2010b).

Large uncertainties can be attributed to the cultivation of fodder and cover crops. They represent a substantial share of total agricultural biomass production, and they are rich in N_r and often N_r fixers. Yet, the production area, the species composition and the production quantity are highly uncertain, and no reliable global statistics exist. The estimate from FAOSTAT (2005) used by our study has been withdrawn without replacement in newer FAOSTAT releases. It counts 2900 Tg fresh matter fodder production on 190 million ha (Mha). Smil (1999) appraises the statistical yearbooks of 20 large countries and provides a lower estimate of only 2500 Tg that are produced on 100–120 Mha.

Estimates for N_r in animal excreta diverge largely in the literature. Using bottom-up approaches based on typical excretion rates and N_r content of manure, Mosier et al. (1998) and Bouwman et al. (2011) calculate total excretion to be above 100 Tg N_r. Smil (1999) assumes total excretion to be significantly lower with only 75 Tg N_r. Our top-down approach, using the fairly reliable feed data of the FAOSTAT database, can support the higher estimates of Mosier et al. (1998) and Bouwman et al. (2011), with an estimate of 111 Tg N_r. The same global total of 111 Tg N_r can be obtained bottom-up if one multiplies typical animal excretion rates taken from Eggleston et al. (2006) with the number of living animals (FAOSTAT, 2011). Yet, regional excretion rates diverge significantly; the top-down approach leads to considerably higher rates in Africa and the Middle East and lower rates in South and Pacific Asia.

Biological N_r fixation is another flow of high uncertainty and most studies still use the per ha fixation rates of Smil (1999) for legumes, sugarcane and free-living bacteria. Currently no better estimate exists for free-living bacteria (Herridge et al., 2008). However, they contribute only a minor input to the overall N_r budget with little impacts on our model results. To estimate the fixation by legumes and sugarcane, we use a new approach based on percentages of plant N_r derived from fixation, similar to Herridge et al. (2008). This, in combination with total above- and belowground N_r content of a plant, can predict N_r fixation more accurately. However, the parametrisation of Herridge et al. (2008) probably overestimates N_r fixation, especially for soybeans. Most importantly, the N_r content of the belowground residues as well as the shoot : root ratio seem too high when comparing them

with Eggleston et al. (2006), Sivakumar et al. (1977) or Dogan et al. (2011). Also the N_r content of the shoot seems too high given that soybean residues have a much lower N_r content than the beans (Fritsch, 2007; Wirsenius, 2000; Eggleston et al., 2006). Correcting the estimates of Herridge et al. (2008) for the water content of the harvested crops further reduces their estimate. If one finally accounts for the difference in base year between the two estimates, with global soybean production increasing by 69 % between 1995 and 2005, we come to a global total fixation from legumes and sugarcane of 9 Tg N_r in 1995 as opposed to 21 Tg N_r in 2005 in the case of Herridge et al. (2008). Our estimate is in between the estimates of Smil (1999) and Sheldrick et al. (2002), even though we used a different approach.

Accumulation or depletion of N_r in soils has so far been neglected in future scenarios (Bouwman et al., 2009, 2011), assuming that soil organic matter is stable and all excessive N_r will volatilise or leach. However, the assumption of a steady state for soil organic matter should not be valid for land conversion or for the cultivation of histosols. Our rough bottom-up calculations estimate that the depletion of soil organic matter after transformation of natural vegetation or pasture to cropland releases 25 Tg N_r per year. With a yearly global average release of 122 kg N_r per ha newly converted cropland, the amount of N_r released may exceed the nutrients actually required by the crops, especially in temperate, carbon rich soils. Vitousek et al. (1997) estimates that the cultivation of histosols and the drainage of wetlands releases another 10 Tg N_r per year, although it is unclear how much thereof enters agricultural systems.

The total size of the cropland N_r budget is larger than estimated by previous studies. This can be attributed less to a correction of previous estimates than to the fact that past studies did not cover all relevant flows. In Table 3 we summarise cropland input and withdrawals mentioned by previous studies. The sum of all withdrawals (Total OUT) ranges between 81 and 115 Tg N_r. However, if the unconsidered flows are filled with estimates from other studies, the corrected withdrawals (Total OUT*) shifts to 105–134 Tg N_r. The same applies to inputs, where the range shifts and narrows down from 137–205 Tg N_r total inputs (Total IN) to 198–232 Tg N_r total inputs when all data gaps are filled (Total IN*). The N_r uptake efficiency (NUpE*), defined as the fraction of IN* which is incorporated into OUT* remains within the plausible global range of 0.35–0.65 defined by Smil (1999) for all studies. In our study, this holds even for every MAgPIE world region. SNUpE and SNUpE* are slightly higher, with 49 % and 51 % of N_r applied to soils being taken up by the roots of crops. The corrected estimates for total losses (Losses*) is, with 84–112 Tg N_r, significantly higher than previously estimated.

Table 3. Comparison of global cropland soil balances.

	This study	Smil (1999b)	Sheldrick (1996)	Liu (2010)
Base year	1995	1995	1996	2000
OUT				
Crops	50	50	63	52
Crop residues	31	25	38	29
Fodder	13	10	–	–
Fodder residues	4	–	–	–
BG residues	17	–	–	–
IN				
Residues	12	14	23	11
Fodder residues	4	–	–	–
BG residues	17	–	–	–
Legume fixation	9	10	8	} 22
Other fixation	10	11	–	
Fixation fodder	11	12	–	–
Atm. deposition	15	20	22	14
Manure on field	24	18	25	17
Seed	2	2	–	–
Irrigation water	–	4	–	3
Sewage	–	–	3	–
Soil organic matter loss	25	–	–	–
Fertilizer	78	78	78	68
Histosols	–	–	–	–
BALANCE				
Total OUT	115	85	101	81
Total OUT*	115	105	134	114
Total IN	205	169	159	137
Total IN*	212	217	232	198
Losses	91	80	75	67
Losses*	98	112	97	84
NUpE	0.56	0.50	0.64	0.59
NUpE*	0.54	0.48	0.58	0.58
SNUpE	0.51	0.42	0.62	0.51
SNUpE*	0.49	0.42	0.54	0.48

*Data gaps are filled with estimates from other studies. We use estimates by this study if available; for irrigation we use Smil (1999), for sewage Sheldrick et al. (2002), and for histosols no estimate exists.

4.2 Scenario assumptions

The simulation of the widely used SRES storylines (Nakicenovic et al., 2000) facilitates the comparison with other studies like Bouwman et al. (2009) or Erisman et al. (2008) and allows for the integration of our results into other assessments. However, the SRES storylines provide only a qualitative description of the future. In the following, the key assumptions underlying our parametrisation and model structure shall be discussed.

All SRES storylines tend to assume a continuation of current trends, without external shocks or abrupt changes of

dynamics. They merely diverge in the interpretation of past dynamics or the magnitude of change assigned to certain trends. Population grows at least until the mid of the 21st century, and declines first in developed regions. Per-capita income grows throughout the century in all scenarios and all world regions, and developing regions tend to have higher growth rates than developed regions. This has strong implications on the food demand, which is driven by both population and income growth. As food demand is a concave function of income, it depends mostly on the income growth in low-income regions. In the first half of the century, the pressure from food demand is therefore highest in the high-income A1 scenario. In the second half, the A2 scenario also reaches a medium income and therefore a relatively high per capita food demand. Additionally, the population growth diverges between the scenarios in the second half of the century, with the A2 scenario reaching the highest world population and as a consequence the highest food demand. As food demand is exogenous to our model, price effects on consumption are not captured by the model. However, even in the A2 scenario the shadow prices (Lagrange multipliers) of our demand constraints increase globally by 0.5 % per year until 2045, with no region showing higher rates than 1.1 %. This indicates only modest price pressure, lagging far behind income growth.

Concerning the productivity of the livestock sector, we assume that the feed required to produce one ton of livestock product is decreasing in all scenarios, even though at different rates. Starting from a global level of 0.62 kg N in feed per ton livestock product dry matter, the ratio decreases to 0.4 (A1) or 0.52 (B2) in 2095 (see Supplement). A critical aspect is that as all regions converge towards the European feed baskets, no productivity improvements beyond the European level take place. Beside the improvement of feed baskets, the amount of feed is also determined by the mix of livestock products, with milk and eggs requiring less N_r in feed than meat. As we could not find a historical trend in the mix of products (FAOSTAT, 2011), we assumed that current shares remain constant in the future. This causes continuing high feeding efficiencies in Europe and North America, where the share of milk and non-ruminant meat is high.

As we calculate our livestock excretion rates based on the feed mix, the increased feeding efficiency also translates into lower manure production per ton livestock product. At the same time, our scenario assumptions of an increasing share of either anaerobic digesters or daily spread in manure management also lead to higher recycling rates of manure excreted in confinement. Even though with increasing development an increasing share of collected manure is applied also to pastureland as opposed to cropland, the amount of applied manure N_r per unit crop biomass remains rather constant. Due to the increasing N_r efficiency, its ratio relative to other N_r inputs like inorganic fertilizers increases.

Our closed budget approach to calculate future inorganic fertilizer consumption is based on the concept of cropland soil N_r uptake efficiency (SNUpE). Other indicators of N_r efficiency relate N_r inputs to crop biomass. They include for example N_r use efficiency (NUE), defined as grain dry matter divided by N_r inputs (Dawson et al., 2008), and agronomic efficiency of applied N_r (AE_N), defined as grain dry matter increase divided by N_r fertilizer (Dobermann, 2005). Compared to these indicators, N_r uptake efficiency (NUpE) indicates the share of all N_r inputs that is incorporated into plant biomass (Dawson et al., 2008). Under the condition that all N_r inputs (including the release of soil N_r) are accounted for, this share has the advantage of an upper physical limit of 1. N_r withdrawals cannot exceed N_r inputs. At the same time, this indicator reveals the fraction of losses connected to the application of N_r inputs. SNUpE is similar to NUpE, but regards only soil inputs and withdrawals and excludes seed N_r as well as internal biological fixation from legumes and sugarcane. Prior to the uptake by the plant, these inputs are not subject to leaching and volatilisation losses (Eggleston et al., 2006), and denitrification losses are also inconsiderable (Rochette and Janzen, 2005). Therefore, one regional value of SNUpE suffices to simulate that NUpE of N_r fixing crops is higher compared to the NUpE of normal crops (Peoples and Herridge, 1990).

The level of SNUpE is in our model an exogenous scenario parameter for future simulations which has a large impact on the estimates of inorganic fertilizer consumption and N_2O emissions. If SNUpE would be 5 percentage points lower, fertilizer consumption would increase by 8 to 10 % in 2045, depending on the scenario. At the same time, total agricultural N_2O emissions would increase by 11 to 15 %. If fertilizer efficiency would increase by 5 percentage points, fertilizer consumption would fall by 7 to 8 % and emissions would decrease by 9 to 13 %. As the magnitude of N_r flows is higher in some scenarios, a ± 5 % variation of SNUpE translates in the A1 scenario into a change of fertilizer consumption of -32 to $+37$ Tg N_r and a change of -1.1 to $+1.3$ Tg N_2O-N of emissions in 2045, while in the B2 scenario fertilizer changes only by -20 to $+24$ Tg N_r and emissions by -0.7 to $+0.8$ Tg N_2O-N.

The future development of SNUpE is highly uncertain. It depends on numerous factors, most importantly on the management practices like timing placing and dosing of fertilizers and the use of nutrient trap crops. Also, a general improvement of agricultural practices like providing adequate moisture and sufficient macro- and micronutrients, pest control and avoiding soil erosion can contribute their parts. Finally, climate, soils, crop varieties and the type of nutrient inputs also influence N_r uptake efficiency. The complexity of these dynamics and the numerous drivers involved still do not allow making long-term model estimates for N_r efficiencies, but this should be a target for future research.

Meanwhile, we use SNUpE as an explicitly defined scenario parameter. As it descriptively indicates the share of losses, and as the theoretical upper limit of 1 is clearly fixed, it makes our model assumptions transparent and

easily communicable. Our assumptions concerning the development of SNUpE are rather optimistic. In 1995, none of the 10 world regions reached a SNUpE of 60 %, and four regions (CPU, FSU, PAS, SAS) were even below 50 %. The current difference between the region with the lowest SNUpE (CPA with 43 %) and the region with the highest SNUpE (EUR with 57 %) is thereby still lower than the difference of EUR and our scenario parameter of 70 % for the environmentally oriented scenarios.

We assumed that trade liberalisation continues in all scenarios, even though at different paces. The trade patterns diverge strongly between the scenarios, even though certain dynamics persist. Sub-Saharan Africa, Europe and Latin America tend to become livestock exporting regions, while South, Central and Southeast Asia as well as the Middle East and Northern Africa become importers of livestock products. On the other hand, sub-Saharan Africa and Pacific Asia become importers of crop products, while the former Soviet Union and Australia become exporters of crops. Trade dynamics in MAgPIE are determined partly on the basis of historical trade patterns, partly by competitiveness. However, certain other dynamics that are of great importance in reality, most importantly political decisions like tariffs or export subsidies, are not represented explicitly in the model. Due to the uncertainty regarding trade patterns, regional production estimates are therefore of higher uncertainty than global estimates. Trade patterns have strong implications on the N_r cycle. As soon as two regions are trading, the fertilizer consumption also shifts from the importing to the exporting region. Even more, sub-Saharan Africa currently imports crops and exports livestock products. Livestock fed with imported crops contributes in the form of manure to the cropland soil budgets and facilitates sub-Saharan Africa to use little inorganic fertilizer. Also in our future scenarios, the African livestock sector is very competitive and the inorganic fertilizer consumption does not increase until the mid of the century. A similar dynamic can be observed in Latin America, where inorganic fertilizer consumption also stays rather low.

In our environmentally oriented scenarios B1 and B2, vulnerable ecosystems are protected from land expansion. However, these protection schemes are assumed to be implemented gradually until 2045 and include only some of the most vulnerable forest areas. Large forest areas are still cleared in the beginning of the century, most importantly in the Congo river basin and the southern part of the Amazonian rainforest. Due to the land restrictions in the B scenarios, crop yields have to increase faster to be able to settle the demand with the available cropland area.

4.3 The future expansion of the N_r cycle

The size of the agricultural N_r cycle has increased tremendously since the industrial revolution. While in 1860 agriculture fixed only 15 Tg N_r (Galloway et al., 2004), in 1995 the Haber–Bosch synthesis, biological fixation and soil organic matter loss injected 133 Tg new N_r into the N_r cycle. Our scenarios suggest that this surge will persist into the future, and will not stop before the middle of this century. The development is driven by a growing population and a rising demand for food with increasing incomes, along with a higher share of livestock products within the diet. The N_r in harvested crops may more than triple. Fixation by inorganic fertilizers and legumes as well as recycling in the form of crop residues and manure may also increase by a factor of 2–3.

Our top-down estimates of future animal excreta are higher than the bottom-up estimates by Bouwman et al. (2011). In our scenarios, N_r excretion rises from 111 Tg N_r in 1995 to 217 Tg N_r (B1)–262 Tg N_r (A1) in 2045. Bouwman et al. (2011) estimate that N_r excretion increases from 102 Tg N_r in 2000 to 154 Tg N_r in 2050. These differences are caused by diverging assumptions. Firstly, while Bouwman et al. (2011) assume an increase of global meat demand by 115 % within 50 yr, our study estimates an increase by 136 % (A2)–200 % (A1). Secondly, Bouwman et al. (2011) assume rising N_r excretion rates per animal for the past, but constant rates for the future, such that weight gains of animals are not connected to higher excretion rates. As the current excretion rates in developing regions are still lower than in developed regions (IPCC, 1996), this assumption will underestimate the growth of excretion rates in developing regions. Our implementation calculates excretion rates based on the feed baskets and the N_r in livestock products. Under the assumption that developing regions increasingly adopt the feeding practices of Europe, this top-down approach results in increasing excretion rates per animal in developing regions. However, as we assume no productivity improvements in developed regions, we tend to overestimate future manure excretion in developed regions.

N_r release from soil organic matter (SOM) loss contributes to the N_r budget also in the future, yet with lower rates. In the environmentally oriented B scenarios, cropland expansion and therefore also SOM loss almost ceases due to forest protection, while in the economically oriented scenarios, the loss of SOM still contributes 10 (A1) and 18 (A2) Tg N_r per year. In the A2 scenario the loss even continues at low rates until the end of the century. The reduced inputs of soil organic matter loss have to be replaced by inorganic fertilizers.

Our estimates of inorganic fertilizer consumption are within the range of previous estimates. Figure 3 compares our results to estimates by Daberkow et al. (2000), Davidson (2012), Erisman et al. (2008), Tilman et al. (2001), Tubiello and Fischer (2007) and Bouwman et al. (2009). The differences in estimates is enormous, ranging in 2050 from 68 (Bouwman et al., 2009) to 236 Tg N_r (Tilman et al., 2001). In contrast to Bouwman et al. (2009) and Erisman et al. (2008), who also created scenarios based on the SRES storylines, our highest estimate is the A2 scenario, while the other two models have the A1 scenario as highest scenario. Also, our scenarios have in general a higher fertilizer consumption, especially compared to Bouwman et al. (2009). This may be

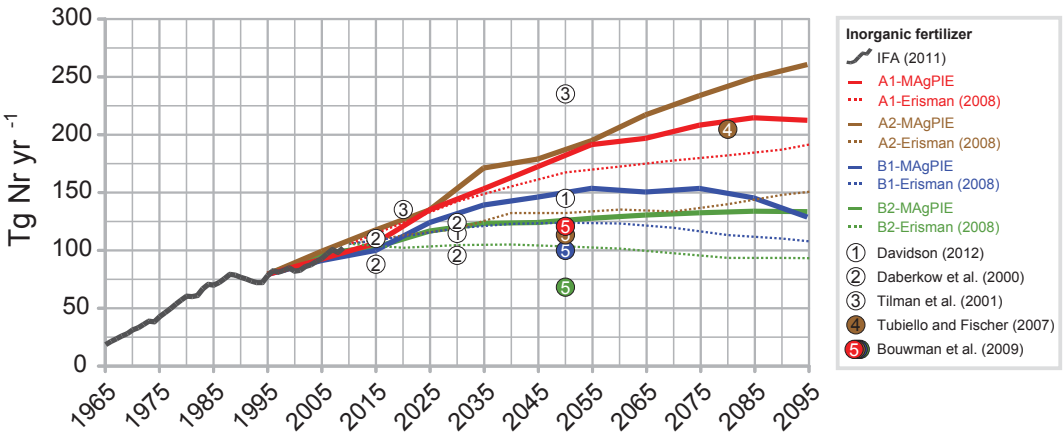

Fig. 3. Fertilizer consumption: historic dataset of IFADATA (2011), SRES scenario estimates by Erisman et al. (2008), Bouwman et al. (2009), Tubiello and Fischer (2007) and our study, as well as other estimates by Davidson (2012), Daberkow et al. (2000) and Tilman et al. (2001).

rooted in a different scenario parametrisation and a different methodological approach: Our scenarios assume a strong demand increase also for relatively low income growth as we explained in Sect. 4.2. At the same time, low income growth goes along with slow efficiency improvements in production. The combined effects explain the strong rise of inorganic fertilizer consumption in the A2 scenario. At the same time, our estimates are based on a top-down approach, compared to the bottom-up approach of Bouwman et al. (2009, 2011) or Daberkow et al. (2000). Both approaches have advantages and disadvantages. Data availability for bottom-up estimates of fertilizer application is currently poor, and may be biased by crop-rotations and different manure application rates. Our top-down approach has the disadvantage that it has to rely on an exogenous path for the development of N_r uptake efficiency. Also, as the closing entry of the budget, it accumulates the errors of other estimated N_r flows. But the top-down approach has the advantage that it can consistently simulate substitution effects between different N_r sources or a change in crop composition. This is of special importance if one simulates large structural shifts in the agricultural system like an increasing importance of the livestock sector.

Data on historic fertilizer consumption is provided by IFADATA (2011) and FAOSTAT (2011). Both estimates diverge, as they use different data sources and calendar years. On a regional level, differences can be substantial. FAO's estimate for fertilizer consumption in China in the year 2002 is 13 % higher than the estimate by IFA. As IFADATA (2011) provides longer continuous time series, we will refer to this dataset in the following. Fertilizer consumption between 1995 and 2009 (IFADATA, 2011) grows by +1.8 % per year. The estimates of Daberkow et al. (2000) and Bouwman et al. (2009, 2011) show lower growth rates of −0.4 % to +1.7 % over the regarded period of 20 to 50 yr. Our 50 yr average growth rate also stays with +0.9 % (B1) to +1.7 % (A2) below the observations. Yet, our short-term growth rate from 1995 to 2005 captures the observed development with a range of +1.5 % (B1) to +2.4 % (A2) between the scenarios. Due to trade our regional fertilizer projections are more uncertain than the global ones (see Sect. 4.2). Our results still meet the actual consumption trends of the last decades for most regions. However, fertilizer consumption in India rises slower than in the past or even stagnates, while the Pacific OECD region shows a strong increase in fertilizer consumption.

The range of our scenario outcomes is large for all N_r flows, and continues to become larger over time. It can be observed that the assumptions on which the globalised and environmentally oriented scenarios are based lead to a substantially lower turnover of the N_r cycle than the regional fragmented and economically oriented scenarios.

4.4 The importance of the livestock sector

The agricultural N_r cycle is dominated by the livestock sector. According to our calculations, livestock feeding appropriates 40 % (25 Tg) of N_r in global crop harvests and one third (11 Tg) of N_r in aboveground crop residues. Conversion byproducts add another 13 Tg N_r to the global feed mix. Moreover, 70 Tg N_r may be grazed by ruminants on pasture land, even though this estimate is very uncertain due to poor data availability on grazed biomass and N_r content of grazed pasture. The feed intake of 123 Tg results in solely 8 Tg N_r in livestock products.

In developed countries, the relative share of animal calories in total consumption already declined in the last decades. However, developing and transition countries still feature a massive increase in livestock consumption (FAOSTAT, 2011). According to our food demand projections, the rising global demand for livestock products will not end before the middle of the century. In the second half of the century, both an upward or a downward trend is possible.

More efficient livestock feeding will not necessarily relieve the pressure from the N_r cycle. Although the trend towards energy efficient industrial livestock feeding may reduce the demand for feed, this also implies a shift from pasture grazing, crop residues and conversion byproducts towards feedstock crops. Pasture grazing and crop residues do not have the required nutrient-density for highly productive livestock systems (Wirsenius, 2000). According to our calculations, conversion byproducts today provide one fourth of the proteins fed to animals in developed regions. Latin America exports twice as much N_r in conversion byproducts as in crops. At the same time, Europe cannot settle its conversion byproduct demand domestically. Conversion byproducts will not be sufficiently available if current industrialised feeding practices are adopted by other regions. The feedstock crops required to substitute conversion byproducts, pasture and crop residues will put additional pressure on the cropland N_r flows. The pressure on pasture however will most likely be only modest.

4.5 The future expansion of N_r pollution

All N_r that is not recycled within the agricultural sector is a potential environmental threat. Bouwman et al. (2009) estimate that over the next 50 yr, only 40–60 % of the lost N_r will be directly denitrified. The remaining N_r will either volatilise in the form of N_2O, NO_x and NH_y or leach to water bodies. With the surge of the N_r cycle, air, water and atmospheric pollution will severely increase, which has strong negative consequences for human health, ecosystem services and the stability of ecosystems.

Along with local and regional impacts, it is still under debate whether a continuous accumulation of N_r could destabilize the earth system as a whole (Rockström et al., 2009a,b). While there is little evidence supporting abrupt changes on a global level, N_r pollution contributes gradually to global phenomena such as biodiversity loss, ozone depletion and global warming. For the latter two, N_2O emissions play a crucial role. N_2O, is currently the single most important ozone depleting substance, as it catalyses the destruction of stratospheric ozone (Ravishankara et al., 2009). In addition, N_2O has an extraordinarily long atmospheric lifetime and absorbs infrared radiation in spectral windows not covered by other greenhouse gases (Vitousek et al., 1997). Fortunately, the greenhouse effect of N_2O might be offset by NO_x and NH_y emissions. By reducing the atmospheric lifetime of CH_4, scattering light and increasing biospheric carbon sinks, these emissions have a cooling effect (Butterbach-Bahl et al., 2011).

According to our calculations, N_2O emissions from managed soils and manure contributed 3.9 Tg N_2O-N, or approximately half of total anthropogenic N_2O emissions (Vuuren et al., 2011). However, the uncertainty involved is high. The result of our Monte Carlo variation of the emission parameters suggests that the emissions may lie with a 90 % probability in the range of 3.0 to 4.9 Tg N_2O-N. This only covers parts of the uncertainty, as the underlying activity data is also uncertain. Finally, actual agricultural emissions should be slightly higher than our estimate, as we do not cover all agricultural N_2O emission sources of the National Greenhouse Gas Inventories (Eggleston et al., 2006) and as also these inventories have no full coverage. Crutzen et al. (2008), using a top-down approach, estimate total agricultural N_2O emissions in 2000 to be in the range of 4.3 to 5.8 Tg N_2O-N, which is modestly higher than our estimate of 3.4 to 5.5 (90 % confidence, mean: 4.4) Tg N_2O-N in the year 2000.

Compared to the SRES marker scenarios (Nakicenovic et al., 2000), our results suggest that emissions will increase with substantially higher growth rates in the first half of the century. Especially in the case of the A1 and B2 scenarios, we come to 66 % (A1) and 36 % (B2) higher cumulative emissions over the century. In scenario A2 our estimates are continuously approximately 20 % lower (A2), while in the B1 scenario cumulative emissions are 6 % higher (B1) but occur later in the century (Fig. 3). None of our agricultural N_2O emission scenarios would be compatible with the RCP2.6 scenario, which keeps the radiative forcing below 2.6 $\frac{W}{m^2}$ in 2100 (Moss et al., 1998). To reach a sustainable climate target, explicit GHG mitigation efforts would therefore be required even in optimistic scenarios. If the non-agricultural N_2O emissions grow in similar pace than agricultural N_2O emissions, the A2 scenario might even outpace the RCP8.5 scenario.

In the beginning of the century, the uncertainty of emission parameters is much larger than the spread of scenario mean values. Only in the second half of the century, the differences of the scenarios are of similar magnitude to the emission parameter uncertainty. While the scenarios are just representative pathways and have no pretension to cover a specific probability space, this still indicates that a better representation of the underlying biophysical processes would largely improve our emission estimates.

5 Conclusions

The current state of the global agricultural N_r cycle is highly inefficient. Only around half of the N_r applied to cropland soils is taken up by plants. Furthermore, only one tenth of the N_r in cropland plant biomass and grazed pasture is actually consumed by humans. During the 21st century, our scenarios indicate a strong growth of all major flows of the N_r cycle. In the materialistic, unequal and fragmented A2 scenario, inorganic fertilizer consumption more than triples due to a strong population growth and slow improvement in N_r efficiencies in livestock and crop production. In the prosperous and materialistic A1 scenario, the strong increase of livestock consumption in the first half of the century and the industrialisation of livestock production quadruple the demand for N_r in feed crops already in 2045. In the heterogeneous,

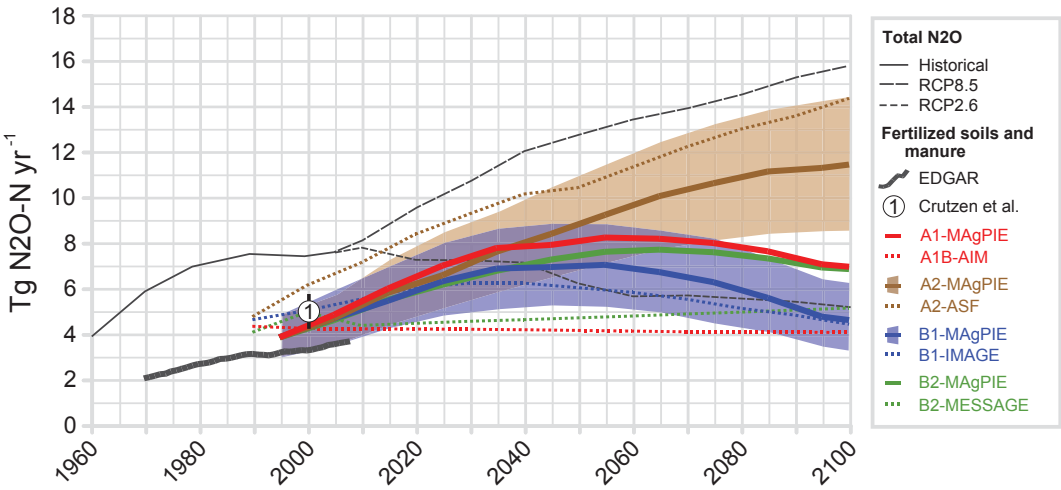

Fig. 4. Total anthropogenic N_2O emissions: historic emissions, highest and lowest RCP scenarios (Vuuren et al., 2011). N_2O emissions from soils and manure: historic estimates for 1970–2008 of the EDGAR 4.2 database (EC-JRC/PBL, 2011), a top-down estimate by Crutzen et al. (2008) for the year 2000, the SRES marker scenarios (Nakicenovic et al., 2000) for 1990–2100 and our scenarios for the SRES storylines for 1995–2095. The shaded areas represent a 90 % probability range in respect to the uncertainty of emission parameters of our A2 and B1 scenarios. Our A1 and B2 scenarios have a similar relative uncertainty range.

environmentally oriented B2 scenario, food demand is lower, especially in the first half of the century. However, the livestock sector productivity is improving only slowly and requires high amounts of N_r in feed. Finally, even in the globalised, equitable, environmental B1 scenario, N_r in harvested crops more than doubles and fertilizer consumption increases by 60 % and emissions by 23 % until the end of the century, with a peak in the middle of the century. In this scenario, the low meat consumption and large N_r efficiency improvements both in livestock and crop production are outbalanced by population growth and the catch-up of the less developed regions with the living standard of the rich regions.

Losses to natural systems will also continuously increase. This has negative consequences on both human health and local ecosystems. Moreover, it threatens the earth system as a whole by contributing to climate change, ozone depletion and loss of biodiversity. N_r mitigation is therefore one of the key global environmental challenges of this century.

Our model of the agricultural sector as a complex interrelated system shows that a large variety of dynamics influence N_r pollution. Each process offers a possibility of change, such that mitigation activities can take place not only where pollution occurs physically, but on different levels of the agricultural system: (a) already at the household level, the consumer has the choice to lower his N_r footprint by replacing animal with plant calories and reducing household waste (Popp et al., 2010; Leach et al., 2012); (b) substantial wastage during storage and processing could be avoided (Gustavsson et al., 2011); (c) information and price signals on the environmental footprint are lost within trade and retailing, such that sustainable products do not necessarily have a market advantage (Schmitz et al., 2012); (d) livestock products have potential to be produced more efficiently, both concerning the amount of N_r required for one ton of output and the composition of feed with different N_r footprints; (e) higher shares of animal manure and human sewage could be returned to farmlands (Wolf and Snyder, 2003); (f) nutrient uptake efficiency of plants could be improved by better fertilizer selection, timing and placing, as well as enhanced inoculation of legumes (Herridge et al., 2008; Roberts, 2007); (g) finally, unavoidable losses to natural systems could be directed or retained to protect vulnerable ecosystems (Jansson et al., 1994).

Appendix A

Extended methodology

A1 Model of Agricultural Production and its Impact on the Environment (MAgPIE): general description

MAgPIE is a global land-use allocation model which is linked with a grid-based dynamic vegetation model (LPJmL) (Bondeau et al., 2007; Sitch et al., 2003; Gerten et al., 2004; Waha et al., 2012). It takes into account regional economic conditions as well as spatially explicit data on potential crop yields and land and water constraints, and derives specific land-use patterns, yields and total costs of agricultural production for each grid cell. The following will provide only a brief overview of MAgPIE, as its implementation and validation is presented in detail elsewhere (Lotze-Campen et al., 2008; Popp et al., 2010, 2012; Schmitz et al., 2012).

The MAgPIE model works on three different levels of disaggregation: global, regional, and cluster cells. For the model-runs of this paper, the lowest disaggregation level

contains 500 cluster cells, which are aggregated from 0.5 grid cells based on an hierarchical cluster algorithm (Dietrich, 2011). Each cell has individual attributes concerning the available agricultural area and the potential yields for 18 different cropping activities derived from the LPJmL model. The geographic grid cells are grouped into ten economic world regions (Fig. 1). Each economic region has specific costs of production for the different farming activities derived from the GTAP model (Schmitz et al., 2010).

Food demand is inelastic and exogenous to the model, as described in further detail in the Sect. A4. Demand distinguishes between livestock and plant demand. Each calorie demand can be satisfied by a basket of crop or livestock products with fixed shares based on the historic consumption patterns. There is no substitution elasticity between the consumption of different crop products.

The demand for livestock calories requires the cultivation of feed crops. Weindl et al. (2010) uses a top-down approach to estimate feed baskets from the energy requirements of livestock, dividing the feed use from FAOSTAT (2011) between the five MAgPIE livestock categories.

Two virtual trading pools are implemented in MAgPIE which allocate the demand to the different supply regions. The first pool reflects the situation of no further trade liberalisation in the future and minimum self-sufficiency ratios derived from FAOSTAT (2011) are used for the allocation. Self-sufficiency ratios describe how much of the regional agricultural demand quantity is produced within a region. The second pool allocates the demand according to comparative advantage criteria to the supply regions. Assuming full liberalisation, the regions with the lowest production costs per ton will be preferred. More on the methodology can be found in Schmitz et al. (2012).

The non-linear objective function of the land-use model is to minimise the global costs of production for the given amount of agricultural demand. For this purpose, the optimization process can choose endogenously the share of each cell to be assigned to a mix of agricultural activities, the share of arable land left out of production, the share of non-arable land converted into cropland at exogenous land conversion costs and the regional distribution of livestock production. Furthermore, it can endogenously acquire yield-increasing technological change at additional costs (Dietrich, 2011). For future projections, the model works in time steps of 10 yr in a recursive dynamic mode, whereby the technology level of crop production and the cropland area is handed over to the next time step.

The calculations in this paper are created with the model-revision 4857 of MAgPIE. While a mathematical description of the core model can be found in the Supplement, the following Sects. A2, A3 and A4 explain the model extensions which are implemented for this study. The interface between the core model and the nutrient module consists of cropland area ($X^{\text{area}}_{t,j,v,w}$), crop and livestock dry-matter production ($\mathrm{P}(x_t)^{\text{prod}}_{t,i,k}$) and its use ($\mathrm{P}(x_t)^{\text{ds}}_{t,i,k,u}$). All parameters are described in Table A2. The superscripts are no exponents, but part of the parameter name. The arguments in the subscripts of the parameters include most importantly time (t), regions (i), crop types (v) and livestock types (l) (Table A1).

A2 Crop residues and conversion byproducts

A2.1 Crop residues

Eggleston et al. (2006) offer one of the few consistent datasets to estimate both aboveground (AG) and belowground (BG) residues. Also, by providing crop-growth functions (CGF) instead of fixed harvest indices, it can well describe current international differences of harvest indices and also their development in the future. The methodology is thus well eligible for global long-term modelling. Eggleston et al. (2006) provide linear CGFs with positive intercept for cereals, leguminous crops, potatoes and grasses. As no values are available for the oilcrops rapeseed, sunflower, and oilpalms as well as sugar crops, tropical roots, cotton and others, we use fixed harvest indices for these crops based on (Wirsenius, 2000; Lal, 2005; Feller et al., 2007). If different CGFs are available for crops within a crop group, we build a weighted average based on the production in 1995. The resulting parameters $r^{\text{cgf\_i}}_v$, $r^{\text{cgf\_s}}_v$ and $r^{\text{cgf\_r}}_v$ are displayed in Table A3. The AG crop residue production $\mathrm{P}(x_t)^{\text{prod\_ag}}_{t,i,v}$ is calculated as a function of harvested production $\mathrm{P}(x_t)^{\text{prod}}_{t,i,v}$ and the physical area $X^{\text{area}}_{t,j,v,w}$, and BG crop production as a function of total aboveground biomass.

$$\mathrm{P}(x_t)^{\text{prod\_ag}}_{t,i,v} := \sum_{j\in I_i,w} X^{\text{area}}_{t,j,v,w}\cdot r^{\text{cgf\_i}}_v + \mathrm{P}(x_t)^{\text{prod}}_{t,i,v}\cdot r^{\text{cgf\_s}}_v \tag{A1}$$

$$\mathrm{P}(x_t)^{\text{prod\_bg}}_{t,i,v} := (\mathrm{P}(x_t)^{\text{prod}}_{t,i,v} + \mathrm{P}(x_t)^{\text{prod\_ag}}_{t,i,v})\cdot r^{\text{cgf\_r}}_v \tag{A2}$$

While it is assumed that all BG crop residues remain on the field, the AG residues are assigned to four different categories: feed, on-field burning, recycling and other uses. Residues fed to livestock ($\mathrm{P}(x_t)^{\text{ds\_ag}}_{t,i,v,\text{feed}}$) are calculated based on livestock production and livestock and regional specific residue feed baskets $r^{\text{fb\_ag}}_{t,i,l,v}$ from Weindl et al. (2010). The demand rises with the increase in livestock production $\mathrm{P}(x_t)^{\text{prod}}_{t,i,l}$ and can be settled either by residues $\mathrm{P}(x_t)^{\text{ds\_ag}}_{t,i,v,\text{feed}}$ or by additional feedstock crops $\mathrm{P}(x_t)^{\text{ds}}_{t,i,l,v,\text{sag}}$. The latter prevents that crops are produced just for their residues.

$$\sum_v \mathrm{P}(x_t)^{\text{ds\_ag}}_{t,i,v,\text{feed}} = \sum_{l,v}(\mathrm{P}(x_t)^{\text{prod}}_{t,i,l}\cdot r^{\text{fb\_ag}}_{t,i,l,v} - \mathrm{P}(x_t)^{\text{ds}}_{t,i,l,v,\text{sag}}) \tag{A3}$$

Residue burning ($\mathrm{P}(x_t)^{\text{ds\_ag}}_{t,i,v,\text{burn}}$) is fixed to 15 % of total AG crop residue dry matter in developed and 25 % in developing

Table A1. Attributes.

Set	Description	Elements
t	timesteps	y1995 (1), y2005 (2) .. y2095 (11)
i	economic world regions	AFR, CPA, EUR, FSU, LAM, MEA, NAM, PAO, PAS, SAS (Fig. 1)
j	cells, each assigned to a region i ($I_{AFR}=\{1..30\}$,...)	1:300
w	irrigation	irrigated, rainfed
v	crops	temperate cereals, maize, tropical cereals, rice, soybeans, rapeseed, groundnut, sunflower, oilpalm, pulses, potatoes, tropical roots, sugar cane, sugar beet, fodder crops, fibres, others
l	livestock	ruminant livestock, non-ruminant livestock, poultry, eggs, milk
k	products	$v \cup l$
f	feeding systems	grazing on cropland (grazc), grazing on pasture (grazp), animal houses (house)
c	animal waste management systems	anaerobic lagoons, liquid/slurry, solid storage, daily spread, anaerobic digester, chicken layers, pit storage < 1 month, pit storage > 1 month, others
u	product use	food (food), feed (feed), seed (seed), other use (other), substitution for byproducts (sby), substitution for aboveground crop residues (sag)
r	AG residue use	feed (feed), recycling to soils (rec), burning in the field (burn), other use (other)
b	conversion byproduct use	feed (feed), other use (other)

regions for each crop. Other removals ($P(x_t)^{ds\_ag}_{t,i,l,v,other}$) are assumed to be only in developing regions of major importance and is set in these regions to 10 % of total residue dry matter production (Smil, 1999). All residues not assigned to feed, food, burning or other removals are assumed to remain in the field ($P(x_t)^{ds\_ag}_{t,i,v,rec}$). Trade of residues between regions is not considered.

$$P(x_t)^{prod\_ag}_{t,i,v} = \sum_r P(x_t)^{ds\_ag}_{t,i,v,r} \tag{A4}$$

A2.2 Conversion byproducts

Conversion byproducts are generated in the manufacturing of harvested crops into processed food. Of major importance are press cakes from oil production, molasses and bagasses from sugar refinement and brans from cereal milling. While they are also consumed as food, used for bioenergy production or as fertilizer, their most important usage lies currently in livestock feeding. Until recently, they were also reported in FAOSTAT. As the feed baskets used by MAgPIE from Weindl et al. (2010) are not in line with the then unpublished but probably more accurate statistics of FAOSTAT (2011), we decided to use the latter estimates on production and use (for feed or other purposes). We distributed the byproducts between the different livestock production types proportional to their energy in the feed baskets from Weindl et al. (2010) to create livestock-specific feed baskets for conversion byproducts $r^{fb\_by}_{t,i,l,v}$.

In the model, the production of 8 different conversion byproducts $P(x_t)^{prod\_by}_{t,i,v}$ (brans, molasses and 6 types of oilcakes) is linked to the total domestic supply $\sum_u P(x_t)^{ds}_{t,i,v,u}$ of their belonging crop groups (Table A3.1) by a factor $r^{by\_conv}_{i,v}$ fixed to the ratio of conversion byproduct production to their belonging crop domestic supply in 1995 (FAOSTAT, 2011). If the demand for byproducts is higher than the production, byproducts from other regions can be imported or the model can also feed feedstock crops $P(x_t)^{ds}_{t,i,l,v,sby}$.

$$P(x_t)^{prod\_by}_{t,i,v} := \sum_u P(x_t)^{ds}_{t,i,v,u} \cdot r^{by\_conv}_{i,v} \tag{A5}$$

$$P(x_t)^{ds\_by}_{t,i,v,feed} = \sum_l (P(x_t)^{prod}_{t,i,l} \cdot r^{fb\_by}_{t,i,l,v} - P(x_t)^{ds}_{t,i,l,v,sby}) \tag{A6}$$

$$\sum_i P(x_t)^{prod\_by}_{t,i,v} = \sum_{i,b} P(x_t)^{ds\_by}_{t,i,v,b} \tag{A7}$$

A3 N_r flows

A3.1 Attributes of plant biomass, conversion byproducts and food

The parametrisation of the goods represented in the model is a core task in a material flow model. From the literature, we derived N_r content of dry matter of harvested organs $r^{Nharvest}_v$ (Wirsenius, 2000; Fritsch, 2007; FAO, 2004; Roy et al., 2006), aboveground crop residues r^{Nag}_v (Wirsenius, 2000; Fritsch, 2007; FAO, 2004; Eggleston et al., 2006; Chan and Lim, 1980), belowground crop residues r^{Nbg}_v(Eggleston et al., 2006; Fritsch, 2007; Wirsenius, 2000; Khalid et al., 2000) and conversion byproducts r^{Nby}_v(Wirsenius, 2000; Roy et al., 2006) (Table A3.1). For the aggregation to MAgPIE crop groups, we weighted the parameters of each crop group with its global dry matter biomass in 1995. In the case of missing values for a specific FAO crop, we adopted the parametrisation of a selected representative crop of its crop group (e.g. we assign the value of wheat, being the representative crop of *temperate cereals*, to the FAO item *mixed grain*). The N_r in crop and residue production and its subsequent use is thus

Table A2. Parameters, descriptions and units (all units per year). The name of the equivalent parameter in Eggleston et al. (2006) is indicated in brackets.

Parameter	Description	Unit
Area		
$X^{\text{area}}_{t,j,v,w}$	Cropland area under cultivation	Mha
$\mathrm{P}(x_t)^{\text{landconv}}_{t,j}$	Land conversion	Mha
Production		
$\mathrm{P}(x_t)^{\text{prod}}_{t,i,k}$ $\mathrm{N}(x_t)^{\text{prod}}_{t,i,k}$	Crop production	TgDM $\mathrm{TgN_r}$
$\mathrm{P}(x_t)^{\text{prod\_ag}}_{t,i,v}$ $\mathrm{N}(x_t)^{\text{prod\_ag}}_{t,i,v}$	AG residue production	TgDM $\mathrm{TgN_r}$
$\mathrm{P}(x_t)^{\text{prod\_bg}}_{t,i,v}$ $\mathrm{N}(x_t)^{\text{prod\_bg}}_{t,i,v}$	BG residue production	TgDM $\mathrm{TgN_r}$
$\mathrm{P}(x_t)^{\text{prod\_by}}_{t,i,v}$ $\mathrm{N}(x_t)^{\text{prod\_by}}_{t,i,v}$	Conversion byproduct production	TgDM $\mathrm{TgN_r}$
Domestic supply and its use		
$\mathrm{P}(x_t)^{\text{ds}}_{t,i,v,u}$ $\mathrm{N}(x_t)^{\text{ds}}_{t,i,v,u}$	Crop use	TgDM $\mathrm{TgN_r}$
$\mathrm{P}(x_t)^{\text{ds\_ag}}_{t,i,v,r}$ $\mathrm{N}(x_t)^{\text{ds\_ag}}_{t,i,v,r}$	AG residues use	TgDM $\mathrm{TgN_r}$
$\mathrm{P}(x_t)^{\text{ds\_by}}_{t,i,v,b}$ $\mathrm{N}(x_t)^{\text{ds\_by}}_{t,i,v,b}$	Conversion byproduct use	TgDM $\mathrm{TgN_r}$
$\mathrm{N}(x_t)^{\text{fs}}_{t,i,k}$	Food supply	$\mathrm{TgN_r}$
$r^{\text{int}}_{t,i,k}$	Intake share of food supply	$\frac{\mathrm{TgN_r}}{\mathrm{TgDM}}$
$\mathrm{N}(x_t)^{\text{int}}_{t,i,k}$	Intake	$\mathrm{TgN_r}$
P^{tb}_t	Trade balance reduction	1

Table A2. Continued.

Parameter	Description	Unit
Crop growth functions, processing rates and biological fixation		
$r^{\text{cgf\_i}}_v$	AG residues intercept	$\frac{\mathrm{TgDM}}{\mathrm{Mha}}$
$r^{\text{cgf\_s}}_v$	AG residues slope	$\frac{\mathrm{TgDM}}{\mathrm{TgDM}}$
$r^{\text{cgf\_r}}_v$	AG to BG biomass ratio	$\frac{\mathrm{TgDM}}{\mathrm{TgDM}}$
$r^{\text{by\_conv}}_{\text{i,v}}$	Conversion byproducts generated per unit of crop production	$\frac{\mathrm{TgDM}}{\mathrm{TgDM}}$
r^{ndfa}_v	Plant $\mathrm{N_r}$ derived from atmospheric fixation	$\frac{\mathrm{TgN_r}}{\mathrm{TgN_r}}$
r^{Nfix}_v	Fixation of free-living bacteria	$\frac{\mathrm{TgN_r}}{\mathrm{TgMha}}$
Products		
r^{Nharvest}_v	$\mathrm{N_r}$ content of harvested crops	$\frac{\mathrm{TgN_r}}{\mathrm{TgDM}}$
r^{Nag}_v	$\mathrm{N_r}$ content of AG residues	$\frac{\mathrm{TgN_r}}{\mathrm{TgDM}}$
r^{Nbg}_v	$\mathrm{N_r}$ content of BG residues	$\frac{\mathrm{TgN_r}}{\mathrm{TgDM}}$
r^{Npast}	$\mathrm{N_r}$ content of grazed pasture	$\frac{\mathrm{TgN_r}}{\mathrm{TgDM}}$
r^{Nby}_v	$\mathrm{N_r}$ content of conversion byproducts	$\frac{\mathrm{TgN_r}}{\mathrm{TgDM}}$
r^{PR}_l	Protein content of livestock products	$\frac{\mathrm{TgPr}}{\mathrm{TgDM}}$
r^{NtoPR}_l	Protein to $\mathrm{N_r}$ content ratios	$\frac{\mathrm{TgN_r}}{\mathrm{TgPr}}$

obtained as follows:

$$\mathrm{N}(x_t)^{\text{prod}}_{t,i,v} := \mathrm{P}(x_t)^{\text{prod}}_{t,i,v} \cdot r^{\text{Nharvest}}_v \tag{A8}$$

$$\mathrm{N}(x_t)^{\text{prod\_ag}}_{t,i,v} := \mathrm{P}(x_t)^{\text{prod\_ag}}_{t,i,v} \cdot r^{\text{Nag}}_v \tag{A9}$$

$$\mathrm{N}(x_t)^{\text{prod\_bg}}_{t,i,v} := \mathrm{P}(x_t)^{\text{prod\_bg}}_{t,i,v} \cdot r^{\text{Nbg}}_v \tag{A10}$$

$$\mathrm{N}(x_t)^{\text{ds}}_{t,i,v,u} := \mathrm{P}(x_t)^{\text{ds}}_{t,i,v,u} \cdot r^{\text{Nharvest}}_v \tag{A11}$$

$$\mathrm{N}(x_t)^{\text{ds\_ag}}_{t,i,v,r} := \mathrm{P}(x_t)^{\text{ds\_ag}}_{t,i,v,r} \cdot r^{\text{Nag}}_v \tag{A12}$$

.

A3.2 Manure management

Feed $\mathrm{N_r}$ is assigned to three feeding systems (f): pasture grazing (grazp), cropland grazing (grazc) and animal houses (house). All $\mathrm{N_r}$ from pasture was assigned to grazp. $\mathrm{N_r}$ in feedstock crops and conversion byproducts is assumed to be eaten in confinement houses. Crop residues in developed regions are fully assigned to house, while in developing regions we assume that 25 % of the $\mathrm{N_r}$ in residues are consumed directly on croplands during stubble grazing ($r^{\text{grazC}}_{t,i}$).

$$\mathrm{N}(x_t)^{\text{feed}}_{t,i,l,\text{grazp}} := r^{\text{fb\_past}}_{t,i,l} \cdot \mathrm{P}(x_t)^{\text{prod}}_{t,i,l} \cdot r^{\text{Npast}} \tag{A13}$$

$$\mathrm{N}(x_t)^{\text{feed}}_{t,i,l,\text{grazc}} := \sum_v r^{\text{fb\_ag}}_{t,i,l,v} \cdot \mathrm{P}(x_t)^{\text{prod}}_{t,i,l} \cdot r^{\text{Nag}}_v \cdot r^{\text{grazC}}_{t,i} \tag{A14}$$

$$\begin{aligned}\mathrm{N}(x_t)^{\text{feed}}_{t,i,l,\text{house}} := \sum_v \Big(& r^{\text{fb\_by}}_{t,i,l,v} \cdot \mathrm{P}(x_t)^{\text{prod}}_{t,i,l} \cdot r^{\text{Nby}}_v \\ & + r^{\text{Nharvest}}_v \cdot (r^{\text{fb\_conc}}_{t,i,l,v} \cdot \mathrm{P}(x_t)^{\text{prod}}_{t,i,l} \\ & + \mathrm{P}(x_t)^{\text{ds}}_{t,i,l,v,\text{sby}} + \mathrm{P}(x_t)^{\text{ds}}_{t,i,l,v,\text{sag}}) \\ & + r^{\text{fb\_ag}}_{t,i,l,v} \cdot \mathrm{P}(x_t)^{\text{prod}}_{t,i,l} \cdot r^{\text{Nag}}_v \cdot (1 - r^{\text{grazC}}_{t,i}) \Big)\end{aligned} \tag{A15}$$

In a second step, we use a top-down approach to estimate regional livestock specific annual average $\mathrm{N_r}$ excretion rates, rooted in the Tier 2 methodology of Eggleston et al. (2006). From the feed in all feeding systems (f) we subtract the amount of $\mathrm{N_r}$ which is integrated into animal biomass

Table A2. Continued.

Parameter	Description	Unit
Livestock		
$r^{\text{fb\_conc}}_{t,i,l,v}$	Feedstock crops in feed basket	$\frac{\text{TgDM}}{\text{TgDM}}$
$r^{\text{fb\_ag}}_{t,i,l,v}$	AG residues in feed basket	$\frac{\text{TgDM}}{\text{TgDM}}$
$r^{\text{fb\_past}}_{t,i,l}$	Grazed pasture in feed basket	$\frac{\text{TgDM}}{\text{TgDM}}$
$r^{\text{fb\_by}}_{t,i,l,v}$	Byproducts in feed basket	$\frac{\text{TgDM}}{\text{TgDM}}$
$r^{\text{grazC}}_{t,i}$	Fraction of feed residues consumed during stubble grazing	$\frac{\text{TgDM}}{\text{TgDM}}$
$\text{N}(x_t)^{\text{feed}}_{t,i,l,f}$	Feed N_r distributed to livestock types in feeding systems	$\frac{\text{TgN}_r}{\text{TgN}_r}$
r^{sl}_{l}	Ratio between marketable product and whole body weight	$\frac{\text{TgDM}}{\text{TgDM}}$
r^{Nl}_{l}	Whole body N_r content	$\frac{\text{TgN}_r}{\text{TgDM}}$
$\text{N}(x_t)^{\text{sl}}_{t,i,l}$	N_r in whole animal bodies	TgN_r
$r^{\text{fs}}_{t,i,l,f}$	Fraction of manure in feeding system (based on $\text{MS}_{(\text{T,S})}$)	$\frac{\text{TgN}_r}{\text{TgN}_r}$
$r^{\text{cs}}_{t,i,l,c}$	Fraction of manure managed in animal waste management systems (based on $\text{MS}_{(\text{T,S})}$)	$\frac{\text{TgN}_r}{\text{TgN}_r}$
$\text{N}(x_t)^{\text{ex}}_{t,i,l,f}$	N_r in excretion ($\text{Nex}_{(\text{T})}$)	TgN_r
$r^{\text{fuel}}_{t,i,l}$	Fraction of manure collected for fuel	$\frac{\text{TgN}_r}{\text{TgN}_r}$
$\text{N}(x_t)^{\text{closs}}_{t,i}$	Manure N_r lost in animal houses and waste management	TgN_r

$\text{N}(x_t)^{\text{sl}}_{t,i,l}$ and assume that the remaining N_r is excreted as manure. For meat products, we calculate the N_r in the whole animal body $\text{N}(x_t)^{\text{sl}}_{t,i,l}$ using livestock product to whole body ratios r^{sl}_{l} from Wirsenius (2000), and whole body N_r content r^{Nl}_{l} based on Poulsen and Kristensen (1998) (Table A5). For milk and eggs, we calculate $\text{N}(x_t)^{\text{sl}}_{t,i,l}$ by the N_r content in milk and eggs (Poulsen and Kristensen, 1998) (Table A5). $\text{N}(x_t)^{\text{sl}}_{t,i,l}$ is assigned to one of the three feeding systems by the parameter $r^{\text{fs}}_{t,i,l,f}$, which is based on Eggleston et al. (2006).

$$\text{N}(x_t)^{\text{sl}}_{t,i,l} := \text{P}(x_t)^{\text{prod}}_{t,i,l}\,\frac{r^{\text{Nl}}_{l}}{r^{\text{sl}}_{l}} \tag{A16}$$

$$\text{N}(x_t)^{\text{ex}}_{t,i,l,f} := \text{N}(x_t)^{\text{feed}}_{t,i,l,f} - r^{\text{fs}}_{t,i,l,f}\cdot\text{N}(x_t)^{\text{sl}}_{t,i,l} \tag{A17}$$

In a third step, the N_r excreted in animal houses is divided between 9 animal waste management systems (c) using the parameter $r^{\text{cs}}_{t,i,l,c}$. When available, we used the regional and livestock specific shares from Eggleston et al. (2006); for chicken, sheep, goats and other animals, we used the default parameters of IPCC (1996). The category *others* for chicken is assumed to be *poultry with litter*.

Not all the manure excreted in animal houses is recycled within the agricultural system, but large fractions are lost to volatilisation and leaching or is simply not brought out to the farmland. We use animal waste management system specific

Table A2. Continued.

Parameter	Description	Unit
Soil Budget		
$\text{N}(x_t)^{\text{withd}}_{t,i}$	Soil N_r withdrawals	TgN_r
$\text{N}(x_t)^{\text{inp}}_{t,i}$	Soil N_r inputs	TgN_r
$\text{N}(x_t)^{\text{loss}}_{t,i}$	Soil N_r losses	TgN_r
$r^{\text{SNUpE}}_{t,i}$	Cropland soil N_r uptake efficiency	$\frac{\text{TgN}_r}{\text{TgN}_r}$
$\text{N}(x_t)^{\text{dep}}_{t,i}$	Atmospheric deposition of N_r	TgN_r
$\text{N}(x_t)^{\text{volat}}_{t,i}$	Volatilisation of NO_x and NH_y	TgNO_xNH_y
$\text{N}^{\text{som}}_{t,i}$	N_r release by soil organic matter loss (F_{SOM})	TgN_r
$r^{\text{som}}_{t,j}$	N_r release by soil organic matter loss	$\frac{\text{TgN}_r}{\text{Mha}}$
$\text{N}(x_t)^{\text{fert}}_{t,i}$	Inorganic N_r fertilizer (F_{SN})	TgN_r
$\text{N}(x_t)^{\text{res}}_{t,i}$	N_r in recycled AG and BG residues (F_{CR})	TgN_r
$\text{N}(x_t)^{\text{FixFree}}_{t,i}$	N_r fixed by free-living microorganisms (F_{CR})	TgN_r
$\text{N}(x_t)^{\text{m}}_{t,i}$	N_r in manure excreted in animal houses and applied to agricultural soils (F_{AM})	TgN_r
$r^{\text{msplit}}_{t,i}$	Fraction of manure in animal houses applied to cropland soils	$\frac{\text{TgN}_r}{\text{TgN}_r}$
$\text{N}(x_t)^{\text{m\_cs}}_{t,i}$	N_r in manure applied or excreted on cropland soils	TgN_r
$\text{N}(x_t)^{\text{m\_ps}}_{t,i}$	N_r in manure applied or excreted on pasture soils	TgN_r
Emissions		
$r^{\text{gas\_fert}}$	Fraction of industrial fertilizer N_r that volatises as NO_x and NH_y ($\text{Frac}_{\text{GasF}}$)	$\frac{\text{TgNO}_x\text{NH}_y}{\text{TgN}_r}$
$r^{\text{gas\_awms}}_{l,c}$	Fraction of manure N_r that volatises in waste management facilities as NO_x and NH_y ($\text{Frac}_{\text{GasMS}}$)	$\frac{\text{TgNO}_x\text{NH}_y}{\text{TgN}_r}$
$r^{\text{loss\_awms}}_{l,c}$	Fraction of manure N_r that is lost in waste management ($\text{Frac}_{\text{LossMS}}$)	$\frac{\text{TgNO}_x\text{NH}_y}{\text{TgN}_r}$

Table A2. Continued.

Parameter	Description	Unit
$r^{\text{gas\_m}}$	Fraction of manure N_r that volatises during application as NO_x and NH_y ($\text{Frac}_{\text{GasM}}$)	$\frac{\text{TgNO}_x\text{NH}_y}{\text{TgN}_r}$
r^{leach}	Fraction of N_r that leaches to water bodies ($\text{Frac}_{\text{Leach-H}}$)	$\frac{\text{TgN}_r}{\text{TgN}_r}$
r_v^{CF}	Combustion factor for on-field residue burning (C_f)	$\frac{\text{TgN}_r}{\text{TgN}_r}$
r^{dir}	Direct emission factor for N inputs to managed soils (EF_1)	$\frac{\text{TgN}_2\text{O-N}}{\text{TgN}_r}$
$r^{\text{dir\_rice}}$	Direct emission factor for N inputs to flooded rice fields ($\text{EF}_{1\text{fr}}$)	$\frac{\text{TgN}_2\text{O-N}}{\text{TgN}_r}$
$r_c^{\text{dir\_house}}$	Direct emission factor for manure excreted in animal houses ($\text{EF}_{3(S)}$)	$\frac{\text{TgN}_2\text{O-N}}{\text{TgN}_r}$
$r_l^{\text{dir\_graz}}$	Direct emissions from manure excreted on pasture, range and paddock ($\text{EF}_{3\text{PRP}}$)	$\frac{\text{TgN}_2\text{O-N}}{\text{TgN}_r}$
$r^{\text{indir\_gas}}$	N_2O emission factor for volatised N_r (EF_{iv})	$\frac{\text{TgN}_2\text{O-N}}{\text{TgNO}_x\text{NH}_y}$
$r^{\text{indir\_leach}}$	N_2O emission factor for leached N_r (EF_{v})	$\frac{\text{TgN}_2\text{O-N}}{\text{TgN}_r}$
$N_2O(x_t)_{t,i}^{\text{fert}}$	N_2O from industrial fertilizer	$\text{TgN}_2\text{O}-\text{N}$
$N_2O(x_t)_{t,i}^{\text{res}}$	N_2O from crop residues	$\text{TgN}_2\text{O}-\text{N}$
$N_2O(x_t)_{t,i}^{\text{m}}$	N_2O from animal manure applied to croplands	$\text{TgN}_2\text{O}-\text{N}$
$N_2O(x_t)_{t,i}^{\text{past}}$	N_2O from pasture range and paddock	$\text{TgN}_2\text{O}-\text{N}$
$N_2O(x_t)_{t,i}^{\text{house}}$	N_2O from animal waste management systems	$\text{TgN}_2\text{O}-\text{N}$
$N_2O(x_t)_{t,i}^{\text{som}}$	N_2O from soil organic matter loss	$\text{TgN}_2\text{O}-\text{N}$

shares of the total amount of managed manure $r_{l,c}^{\text{loss\_awms}}$ not being recycled, including a fraction $r_{l,c}^{\text{gas\_awms}}$ that is lost in the form of volatilisation in the form of NO_x and NH_y. Because default parameters for $r_{l,c}^{\text{gas\_awms}}$ and $r_{l,c}^{\text{loss\_awms}}$ are not available for all animal waste management systems, we made the following assumptions: For pit storage < 1 month of swine manure, we used the lower value of the proposed range (0.15), and the upper value (0.3) for pit storage > 1 month. If no estimates are available, drylots and solid storage received the same emission factor, as was done in the old methodology (IPCC, 1996). Based on Marchaim (1992), we assumed that losses for manure managed in *anaerobic digesters* are negligible. In the absence of default parameters for $r_{t,i,l,c}^{\text{cs}}$ for chicken, sheep, goats and other animals, we used the default parameters of Eggleston et al. (2006). *Others*

Table A3. Estimates of crop growth functions: AG residues intercept ($r_v^{\text{cgf\_i}}$), slope ($r_v^{\text{cgf\_s}}$) and AG to BG biomass ratio ($r_v^{\text{cgf\_r}}$) (for sources see text).

Crop type (kcr)	$r_v^{\text{cgf\_i}}$	$r_v^{\text{cgf\_s}}$	$r_v^{\text{cgf\_r}}$
Temperate cereals	0.58	1.36	0.24
Tropical cereals	0.61	1.03	0.22
Maize	0.79	1.06	0.22
Rice	2.46	0.95	0.16
Soybeans	1.35	0.93	0.19
Rapeseed	0	1.86	0.22
Groudnut	1.54	1.07	0.19
Sunflower	0	1.86	0.22
Oilpalm	0	1.86	0.24
Pulses	0.79	0.89	0.19
Potatoes	1.06	0.10	0.20
Tropical roots	0	0.85	0.20
Sugar cane	0	0.67	0.07
Sugar beet	0	0.54	0.20
Others	0	0.39	0.22
Fodder	0.26	0.28	0.45
Fibres	0	1.48	0.13

Table A4. N_r contents of harvested crops (r_v^{Nharvest}), aboveground crop residues (r_v^{Nag}), belowground crop residues (r_v^{Nbg}) and conversion byproducts (r_v^{Nby}) for the MAgPIE crop types. All N_r contents are in % of dry matter biomass. Collected and aggregated from Wirsenius (2000), Fritsch (2007), Eggleston et al. (2006), FAO (2004), Roy et al. (2006), Chan and Lim (1980) and Khalid et al. (2000).

Crop type (v)	r_v^{Nharvest}	r_v^{Nag}	r_v^{Nbg}	r_v^{Nby}
Temperate cereals	2.17	0.74	0.98	2.93
Maize	1.60	0.88	0.70	
Tropical cereals	1.63	0.70	0.60	
Rice	1.28	0.70	0.90	
Soybeans	5.12	0.80	0.80	7.90
Rapeseed	3.68	0.81	0.81	6.43
Groudnut	2.99	2.24	0.80	7.28
Sunflower	2.16	0.80	0.80	5.92
Oilpalm	0.57	0.52	0.53	6.43
Pulses	4.21	1.05	0.80	
Potatoes	1.44	1.33	1.40	
Tropical roots	0.53	0.86	1.40	
Sugar cane	0.24	0.80	0.80	1.36
Sugar beet	0.56	1.76	1.40	
Others	2.85	0.81	0.70	5.72
Fodder	2.01	1.91	1.41	
Fibres	2.39	0.93	0.70	
Pasture	1.60			
Pasture	r^{Npast}			
Past	1.60			

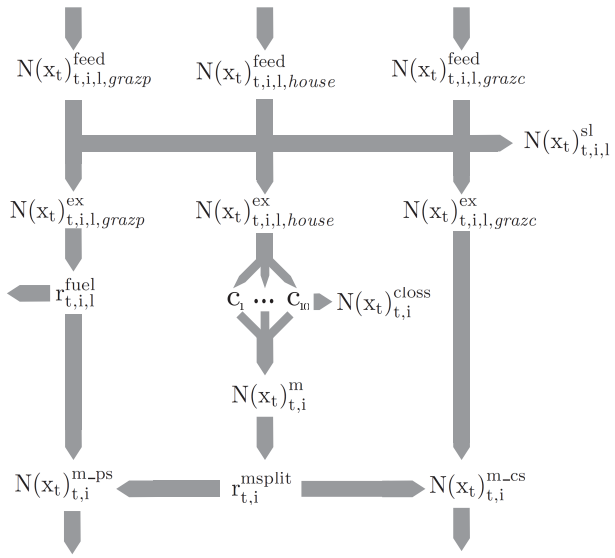

Fig. A1. Modelling N_r flows in the livestock sector.

Table A5. Estimates of whole body N_r content (r_l^{Nl}) in % of dry matter, and estimates of the ratio between marketable product and whole body weight (r_l^{sl}).

	r_l^{Nl}	r_l^{sl}
Ruminant livestock	6.3[a]	0.66[c]
Non-ruminant livestock	6.0[a]	0.81[c]
Poultry	7.1[a]	0.76[c]
Eggs	5.6[a]	1
Milk	4.6[b]	1

[a]Based on cows, market pigs, chicken and chicken eggs in Poulsen and Kristensen (1998).
[b]Based on milk with 3.5 % proteins in line with Smil (2002).
[c]Based on medium quality cows, swine and broilers from Wirsenius (2000).

is assumed to be *deep bedding* for pigs, cattle and others. All remaining gaps in the loss factors are filled with the values for cattle of the respective animal waste management system.

While all remaining manure in animal houses is fully applied to cropland soils in developing regions, we assume that in NAM and EUR only a fraction $r_{t,i}^{msplit}$ of 87 % and 66 % is returned on cropland soils (Liu et al., 2010b), while the rest is applied to pasture soils. Furthermore, in developing regions, a certain share of manure excreted on pasture is dedicated for household fuel and does not return to pasture soils (Eggleston et al., 2006). Because the N_r in fuel is leaving the agricultural sector, it is not further considered in this study, while the N_r from pasture grazing is assumed to be returned to pasture soils.

Losses of N_r in animal houses and waste handling ($N(x_t)_{t,i}^{closs}$), recycled manure ($N(x_t)_{t,i}^{m}$) and manure arriving on cropland soils ($N(x_t)_{t,i}^{m\_cs}$) and pasture soils ($N(x_t)_{t,i}^{m\_ps}$) are calculated as follows:

$$N(x_t)_{t,i}^{closs} := \sum_c N(x_t)_{t,i,l,\text{house}}^{ex} \cdot r_{t,i,l,c}^{cs} \cdot r_{l,c}^{loss\_awms} \quad \text{(A18)}$$

$$N(x_t)_{t,i}^{m} := \sum_c N(x_t)_{t,i,l,\text{house}}^{ex} \cdot r_{t,i,l,c}^{cs} \cdot (1 - r_{l,c}^{loss\_awms}) \quad \text{(A19)}$$

$$N(x_t)_{t,i}^{m\_cs} := N(x_t)_{t,i}^{m} \cdot r_{t,i}^{msplit} + \sum_l N(x_t)_{t,i,l,\text{grazc}}^{ex} \quad \text{(A20)}$$

$$N(x_t)_{t,i}^{m\_ps} := N(x_t)_{t,i}^{m} \cdot (1 - r_{t,i}^{msplit}) + \sum_l N(x_t)_{t,i,l,\text{grazp}}^{ex} \cdot (1 - r_{t,i,l}^{fuel}) \quad \text{(A21)}$$

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A3.3 Cropland N_r inputs

Inorganic fertilizer is the only N_r flow appearing in international statistics. We aggregate the values of IFADATA (2011) for all N_r fertilizer products to the 10 MAgPIE regions to determine $N(x_t)_{t,i}^{fert}$ in 1995. For the scenario analysis, inorganic fertilizer consumption is determined endogenously as described in Sect. A3.4.

The amount of crop residues left in the field is estimated as described in Sect. A2 as the remainder of the produced residues which are not used for feed, construction, fuel or burned in the field. While the nutrients of these residues are fully returned to cropland soils, the largest part of the N_r in the crop residues burned in the field (r_v^{CF}) is combusted; only a fraction of 10 % for temperate cereal residues and 20 % for all other residues (Eggleston et al., 2006) remains uncombusted and returns to cropland soils.

$$N(x_t)_{t,i}^{res} := \sum_v \left(N(x_t)_{t,i,v}^{prod\_bg} + N(x_t)_{t,i,v,\text{rec}}^{ds\_ag} + N(x_t)_{t,i,v,\text{burn}}^{ds\_ag} \cdot (1 - r_v^{CF}) \right) \quad \text{(A22)}$$

A major part of the N_r lost from field in the form of NO_x and NH_y as well as other N_r compounds from the combustion of fossil fuels are later on deposited from the atmosphere on cropland area. Based on spatial datasets for atmospheric deposition rates (Dentener, 2006) and cropland area (Klein Goldewijk et al., 2011a), we derive the regional atmospheric deposition on croplands $N(x_t)_{t=l,i}^{dep}$. As a large part of volatilised N_r will be deposited close to the emission source, the largest part of cropland atmospheric deposition probably stems from agricultural NO_x and NH_y. For the future we therefore assume that the deposition rates grow with the same growth rate as the agricultural NO_x and NH_y emissions $N(x_t)_{t,i}^{volat}$ (see Eq. (A38) in Sect. A3.5).

$$N(x_t)_{t,i}^{dep} := \frac{N(x_t)_{t,i}^{volat}}{N(x_t)_{t=l,i}^{volat}} \cdot N(x_t)_{t=l,i}^{dep} \quad \text{(A23)}$$

Table A6. Estimates of N_r fixation rates per area (r_v^{Nfix}) or as percentage of plant N_r (r_v^{ndfa}), based on Herridge et al. (2008) and aggregated to MAgPIE crop types.

Crop type	r_v^{Nfix} $\frac{\mathrm{TgN_r}}{\mathrm{Mha}}$	r_v^{ndfa} $\frac{\mathrm{TgN_r}}{\mathrm{TgN_r}}$
Temperate Cereals	0.005	–
Maize	0.005	–
Tropical Cereals	0.005	–
Rice	0.033	–
Soybeans	–	0.5[a], 0.6 [b], 0.8 [c], 0.68[d]
Rapeseed	0.005	–
Groudnut	–	0.5[a], 0.6 [b], 0.8 [c], 0.68[d]
Sunflower	0.005	–
Oilpalm	0.005	–
Pulses	–	0.53
Potatoes	0.005	–
Tropical roots	0.005	–
Sugar Cane	–	0.2[b], 0.1[d]
Sugar Beet	0.005	–
Others	0.005	–
Fodder	0.004	0.31
Fibres	0.005	–

[a]For the region CPA
[b]For the region LAM
[c]For the region NAM
[d]For all other regions

While plants are unable to fix nitrogen from N_2 in the atmosphere, some microorganisms are able to do this. These microorganisms either live free in soils, or in symbiosis with certain crops or cover crops. The symbiosis is typical mainly for leguminous crops (beans, groundnuts, soybean, pulses, chickpeas, alfalfa), which possess special root nodules in which the microorganisms live. Also, sugar cane can fix N_r in symbiosis with endophytic bacteria. In the case of rice paddies, free-living cyanobacteria and cyanobacteria living in symbiosis with the water-fern Azolla can also fix substantial amounts of N_r. While N_r fixation by leguminous plants has been well investigated, estimates for N_r fixation by sugar cane and free-living bacteria is much more uncertain or even speculative.

For legumes and sugar cane, where N_r fixation is the direct product of a symbiosis of the microorganisms with the crop, we assumed that fixation rates are proportional to the N_r in the plant biomass. The percentage of fixation-derived N_r is taken from Herridge et al. (2008). In the case of soybeans, groundnuts and sugarcane, fixation rates vary between regions to account for differences in management practices like fertilization or inoculation.

For legumes and sugar cane, where N_r fixation is the direct product of a symbiosis of the microorganisms with the crop, we assumed that fixation rates are proportional to the N_r in the plant biomass. The percentage of fixation-derived N_r is taken from Herridge et al. (2008). In the case of soybeans, groundnuts and sugarcane, fixation rates vary between regions to account for differences in management practices like fertilization or inoculation. N_r fixation by free-living bacteria in cropland soils and rice paddies does not necessarily depend on the biomass production of the harvested crop, so we used fixation rates per area r_v^{Nfix}. In the case of the MAgPIE crop types fodder and pulses, which contain crop species with different rates of N_r fixation, a weighted mean is calculated based on the relative share of biomass production in 1995 for r_v^{ndfa} or on the relative share of harvested area in 1995 for r_v^{Nfix} (Table A6). Our model does not cover that the fixation rates might change in the future due to the change of management practices. Improved inoculation of root nodules could increase fixation rates, while fertilization of legumes could reduce the biological fixation.

$$\mathrm{N}(x_t)_{t,i}^{\mathrm{FixFree}} := \sum_{j \in I_i, v, w} X_{t,j,v,w}^{\mathrm{area}} \cdot r_v^{\mathrm{Nfix}}) \tag{A24}$$

A certain share of the N_r in a plant is already incorporated in the seed. The amount of seed required for production $\mathrm{P}(x_t)_{t,i,v,\mathrm{seed}}^{\mathrm{ds}}$ is estimated crop and region specific using seed shares from FAOSTAT (2011).

$$\mathrm{N}(x_t)_{t,i,v,\mathrm{seed}}^{\mathrm{ds}} := \mathrm{P}(x_t)_{t,i,v,\mathrm{seed}}^{\mathrm{ds}} \cdot r_v^{\mathrm{Nharvest}} \tag{A25}$$

When pastureland or natural vegetation is transformed to cropland, soil organic matter (SOM) is lost. This also releases N_r for agricultural production. Total N_r release by SOM loss $\mathrm{N}_{t,i}^{\mathrm{som}}$ is estimated by multiplying the land conversion $\mathrm{P}(x_t)_{t,j}^{\mathrm{landconv}}$ in each grid cell with the yearly N_r losses per ha $r_{t,j}^{\mathrm{som}}$.

$$\mathrm{N}_{t,i}^{\mathrm{som}} = \sum_{j \in I_i} \left(\mathrm{P}(x_t)_{t,j}^{\mathrm{landconv}} \cdot r_{t,j}^{\mathrm{som}} \right) \tag{A26}$$

Land conversion $\mathrm{P}(x_t)_{t,j}^{\mathrm{landconv}}$ is calculated as the increase of $X_{t,j,v,w}^{\mathrm{area}}$ into area that has previously not been used as cropland. As pastureland and natural vegetation have a similar level of SOM (Eggleston et al., 2006), we can calculate the N_r inputs from SOM loss $\mathrm{N}_{t,i}^{\mathrm{som}}$ on the basis of land conversion for cropland, independent of whether the expansion occurs into natural vegetation or pastureland. After the conversion of cropland, we assume that cropland management releases 20 to 52 % of the original soil carbon, depending on the climatic region (Eggleston et al., 2006), plus the full litter carbon stock of the cell. Soil and litter carbon were estimated using the natural vegetation carbon pools of LPJml. N_r losses per hectare converted cropland $r_{t,j}^{\mathrm{som}}$ are then estimated on a cellular basis from the carbon losses, using a fixed C : N ratio of 15 for the conversion of forest or grassland to cropland. In reality, the soil carbon is released over a period of 20 yr until the carbon stock arrives in the new equilibrium (Eggleston et al., 2006). For simplification, we assume that all N_r

is released in the timestep of conversion (10 yr). To derive the yearly N_r release per ha $r^{\mathrm{som}}_{t,j}$, we divide N_r losses per hectare by 10 and assume no delayed release in the subsequent decade.

As MAgPIE is calibrated to the cropland area in 1995, no land conversion occurs in this timestep. To estimate $P(x_t)^{\mathrm{landconv}}_{t=1,j}$, we use the HYDE database with a 5 arcminutes resolution (Klein Goldewijk et al., 2011a). We define land conversion as the sum of (positive) cropland expansion in each geographic grid cell into land which was not used as cropland since the year 1900. In the case that cropland area first shrinks and then increases again, it is assumed that the same cropland area is taken into management that was abandoned before, so that no new SOM loss takes place. The high spatial resolution of Klein Goldewijk et al. (2011a) is of importance, because with higher aggregation (e.g. country-level estimates by FAOSTAT, 2011) expansion and contraction of cropland area within the same aggregation unit cancel out and land conversion is underestimated. The results for the historical estimates can be found in Table A7. The estimates for 1990–2000 are too high. The HYDE estimates are based on an older release of FAOSTAT data, while more recent FAOSTAT data corrected cropland expansion significantly downwards, reaching even a negative net expansion for the period 1990–2000 (Klein Goldewijk , 2011b). To estimate the contribution of N_r released by SOM loss to the N_r budget in 1995, we therefore only used the period 1980–1990.

A3.4 Losses and inorganic fertilizer

We calculate regional soil nitrogen uptake efficiency (SNUpE) $r^{\mathrm{SNUpE}}_{t=1,i}$ in 1995 by dividing total soil withdrawals $N(x_t)^{\mathrm{withd}}_{t=1,i}$ by total soil inputs $N(x_t)^{\mathrm{inp}}_{t=1,i}$.

$$r^{\mathrm{SNUpE}}_{t=1,i} = \frac{N(x_t)^{\mathrm{withd}}_{t=1,i}}{N(x_t)^{\mathrm{inp}}_{t=1,i}} \tag{A27}$$

The soil inputs include inorganic fertilizer, manure, N_r released from soil organic matter loss, recycled crop residues, atmospheric deposition and N_r fixation by free-living bacteria and algae. N_r in seed as well as N_r fixation by legumes and sugarcane are not counted as soil inputs, as they reach the plant not via the soil. Soil withdrawals are calculated by subtracting from the N_r in plant biomass (harvested organ, above- and belowground biomass) the amount of N_r that is not taken up from the soil and therefore not subject to losses prior to uptake. The latter includes again seed N_r as well as the N_r fixed from the atmosphere by legumes and sugarcane.

$$\begin{aligned} N(x_t)^{\mathrm{withd}}_{t,i} := \sum_v \Big(& (1 - r^{\mathrm{ndfa}}_v) \cdot (N(x_t)^{\mathrm{prod}}_{t,i,v} \\ & + N(x_t)^{\mathrm{prod\_ag}}_{t,i,v} + N(x_t)^{\mathrm{prod\_bg}}_{t,i,v}) \\ & - N(x_t)^{\mathrm{ds}}_{t,i,v,\mathrm{seed}} \Big) \end{aligned} \tag{A28}$$

$$\begin{aligned} N(x_t)^{\mathrm{inp}}_{t,i} := {} & N(x_t)^{\mathrm{fert}}_{t,i} + N(x_t)^{\mathrm{res}}_{t,i} + N(x_t)^{\mathrm{m\_cs}}_{t,i} \\ & + N^{\mathrm{som}}_{t,i} + N(x_t)^{\mathrm{dep}}_{t,i} + N(x_t)^{\mathrm{FixFree}}_{t,i} \end{aligned} \tag{A29}$$

The loss of N_r from cropland soils $N(x_t)^{\mathrm{loss}}_{t,i}$ is defined as the surplus of soil inputs over soil withdrawals.

$$N(x_t)^{\mathrm{loss}}_{t,i} := N(x_t)^{\mathrm{inp}}_{t,i} - \sum_v N(x_t)^{\mathrm{withd}}_{t,i} \tag{A30}$$

For the year 1995, we use historical data on regional fertilizer consumption based on (IFADATA, 2011) to estimate $r^{\mathrm{SNUpE}}_{t=1,i}$. In the following timesteps, $r^{\mathrm{SNUpE}}_{t,i}$ is fixed on an exogenous level (see Sect. A4), while the model balances out the regional budget by endogenously determining the amount of required inorganic fertilizer $N(x_t)^{\mathrm{fert}}_{t,i}$.

$$N(x_t)^{\mathrm{inp}}_{t,i} \geq \frac{N(x_t)^{\mathrm{withd}}_{t,i}}{r^{\mathrm{SNUpE}}_{t,i}} \tag{A31}$$

A3.5 Emissions

We distinguish into emissions from inorganic fertilizer ($N_2O(x_t)^{\mathrm{fert}}_{t,i}$), crop residues ($N_2O(x_t)^{\mathrm{res}}_{t,i}$), animal manure excreted or applied on cropland ($N_2O(x_t)^{\mathrm{m}}_{t,i}$), manure excreted on pasture range and paddock ($N_2O(x_t)^{\mathrm{past}}_{t,i}$), animal waste management ($N_2O(x_t)^{\mathrm{house}}_{t,i}$) and soil organic matter loss ($N_2O(x_t)^{\mathrm{som}}_{t,i}$). Each emission category has direct N_2O emissions plus eventually indirect emissions from volatilisation and leaching.

Direct N_2O emissions from soils are calculated as a fraction r^{dir} of the inputs from manure, fertilizer, crop residues and soil organic matter loss. According to Eggleston et al. (2006), paddy rice has lower direct emissions ($r^{\mathrm{dir\_rice}}$ instead of r^{dir}) from fertilization with inorganic fertilizers. As our methodology is unable to estimate the amount of inorganic fertilizer which is used specifically for rice production, we use EF_{1FR} for all N_r inputs of rice. The direct emission factor for emissions from N_r excreted during pasture range and paddock $r^{\mathrm{dir\_graz}}_l$ diverges between different animal types. For our livestock categories "ruminant meat" and "ruminant milk", containing animals of different types, we used weighted averages according to net excretion rates in 1995.

N_2O emissions from volatilisation occur when inorganic fertilizer or manure is applied to fields. The fraction volatilising in the form of NO_x or NH_y is different between the excretion or application of manure ($r^{\mathrm{gas\_m}}$), the application of inorganic fertilizer ($r^{\mathrm{gas\_fert}}$) and the management of animal

Table A7. Land conversion due to cropland expansion and release of N_r from subsequent soil organic matter (SOM) loss. For sources see text.

		Net expansion[a]	Land conversion[b]	SOM loss from land conversion			
		10^6 ha	10^6 ha	Tg C	Tg N_r	$\frac{kgN_r}{ha}$	$\frac{kgN_r}{ha \cdot yr}$ [c]
World	1960–1970	53	77	2574	172	2226	111
World	1970–1980	30	66	2464	164	2486	124
World	1980–1990	69	103	3754	250	2432	122
– AFR	1980–1990	13	17	529	35	2137	107
– CPA		33	25	848	57	2237	112
– EUR		−3	3	115	8	2885	144
– FSU		−2	9	542	36	4019	201
– LAM		8	12	489	33	2708	135
– MEA		5	4	48	3	738	37
– NAM		−1	13	614	41	3045	152
– PAO		4	5	108	7	1342	67
– PAS		10	10	359	24	2441	122
– SAS		2	5	103	7	1505	75
World	1990–2000[d]	22	325	12 370	825	2535	127

[a] Net expansion counts the aggregated change in regional or global cropland, and thus the difference of expansion and contraction.
[b] Land conversion sums up the expansion of each geographic grid cell into land which was not used as cropland since the year 1900. Contracting cropland is not substracted.
[c] Assuming that the soil organic matter is lost over 20 yr.
[d] Estimates for 1990–2000 are too high and should not be used (see text).

waste($r_{l,c}^{\text{gas\_awms}}$). A fraction $r^{\text{indir\_gas}}$ of these NO_x and NH_y gases transforms later on into N_2O.

Leaching is relevant for inorganic fertilizer application, residue management as well as the excretion or application of animal manure to agricultural soils. We assume, that a fraction r^{leach} of the applied N_r leaches into water bodies. According to Eggleston et al. (2006), r^{leach} is only relevant on croplands where runoff exceeds water holding capacity or where irrigation is employed, while for this model we made the simplification that leaching occurs everywhere. This assumption is also used in IPCC (1996). Of all N_r leaching into water bodies, a fraction $r^{\text{indir\_leach}}$ is assumed to transform later on into N_2O.

The following equations sum up the calculations according to the emission sources:

$$N_2O(x_t)_{t,i}^{\text{fert}} := N(x_t)_{t,i}^{\text{fert}} \cdot (r^{\text{dir}} + r^{\text{gas\_fert}} \cdot r^{\text{indir\_gas}} + r^{\text{leach}} \cdot r^{\text{indir\_leach}}) \quad \text{(A32)}$$

$$N_2O(x_t)_{t,i}^{\text{res}} := N(x_t)_{t,i}^{\text{res}} \cdot (r^{\text{dir}} + r^{\text{leach}} \cdot r^{\text{indir\_leach}}) \quad \text{(A33)}$$

$$N_2O(x_t)_{t,i}^{\text{m}} := N(x_t)_{t,i}^{\text{m}} \cdot (r^{\text{dir}} + r^{\text{gas\_m}} \cdot r^{\text{indir\_gas}} + r^{\text{leach}} \cdot r^{\text{indir\_leach}}) \quad \text{(A34)}$$

$$N_2O(x_t)_{t,i}^{\text{past}} := \sum_{l} (N(x_t)_{t,i,l,\text{grazp}}^{\text{ex}} + N(x_t)_{t,i,l,\text{grazc}}^{\text{ex}}) \cdot (r_l^{\text{dir\_graz}} + r^{\text{gas\_m}} \cdot r^{\text{indir\_gas}} + r^{\text{leach}} \cdot r^{\text{indir\_leach}}) \Big) \quad \text{(A35)}$$

$$N_2O(x_t)_{t,i}^{\text{house}} := \sum_{l,c} \Big(N(x_t)_{t,i,l,\text{house}}^{\text{ex}} \cdot r_{t,i,l,c}^{\text{cs}} \cdot (r_{l,c}^{\text{gas\_awms}} \cdot r^{\text{indir\_gas}} + r_c^{\text{dir\_house}}) \Big) \quad \text{(A36)}$$

$$N_2O(x_t)_{t,i}^{\text{som}} := N_{t,i}^{\text{som}} \cdot (r^{\text{dir}} + r^{\text{leach}} \cdot r^{\text{indir\_leach}}) \,. \quad \text{(A37)}$$

The NO_x and NH_y volatilisation on cropland area $N(x_t)_{t,i}^{\text{volat}}$, which is required for the calculation of atmospheric deposition in Eq. A23, is calculated as follows:

$$N(x_t)_{t,i}^{\text{volat}} := N(x_t)_{t,i}^{\text{fert}} \cdot r^{\text{gas\_fert}} + (N(x_t)_{t,i}^{\text{m}} + N(x_t)_{t,i,l,\text{grazp}}^{\text{ex}} + N(x_t)_{t,i,l,\text{grazc}}^{\text{ex}}) \cdot r^{\text{gas\_m}} + \sum_{l,c} (N(x_t)_{t,i,l,\text{house}}^{\text{ex}} \cdot r_{t,i,l,c}^{\text{cs}} \cdot r_{l,c}^{\text{gas\_awms}}) \,. \quad \text{(A38)}$$

The 2006 guidelines differ from the widely used 1996 guidelines (IPCC, 1996) most importantly in two aspects. Firstly, the N_r fixed by legumes and other N_r-fixing plants is not considered to have significant N_2O emissions. Only their comparably N_r-rich crop residues contribute to the N_2O emissions if they are left on the field. Secondly, the emission factor from leached N_r (EF_5, in our case $r^{\text{indir\_leach}}$) was lowered considerably from 2.5 % to 0.75 %.

To estimate the sensitivity of our results in regard to the uncertainty of the emission parameters, we carried out a Monte Carlo analysis with the software @Risk. We used a log-logistic probability density function (PDF) for the emission parameters r^{dir}, $r_c^{\text{dir\_house}}$, $r_l^{\text{dir\_graz}}$, $r^{\text{indir\_gas}}$, $r^{\text{indir\_leach}}$, r^{leach}, $r^{\text{gas\_fert}}$, $r^{\text{gas\_m}}$, and $r_{l,c}^{\text{gas\_awms}}$. We chose this PDF,

because it is non-negative, and because the median and the quantiles can be defined freely. We used the default value as mean and the uncertainty range from Eggleston et al. (2006) as 2.5 % and 97.5 % confidence intervals. We assumed that emission factors are non-correlated between each other. As the uncertainty range of the emission parameters in Eggleston et al. (2006) were estimated for country inventories, it is questionable whether they should be regarded as correlated between countries or not. We decided to regard the parameters as not correlated between regions, but as fully correlated for all countries within a region. As a consequence, regional uncertainties partly cancel out, and our global emission estimates have a lower relative uncertainty range. To simplify our calculation, we did not differentiate between waste management systems for animals kept in confinement, and simply assumed an error range of −50 % to +100 % for the aggregated mean of $r_c^{\text{dir\_house}}$ and $r_{l,c}^{\text{gas\_awms}}$.

We express the resulting uncertainty range for the emissions as a 90 % confidence interval, as the uncertainty distribution becomes very flat for higher significance levels.

A3.6 Food supply and intake

N_r in food supply is not equal to the N_r in harvested crops and slaughtered animals assigned for food, because the food products are processed. For food supply of crop products $N(x_t)^{\text{fs}}_{t,i,v}$, we therefore subtracted the N_r in conversion byproducts from the N_r in harvest assigned for food. Also, in the case of livestock products, the amount of N_r in the final products is not equal to the amount of N_r in the slaughtered animals, as only certain parts of the slaughtered animal are marketed, while the fifth quarter (often including head, feet, intestines and blood) is not used for food. Therefore, we calculated protein content per food product r_l^{PR} based on FAOSTAT (2011) and multiplied them with product specific protein–N_r ratios r_l^{NtoPR} from Sosulski and Imafidon (1990) and Heidelbaugh et al. (1975) to estimate the amount of N_r in livestock food supply ($N(x_t)^{\text{fs}}_{t,i,l}$).

Finally, the food supply is significantly higher than actual intake $N(x_t)^{\text{int}}_{t,i,k}$ because of significant waste rates on household level or in catering. We used regional intake to supply shares $r^{\text{int}}_{t,i,k}$ from Wirsenius (2000). As these shares will change with rising income, we estimated actual intake only for the year 1995.

$$N(x_t)^{\text{fs}}_{t,i,v} := N(x_t)^{\text{ds}}_{t,i,v,\text{food}} - N(x_t)^{\text{prod\_by}}_{t,i,v} \tag{A39}$$

$$N(x_t)^{\text{fs}}_{t,i,l} := N(x_t)^{\text{prod}}_{t,i,l} \cdot r_l^{\text{PR}} \cdot r_l^{\text{NtoPR}} \tag{A40}$$

$$N(x_t)^{\text{int}}_{t,i,k} := N(x_t)^{\text{fs}}_{t,i,k} \cdot r^{\text{int}}_{t,i,k} \tag{A41}$$

A4 Scenarios

For future projections, we created scenarios based on the SRES storylines (Nakicenovic et al., 2000). Quantitative interpretations of these storylines have been done by various integrated assessment models, whereof marker scenarios were selected. We use downscaled projections of population and per capita income of these marker scenarios as main drivers of the MAgPIE model (CIESIN, 2002a,b).

Bodirsky et al. (2012) create food demand scenarios for plant and livestock products based on the SRES population and GDP marker scenarios. To account for materialistic and non-materialistic lifestyles, they use different regressional forms for the A and B scenarios. In the A scenarios, they apply a log–log regression with a positive continuous time-trend for total caloric intake, and a multiple linear regression model for the livestock demand share. For the sustainable B scenarios, they use a log–log regression with positive declining time trend for total caloric intake, and an inverted u-shape regression model for livestock demand. In the latter, the share of animal products is increasing for low and medium incomes, but decreases for high incomes. The functional forms of the B scenarios tend to result in lower demand than the regression in the A scenarios. Yet, all four regressions are consistent with past observations (Table A8). The calculations are carried out on country level and are subsequently aggregated to the 10 MAgPIE regions. The scenarios are calibrated to meet the food demand in 1995 (FAOSTAT, 2011), the initial year of the MAgPIE model. Afterwards, they converge linearly towards the regression values throughout the 21st century to account for a globalisation of diets.

In all scenarios, the global food demand more than doubles from 1990 to 2070 (Fig. A2), while towards the end of the 21st century, the globalised scenarios A1 and B1 have a slightly declining food demand. Demand for livestock products (Fig. A3) is rising disproportionally strong, yet declines in all but the A2 scenario towards the end of the century.

The food demand projections are based on population and income growth of the SRES scenarios, starting in 1990. As can be seen in figure A2 and A3, the historical data of food demand is met more or less precisely depending on the scenario. Global food calorie demand diverges in 2005 by 98 PJ (+0.4 %) (B1) to 452 PJ (1.7 %) (B1), while meat demand diverges by −244 PJ (−5.2 %) (A2) to +60 PJ (1.2 %) (B2). The largest differences can be observed in the estimates for meat demand in CPA, where the A2 scenario diverges by −422 PJ (−31.5 %) while the B2 scenario almost matches the observed data with 15 PJ (+1.1 %). Large parts of these variations in estimates are determined by the uncertainty of the original SRES projections for population and GDP.

A parameter which is subject to large uncertainty is the development of future trade liberalisation policies. For 1995, we fix the share of domestic demand settled by imported products at their actual level in 1995. For the subsequent timesteps, we assume that an increasing share can be traded according to comparative advantages in production costs. The share of products traded according to historical trade patterns decreases in turn by 10 % per decade in the two globalised scenarios A1 and B1. These scenarios are equivalent to the policy scenario of Schmitz et al. (2012), extended

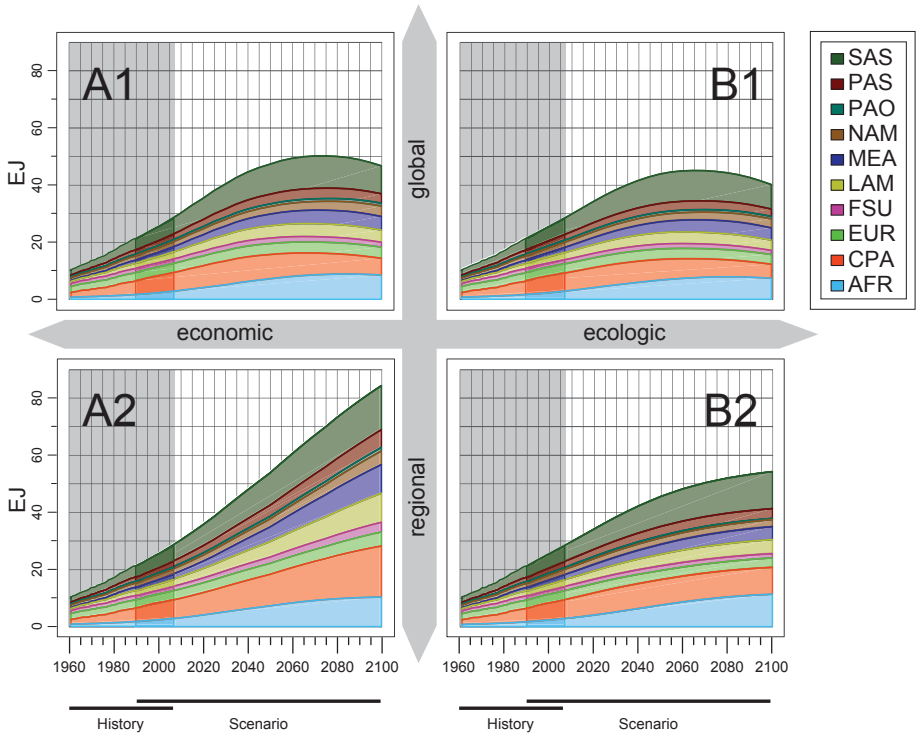

Fig. A2. Total food energy demand in the 10 MAgPIE world regions. History and future developments for the four SRES scenarios (Bodirsky et al., 2012).

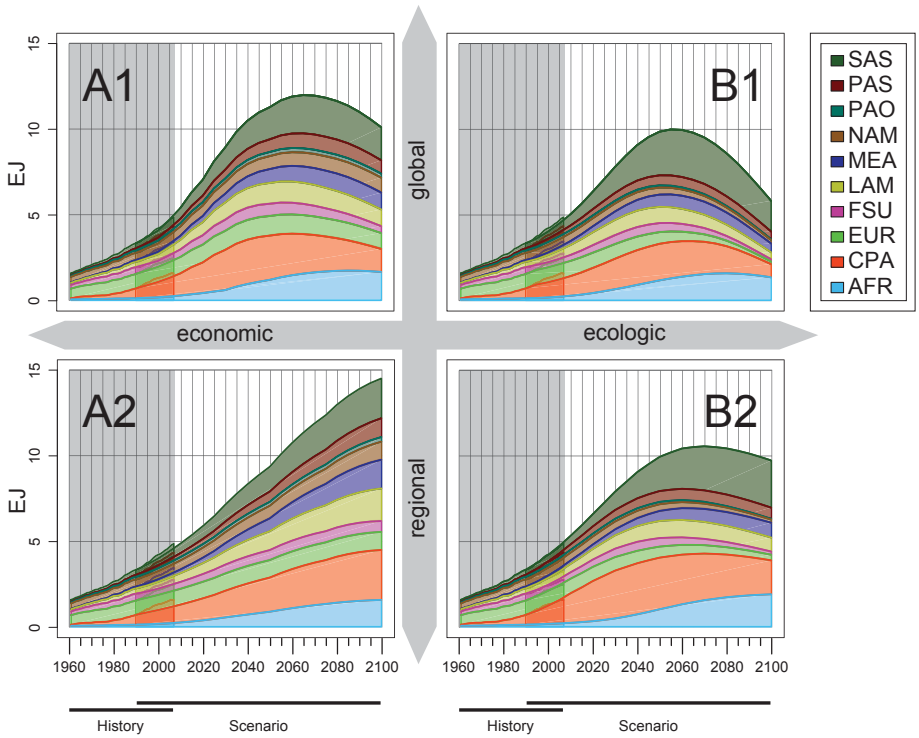

Fig. A3. Demand for energy from livestock products in the 10 MAgPIE world regions. History and future developments for the four SRES scenarios (Bodirsky et al., 2012).

Table A8. Regression models for total calories C_T in kcal and the share of livestock calories in total demand C_S, depending on income I in 2005 US Dollar in market exchange rate.

SRES	Model	Formulae	Parameter	Slope	r^2	p-value	F-statistics
A	Calories	$C_T = a \cdot (I)^b$	$a = \exp(2.825 + 2.131 \times 10^{-3} \cdot \text{year})$, $b = 0.162 - 3.124 \times 10^{-5} \cdot \text{year}$	0.658	0.65	<0.001 (***)	11060
	Livestock share	$C_S = \exp(k + l \cdot \ln(I) + m \cdot \text{year} + n \cdot \ln(I) \cdot \text{year})$	$k = -36.733, l = 4.497$, $m = 0.016, n = -0.002$	0.705	0.63	<0.001 (***)	9913
B	Calories	$C_T = a \cdot (I)^b$	$a = 933.89 + \frac{387.47 \cdot (\text{year}-1960)}{\text{year}-1960+9.77}$, $b = 0.0894 + \frac{0.008445 \cdot (\text{year}-1960)}{\text{year}-1960-0.75569}$	0.678	0.64	<0.001 (***)	10551
	Livestock share	$C_S = p \cdot \sqrt{I} \cdot \exp(-q \cdot I)$	$p = 0.00932 - 3.087 \times 10^{-6} \cdot \text{year}$, $q = -2.654 \times 10^{-4} + 1.420 \times 10^{-7}$	0.706	0.62	<0.001 (***)	9685

to 2095. For the regionalised scenarios, we assume a slower rate of market integration with a reduction of only 2.5 % per decade.

The efficiency of nutrient uptake on croplands is a parameter which has strong impact on the results of the model. While we estimate this parameter for the base year 1995, its development into the future is rather uncertain. Policies like the nitrate directive in Europe seemed to have a large impact in the past (Oenema et al., 2011), so the environmental awareness seems to be a key driver of N_r efficiency. To differentiate the economically orientated from the environmentally orientated scenarios, we adjust the cropland nutrient uptake efficiency $r_{t,i}^{\text{SNUpE}}$ for future scenarios. The starting points for $r_{t=1,i}^{\text{SNUpE}}$ are calculated endogenously in the model, and converge linearly over n timesteps to their scenario values $r_{n,i}^{\text{SNUpE}}$ (Table 1).

$$r_{t,i}^{\text{SNUpE}} := (1 - \frac{t}{n}) \cdot r_{t=1,i}^{\text{SNUpE}} + \frac{t}{n} \cdot r_{n,i}^{\text{SNUpE}} \quad \text{(A42)}$$

We chose to have high efficiency values in the B scenario due to high awareness for local environmental damages. The most efficient agricultural systems currently absorb around 70 % of applied N (Smil, 1999), and Vuuren et al. (2011) estimate that "in practice, recovery rates of 60–70 % seem to be the maximum achievable". So we adopted this value for the environmentally oriented B scenarios. In the A1 scenario, we assumed that $r_{t,i}^{\text{SNUpE}}$ increases due to widespread use of efficient technologies (e.g. precision farming), which saves costs but also resources. Yet, no improvements beyond cost efficiency are made, thus $r_{t,i}^{\text{SNUpE}}$ stays behind the B scenarios towards the end of the century. Finally, the A2 scenario stagnates slightly above the current mean, and only improves towards the end of the century.

A further scenario parameter is the development of livestock production systems. Feed baskets and livestock productivity diverge significantly in different world regions, with some systems being more industrialised and consuming mainly feedstock crops, others being pastoral or mixed systems. While the development of the livestock system is highly uncertain, a trend towards industrialised systems can be observed (Delgado, 1999). For future scenarios, we converge the feed baskets and livestock productivity linearly towards the European livestock system, a system with rather low share of pastoral and traditional systems and a high share of industrialised livestock production. We assume a fast convergence in the globalised systems A1 and B1, while the regional scenarios keep more of their current regional feed mixes (Table 1). To implement this into the model, we converged the parameters $r_{t,i,l,v}^{\text{fb\_conc}}$, $r_{t,i,l}^{\text{fb\_past}}$, $r_{t,i,l,v}^{\text{fb\_ag}}$, $r_{t,i,l,v}^{\text{fb\_by}}$ and $r_{t,i,l,f}^{\text{fs}}$ similar to Eq. (A42) to the European values in 1995. To account for an increasing modernization of the agricultural sector, the same type of convergence is applied to $r_{t,i}^{\text{msplit}}$ and $r_{t,i,l}^{\text{fuel}}$ and the fractions of byproducts and crop residues burned or used for other purposes.

Even more uncertain is the development of the animal waste management. Even for the present, little information exists on the differences of animal waste management around the world, and there is no clear pattern as to which of the systems is dominating with increasing modernization. Similarly, we assumed that manure management for housed animals is changing over time. For the economically orientated scenarios and the B1 scenario, we assumed that bioenergy plants using anaerobic digesters increase in importance, while the B scenarios also have an increasing share of manure being directly brought back on fields as daily spread. The convergence towards these systems is higher in globalised scenarios, while the current regional animal waste management mix partly prevails in the A2 and B2 scenarios. In the model, we implemented the convergence for the parameter $r_{t,i,l,c}^{\text{cs}}$ similar to Eq. (A42).

Supplementary material related to this article is available online at: http://www.biogeosciences.net/9/4169/2012/bg-9-4169-2012-supplement.zip.

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Chapter V

On sustainability of bioenergy production: Integrating co-emissions from agricultural intensification

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Content of chapter V: On sustainability of bioenergy production

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On sustainability of bioenergy production: Integrating co-emissions from agricultural intensification

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ABSTRACT

Biomass from cellulosic bioenergy crops is seen as a substantial part of future energy systems, especially if climate policy aims at stabilizing CO_2 concentration at low levels. However, among other concerns of sustainability, the large-scale use of bioenergy is controversial because it is hypothesized to increase the competition for land and therefore raise N_2O emissions from agricultural soils due to intensification. We apply a global land-use model that is suited to assess agricultural non-CO_2 GHG emissions. First, we describe how fertilization of cellulosic bioenergy crops and associated N_2O emissions are implemented in the land-use model and how future bioenergy demand is derived by an energy-economy-climate model. We then assess regional N_2O emissions from the soil due to large-scale bioenergy application, the expansion of cropland and the importance of technological change for dedicated bioenergy crops. Finally, we compare simulated N_2O emissions from the agricultural sector with CO_2 emissions from the energy sector to investigate the real contribution of bioenergy for low stabilization scenarios.

As a result, we find that N_2O emissions due to energy crop production are a minor factor. Nevertheless, these co-emissions can be significant for the option of removing CO_2 from the atmosphere (by combining bioenergy use with carbon capture and storage (CCS) options) possibly needed at the end of the century for climate mitigation. Furthermore, our assessment shows that bioenergy crops will occupy large shares of available cropland and will require high rates of technological change at additional costs.

1. Introduction

Climate Change represents an unprecedented challenge to global society. Unmitigated climate change will introduce large-scale risks to ecosystems and human societies, while its mitigation could represent a major task for the world economic system. If greenhouse gas emissions from different sectors are not mitigated climate change can lead to an increase of global mean temperature of up to 5 °C or more relative to pre-industrial levels [1], implying large-scale shifts in global and regional climates, ecosystem patterns, and human activities. Current research [2] suggests that extreme weather events such as droughts, heat waves or tropical cyclones occur more frequently than assumed in the Third Assessment Report (TAR) by the IPCC in 2001 and ambitious climate protection is needed to sustain e.g. ecosystems such as tropical coral reefs. Ambitious climate protection strategies aiming at low stabilization levels of greenhouse gas concentrations can be roughly categorized as policies constraining atmospheric GHG concentrations to 450 ppm CO_2-equivalent

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or below. The target of the European Union and of the eight major economies G8 [3] of limiting the rise in global mean temperature to 2 °C above pre-industrial levels falls into this category. A large-scale transformation of the global energy systems will be needed to achieve the deep emission reductions required to avoid dangerous climate change.

In the future, the agricultural sector is seen to contribute much to climate change mitigation as growing energy demand, security of energy supply and concerns about climate change will make bioenergy crops more attractive for a future sustainable energy system. Biomass can be used to provide energy in many forms including heat, electricity, gaseous, solid and liquid fuels. A special point of interest is that scenarios aiming at low GHG concentration targets may require options within the energy sector to remove CO_2 from the atmosphere ("negative emissions"). This implies that human-induced uptake of CO_2 is larger than the amount of human-induced GHG emissions. One of the few technologies that results in negative emissions is the combination of bioenergy and carbon capture and storage (CCS) in case bioenergy is assumed to be carbon-neutral.

Recent estimates on the potential global bioenergy supply range from less than 100 EJ y^{-1} to over 400 EJ y^{-1} for 2050 [4]. The potential for bioenergy supply can be divided into 3 main categories: traditional bioenergy, agricultural and forest residues and bioenergy from dedicated energy crops. These estimates differ due to large discrepancies in land availability for biomass plantations and yield levels (including future yield improvements) in crop production. However, the large-scale use of bioenergy is controversial because it could affect food security, biodiversity and water availability. Furthermore, as dedicated bioenergy carriers will increase the competition for land they have been hypothesized to raise N_2O emissions from soils due to intensification on currently used agricultural land [5–8].

In order to assess additional emissions in the agricultural sector due to large-scale application of dedicated energy crops in a spatially explicit way, we apply a Model of Agricultural Production and its Impact on the Environment (MAgPIE) [9–11]. MAgPIE is a mathematical programming model covering the most important food and feed, livestock and bioenergy production types in 10 economic regions worldwide. It takes regional economic conditions as well as spatially explicit data on potential crop yields, land and water constraints from a global process-based vegetation and hydrology model (LPJmL) [12] into account and derives specific land-use patterns and associated non-CO_2 emissions for each grid cell.

In this article, we first describe the global land-use optimization model MAgPIE, how fertilization of bioenergy crops and associated N_2O emissions are implemented and how future bioenergy demand is derived by the energy-economy-climate model REMIND-R [13]. We focus on GHG emissions from the application of cellulosic bioenergy crops such as Miscanthus or poplar. The reason is that cellulosic bioenergy crops are seen to have the best environmental and economic performance compared to other crops and are also versatile, i.e. the biomass can be converted into various liquid and solid types of bioenergy carriers. Secondly, we apply MAgPIE to assess regional N_2O emissions from the soil due to large-scale bioenergy application and present maps that display simulated land-use patterns for bioenergy in 2055, i.e. the share of a grid cells area used for bioenergy production. We then compare simulated N_2O emissions from the agricultural sector with CO_2 emissions from the energy sector to investigate the real contribution of bioenergy for low stabilization scenarios if co-emissions from bioenergy like N_2O are taken into account. Finally, we discuss our results and conclude with the identification of our main achievements and recommendations based on our simulation results.

2. Methods

2.1. MAgPIE

MAgPIE [9–11] is a global land-use allocation model, which is coupled to the grid-based Lund–Potsdam–Jena dynamic global vegetation model with managed Lands (LPJmL) [12,14]. Hereby, it takes regional economic conditions as well as spatially explicit data on potential crop yields, land and water constraints into account and derives specific land-use patterns, yields and total costs of agricultural production for each grid cell. The information flow in our coupled modeling approach is shown in Fig. 1. Since implementation and validation of MAgPIE is presented in detail elsewhere [9–11], only a short overview will be provided here.

The objective function of the land-use model is to minimize total cost of production for a given amount of agricultural demand. Regional food energy demand is defined for an exogenously given population and income growth in ten food energy categories (cereals, rice, vegetable oils, pulses, roots and tubers, sugar, ruminant meat, non-ruminant meat, and milk), based on regional diets [15].

Food and feed energy for the ten demand categories can be produced by 20 cropping activities (temperate cereals for food or feed, maize for food or feed, tropical cereals for food or feed, rice, five oil crops, pulses, potatoes, cassava, sugar beets, sugar cane, vegetables/fruits/nuts, two fodder crops) and three livestock activities (ruminant meat, non-ruminant meat, milk). Feed for livestock is produced as a mixture of grain, green fodder, and pasture at fixed proportions. Fiber demand is currently fulfilled with one cropping activity (cotton). Cropland, pasture and irrigation water are fixed inputs in limited supply in each grid cell, measured in physical units of hectares and cubic meters. Variable inputs of production are labor, chemicals, and other capital (all measured in US$ (in 1995 purchasing power parity terms)), which are assumed to be in unlimited supply to the agricultural sector at a given price. Moreover, the model can endogenously decide to acquire yield-increasing technological change at additional costs, if otherwise there is no feasible solution under a given set of resource constraints.

For future projections the model works on a time step of 10 years in a recursive dynamic mode. The link between two consecutive periods is established through the land-use pattern. For the base year 1995, total agricultural land is constrained to the area currently used within each grid cell, according to [16]. The optimized land-use pattern from one period is taken as the initial land constraint in the next.

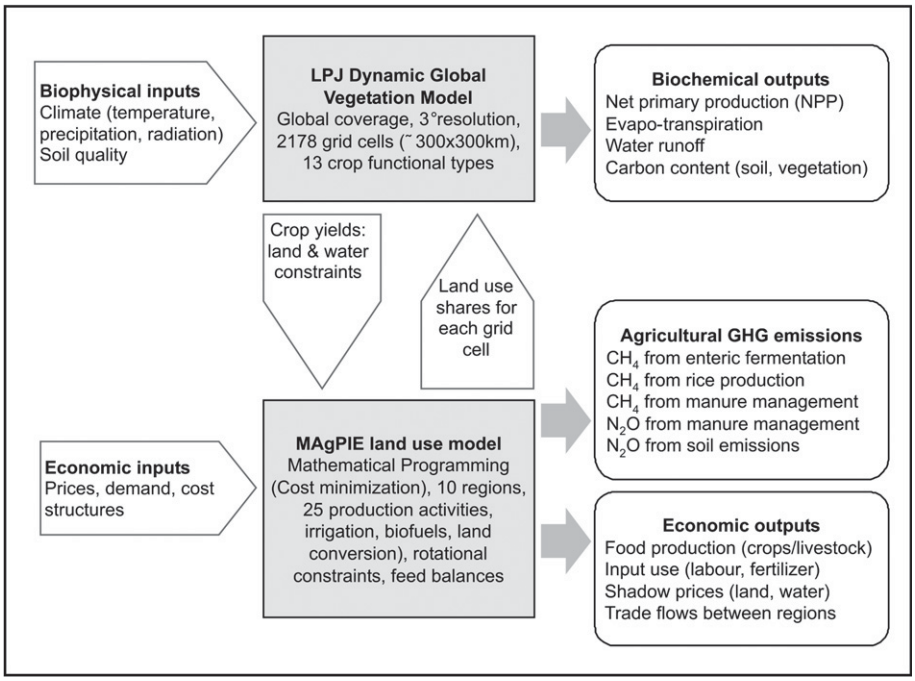

Fig. 1 – Information flow within the modelling system.

Optionally, additional land from the non-agricultural area can be converted into cropland at additional costs. Trade in food products between regions is simulated endogenously, constrained by minimum self-sufficiency ratios for each region. But trade of biomass for bioenergy is not allowed.

Potential crop yields for each grid cell are supplied by the global vegetation model LPJmL [12,14]. In addition to major food and feed crops, also cellulose-based bioenergy crops such as Miscanthus and Poplar have been implemented [17]. LPJmL endogenously models the dynamic processes linking climate and soil conditions, water availability and plant growth, and takes the impacts of CO_2, temperature and radiation on yield directly into account. Spatially explicit data on yield levels and freshwater availability for irrigation is provided to MAgPIE on a regular geographic grid, with a resolution of three by three degrees, dividing the terrestrial land area into 2178 discrete grid cells of an approximate size of 300 km by 300 km at the equator. Towards higher latitudes the grid cells become smaller. Each cell of the geographic grid is assigned to one of ten economic world regions: Sub-Saharan Africa (AFR), Centrally-planned Asia including China (CPA), Europe including Turkey (EUR), the Newly Independent States of the Former Soviet Union (FSU), Latin America (LAM), Middle East/North Africa (MEA), North America (NAM), Pacific OECD including Japan, Australia, New Zealand (PAO), Pacific (or Southeast) Asia (PAS), and South Asia including India (SAS). The regions are initially characterized by data for the year 1995 on population [18], gross domestic product (GDP) [19], food energy demand [15], average production costs for different production activities [20], and current self-sufficiency ratios for food [15].

MAgPIE incorporates a representation of the dominant greenhouse gas (GHG) emissions from different agricultural activities [11]. Here, we focus on N_2O emissions from the soil. As agricultural emissions arise from multiple causes, they depend on the type of agricultural activity. Their extent is heavily influenced by crop or animal type, fertilizer input, climate, soil quality or farm management.

2.1.1. *N_2O emissions from bioenergy*

Nitrous oxide (N_2O) is produced naturally in soils through the microbial process of denitrification and nitrification. Anthropogenic activities such as fertilization, production of nitrogen-fixing crops add nitrogen to the soil, thereby increasing the amount of nitrogen available for nitrification and denitrification.

We calculate N_2O emissions from agricultural soils ($N2O_{SOIL}$) in time t within a grid cell c by including direct ($N2O_D$) as well as indirect emissions ($N2O_I$):

$$N2O_{SOIL_{t,c}} = N2O_{D_{t,c}} + N2O_{I_{t,c}} \quad (1)$$

2.1.2. *Direct N_2O emissions from agricultural soils*

We estimate direct N_2O emissions ($N2O_D$) due to large-scale application of cellulosic bioenergy from the N input by synthetic fertilizer (NS), N-fixing crops (NF) and crop residues (NR):

$$N2O_{D_{t,c}} = (NS_{t,c} + NF_{t,c} + NR_{t,c}) * e_f \quad (2)$$

For the emission factor e_f we apply the default value 12.5 g N_2O–N Kg^{-1} of nitrogen [21].

2.1.2.1. Calculation of total nitrogen input by synthetic fertilizer. We calculated the application of nitrogen fertilizers based on the assumption that nitrogen withdrawn from the agricultural system due to the harvest of biomass has to be replaced by synthetic fertilizers to allow for permanent production. We set the nitrogen content (N-content) of herbaceous biomass such as Miscanthus to 0.44% of dry biomass at harvest time [22]. For woody bioenergy crops,

which consist of fast-growing plants like poplar and eucalyptus we chose 0.52% N-content [23].

2.1.2.2. Calculation of total nitrogen input by N-fixing crops. Nitrogen input from N-fixing crops (NF) within cell c at time t is calculated from dry yield production (P) of all N-fixing crops (F) within this cell multiplied by the fraction of nitrogen (n_F) in this respective crop:

$$NF_{t,c} = \sum_{P_F} 2 * P_{t,c} * n_F \quad (3)$$

2.1.2.3. Calculation of total nitrogen input from crop residues. Nitrogen input from crop residues (NR) is calculated as:

$$NR_{t,c} = 2 * \left(\sum_{P_F} P_{F_{t,c}} * n_F + \sum_{P_{NF}} P_{NF_{t,c}} * n_{NF} \right) * (1 - f_r) * (1 - f_b) \quad (4)$$

where P_F and P_{NF} describe the total annual biomass production of N-fixing and non-N-fixing crops and n_F and n_{NF} the respective fraction of nitrogen in these crops. f_r stands for the fraction of crop residues that is removed from the field as crop and f_b the fraction of crop residue that is burned. The factor 2 converts edible crop production to total crop biomass.

2.1.3. *Indirect N_2O emissions from agricultural soils*

Indirect N_2O emissions ($N2O_I$) enter the atmosphere by one of two pathways: (i) atmospheric deposition of NOx and NH_3 (originating from fertilizer use and livestock excretion of nitrogen), and (ii) leaching and run-off of nitrogen from fertilizer applied to agricultural fields and from livestock excretion.

$$N2O_{I_{t,c}} = N2O_{V_{t,c}} + N2O_{L_{t,c}} \quad (5)$$

2.1.3.1. Volatilization of NH_3 and NOx during fertilization. Only a specific fraction of applied Nitrogen from synthetic fertilizer (NS) and manure (NM) turns into NH_3 and NOx. The emission factor e_v of 10% is the share of NH_3 and NOx turning into N_2O due to volatilization [21]:

$$N2O_{V_{t,c}} = \left(\left(NS_{t,c} * f_{V_S} \right) + \left(NM_{t,c} * f_{V_M} \right) \right) * e_v \quad (6)$$

2.1.3.2. Leaching of N during fertilization. 30% of both synthetic fertilizer (NS) and manure (NM) applied is lost to leaching and surface run-off (f_l) and 2.5% of this lost N is emitted as N_2O–N (e_l) [21]:

$$N2O_{L_{t,c}} = (NS_{t,c} + NM_{t,c}) * f_l * e_l \quad (7)$$

All grid cell specific emission factors are consistent with the Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories [21] and the IPCC Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories [24]. All IPCC national parameters, livestock and crop types are aggregated to the MAgPIE regions, animal (ruminant meat, non-ruminant meat and milk) and crop production types.

2.2. *Bioenergy demand*

Future bioenergy demand is derived by the energy-economy-climate model REMIND-R [13]. REMIND-R is a multi-regional hybrid model which couples an economic growth model with a detailed energy system model and a simple climate model. REMIND-R maximizes global inter-temporal welfare subject to various constraints. The solution is equivalent to equilibria on all markets [25]. A distinguished pareto-optimal solution for international trading between regions, which in the absence of externalities also corresponds to the market equilibrium, is obtained using the Negishi algorithm. The applied version of REMIND-R simulates the economic dynamics over the time horizon 2005–2100 for 11 world regions: USA, EU27, Japan, China, India, Russia, Sub-Saharan Africa, Middle East and North Africa, Other Asia, Latin America and Rest of the World. Macro-economic output, i.e. gross domestic product (GDP), in each region is determined by a macro-economic production function that combines labor, capital and various final energy carriers like electricity or transportation fuels.

The production of the final energy carriers is undertaken by the energy conversion sector that represents various routes for converting primary energy carriers into final energy carriers at the techno-economic level. In the following we concentrate on the role of supply and conversion of biomass in the energy sector. The lingo-cellulosic biomass is subject to a biomass supply function which relates the biomass supply price to the quantity of biomass. Biomass as well as all derived energy carriers are non-tradable goods. Cellulosic biomass is a primary energy carrier that can be converted into all types of final energy. Most notably the REMIND model comprises combined heat power plants, heat plants, synthetic natural gas, solid biomass, hydrogen and Fischer–Tropsch plants. The latter two alternatives can also be equipped with CCS, which leads to negative net emissions at the plant level, since biomass production is assumed to be carbon-neutral. These two technologies are essential for achieving a low carbon economy because (i) they produce non-electric final energy carriers that would be difficult to decarbonize and (ii) the net negative emissions allow higher positive emissions to be maintained from the use of oil and natural gas. Having available a maximum biomass potential of 200 EJ also allows the economy to achieve net negative CO_2 emissions from the overall energy sector as will be pointed out in detail below.

In the business as usual (BAU) scenario, we simulate a development as if climate change has no economically and socially important effects. The global primary energy consumption increases from around 475 EJ in 2005 to more than 1100 EJ in 2100 and the emissions from 27 Gt CO_2 to around 75 Gt CO_2. Within the baseline scenario we assumed a potential of biomass usable for bioenergy production of around 200 EJ. We run three policy scenarios (2 °C) that all aim at the same climate stabilization level, but differ with respect to the assumption of the available biomass potential – 100 EJ, 200 EJ and 400 EJ. The policy scenarios take climate policies, directed to reduce climate change impacts, into account. This is implemented by binding the temperature increase to 2 °C relative to the pre-industrial level. The policy scenarios include an international emissions trading system. Fig. 2 compares the baseline emission trajectory with that of the 2 °C low stabilization scenario. Substantial emission reductions and a final emission level below 0 Gt CO_2 are needed to keep the 2 °C target. We now focus on the demand for bioenergy within the low stabilization scenario. The importance

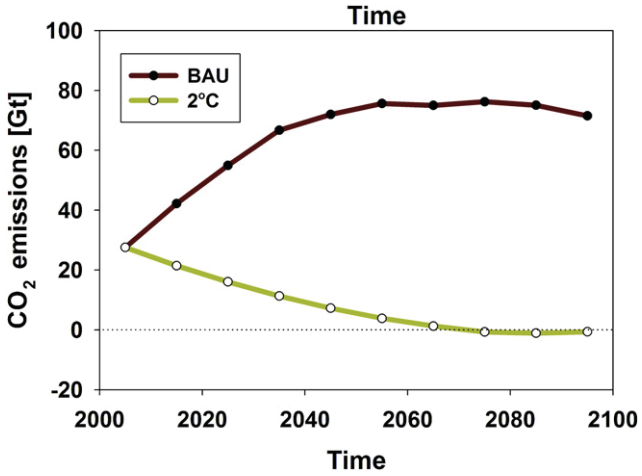

Fig. 2 – GHG emissions from the energy and transport system. The brown line stands for the business as usual (BAU) and the yellow line for the policy scenario (2 °C). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

of bioenergy as key component in a cost-efficient energy mix is demonstrated by Fig. 3. Even a potential of 400 EJ is fully used in the long run. The regional shares on bioenergy indicate that in particular Russia, Latin America and Africa could benefit from the use of the biomass option.

2.3. *Simulation analysis*

In a first step we use REMIND-R to simulate different scenarios of bioenergy demand. We then applied the land-use model MAgPIE to assess the additional N_2O emissions from bioenergy production in the agricultural sector. Compared to MAgPIE, REMIND-R holds one additional world region (Rest Of the World – ROW) that comprises major exporters of fossil fuels such as Australia, Canada or South Africa. Therefore, we allocated the bioenergy demand of ROW equivalently to the world regions of MAgPIE the respective countries belong to.

For each scenario, we run the MAgPIE model in six 10-year time steps from 2005 until 2095 in a recursive dynamic manner to calculate the demand for nitrogen fertilizers based on the average nutrient content (N-content) of the harvested biomass.

Table 1 – Regional share of bioenergy cropland in total cropland (2055).

	Share of bioenergy cropland in total cropland [%]									
	AFR	CPA	EUR	FSU	LAM	MEA	NAM	PAO	PAS	SAS
100 EJ	12	8	26	41	15	9	11	31	11	2
200 EJ	21	17	45	48	29	13	26	56	22	7
400 EJ	26	31	52	59	43	31	36	71	36	9

We applied two scenarios for the efficiency of N-Application, i.e. the share of N applied that is used by the crop: (i) current regional efficiency (current management scenario) and (ii) optimal efficiency (best practice scenario) of N-Application. Current N-efficiency equals 50% with regional differences and optimal efficiency is reached when 70% of the N applied is taken up by the plant [26]. N_2O emissions from agricultural soils are estimated by including direct as well as indirect emissions. We then compare CO_2 reduction in the energy sector for climate change mitigation due to the application of biomass technologies with N_2O emissions from the agricultural sector to assess the real contribution of bioenergy to low stabilization scenarios. Finally, we investigate the expansion of cropland and the importance of technological change for dedicated bioenergy crops until 2055 and present maps that display simulated land-use patterns for bioenergy, i.e. the share of a grid cells area used for bioenergy production in 2055.

In line with international greenhouse accounting practice [1], GHG emissions are expressed as carbon dioxide equivalents. All emissions in the form of N_2O-N are translated to N_2O emissions by multiplying with the molecular weight ratio 44/28. CO_2 and N_2O emissions were converted and summed together to CO_2 equivalents (CO_2-e) using the 'global warming potential' (GWP), which determines the relative contribution of a gas to the greenhouse effect. The GWP (with a time span of 100 years) of CO_2 and N_2O is 1 and 298, respectively (1).

3. Results

3.1. *Land-use patterns and technological change for bioenergy 2nd generation in 2055*

The share of cropland used for bioenergy in total cropland in 2055 increases in all regions with increasing technological

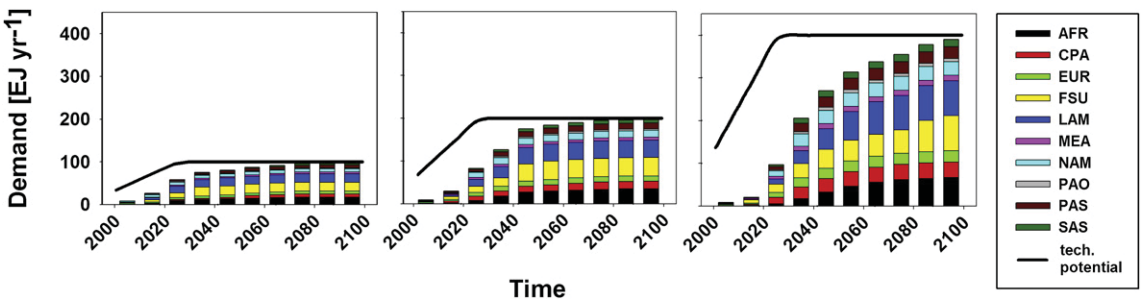

Fig. 3 – Regional bioenergy demand for AFR (black), CPA (red), EUR (light green), FSU (yellow), LAM (dark blue), MEA (pink), NAM (light blue), PAO (grey), PAS (brown) and SAS (dark green). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

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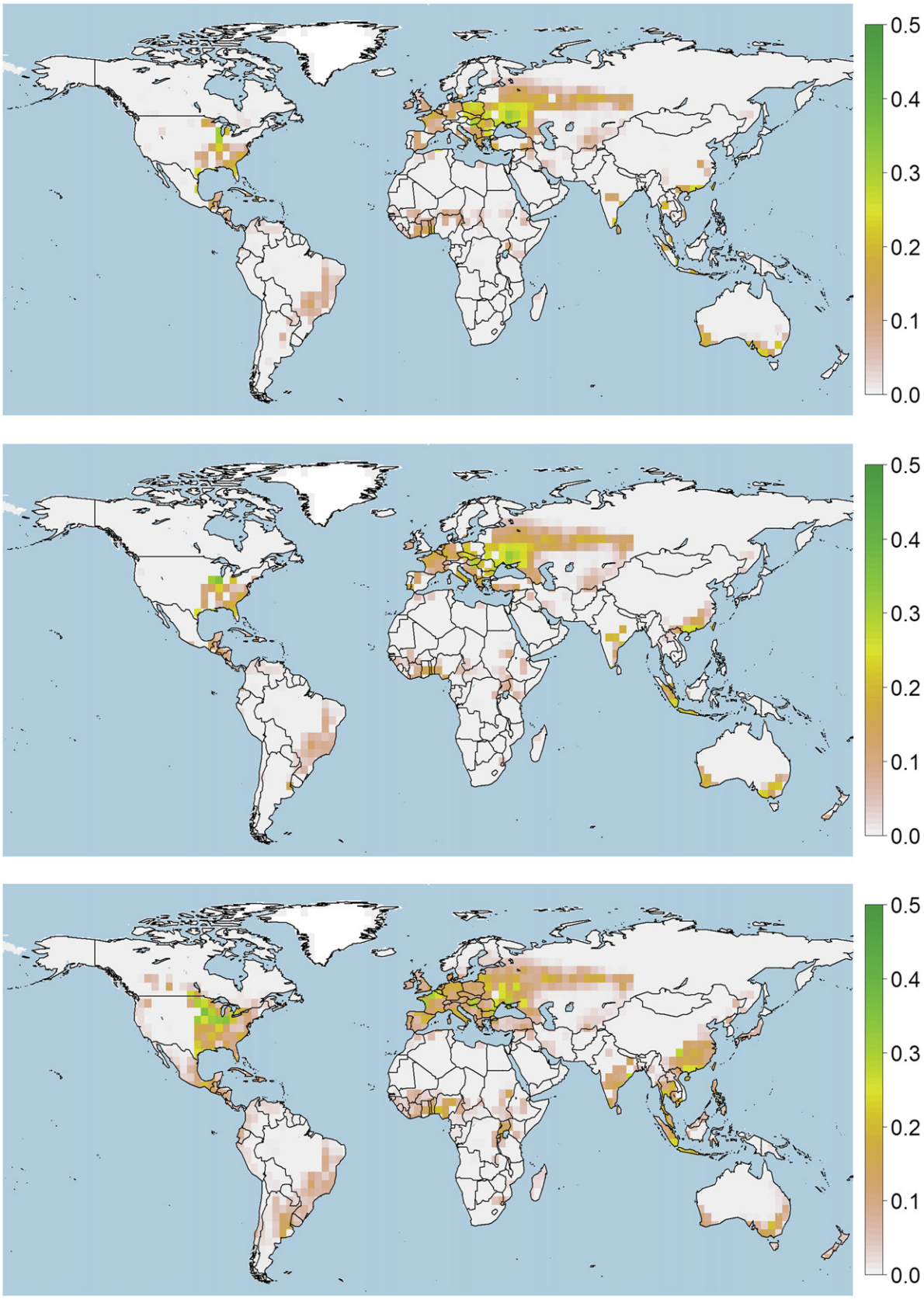

Fig. 4 – Land-use patterns for bioenergy in 2055, i.e. the share of a grid cells area used for bioenergy production, for the 100 EJ (upper row), 200 EJ (middle row) and 400 EJ available biomass potential scenarios (lower row).

Table 2 – Average regional yield increases [%] per year for all crops over the time period 1995–2055.

	100 EJ	200 EJ	400 EJ
AFR	2.3	3.1	2.9
CPA	1.0	2.3	2.3
EUR	0.3	1.7	2.4
FSU	2.3	3.5	4.2
LAM	1.6	2.8	2.9
MEA	4.3	5.3	5.9
NAM	0.9	2.1	2.2
PAO	0.7	1.6	2.6
PAS	1.2	2.3	2.3
SAS	1.5	2.4	2.4

bioenergy potential (Table 1). Highest shares of 71% occur in PAO in the 400 EJ scenario, whereas lowest shares of 2% can be observed in SAS for the 100 EJ scenario. Globally, the share of cropland used for bioenergy in total cropland equals 16% in 2055 in the scenario with technological potential of 100 EJ and increases to 28% and 39% in the scenarios with technological potential of 200 EJ and 400 EJ. Fig. 4 shows land-use patterns for bioenergy, i.e. the share of a grid cells area used for bioenergy production, in 2055 simulated by MAgPIE for all scenarios of technical bioenergy potential.

The resulting rates of technological change for the baseline and the three bioenergy scenarios are provided in Table 2. The numbers describe average regional yield increases per year for all crops over a given period (1995–2055 for the future scenarios). The rate of technological change increases significantly with increasing bioenergy demand. Highest yield increases can be found in MEA and FSU with up to 6 and 4 percent per year in the 400 EJ scenario.

3.2. *N-application*

Simulated N-application under current regional N-efficiency (current management scenario) equals, averaged over the period of 95 years, 13.2 Mt N for the scenario with technological potential of 100 EJ, 25.7 Mt N for the scenario with technological potential of 200 EJ and 45.5 Mt N for the scenario with technological potential of 400 EJ (Fig. 5). N-applications in the year 2100 rise to 17.6 Mt N in the scenario with technological potential of 100 EJ, 36.5 Mt N in the scenario with technological potential of 200 EJ and 70.6 Mt N in the scenario

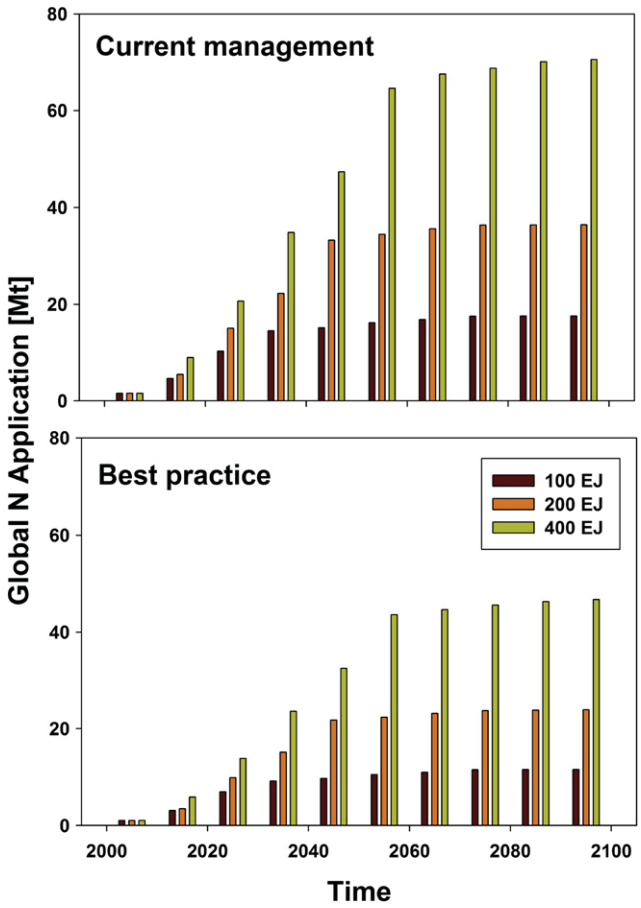

Fig. 5 – Global N-application in Mt per year for current management (upper row) and best practice (lower row) in the 100 EJ (brown), 200 EJ (orange) and 400 EJ (yellow) scenarios. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

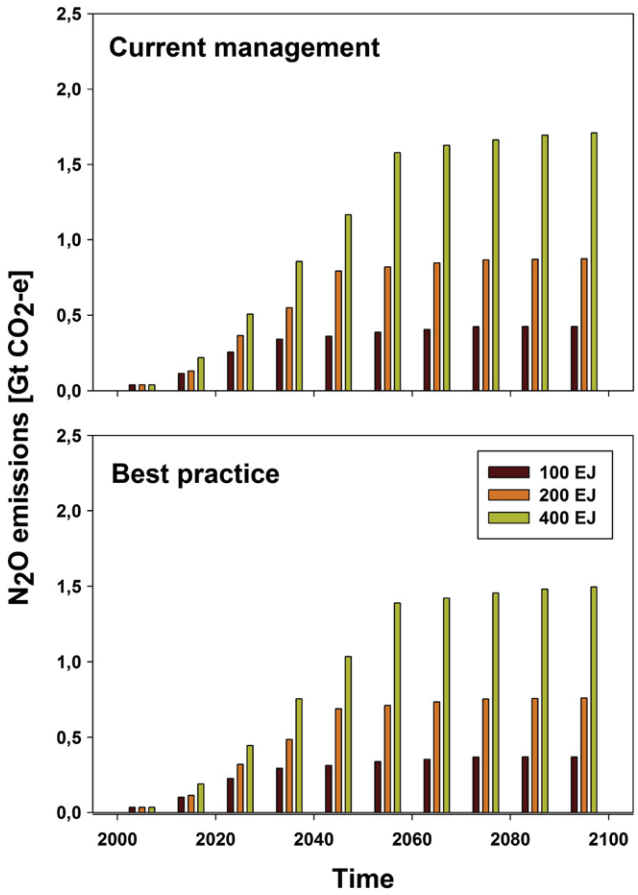

Fig. 6 – Global N_2O emissions in Gt CO_2-e per year for current management (upper row) and best practice (lower row) in the 100 EJ (brown), 200 EJ (orange) and 400 EJ (yellow) scenarios. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

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with technological potential of 400 EJ. Under optimal N-efficiency (best practice scenario) mean N-application over the period of 95 years decreases by about 33% in all scenarios of technological biomass availability, compared to the scenario with current regional N-efficiencies.

3.3. N_2O emissions

In the current management scenario with regional N-efficiency, N_2O emissions amount to the maximum of 1.71 Gt CO_2-e for the scenario with technological potential of 400 EJ in the years 2100 and decrease to 0.87 Gt CO_2-e (technological potential of 200 EJ) and 0.42 Gt CO_2-e (technological potential of 100 EJ) (Fig. 6).

Lower N_2O emissions can be found in the best practice scenario with optimal N-efficiency. Here, mean N_2O emissions over the period of 95 years decrease by about 12% in all scenarios of technological biomass availability, compared to the scenario with current regional N-efficiencies.

3.4. Net greenhouse gas emissions of bioenergy

Until 2095, CO_2 emission reduction from bioenergy contributes 20.4 Gt CO_2 (100 EJ scenario), 36.1 Gt CO_2 (200 EJ scenario) and 52.2 Gt CO_2 (400 EJ scenario) for climate change mitigation. The real contribution of bioenergy to climate change mitigation, i.e. CO_2 emissions reductions from the energy sector due to bioenergy that are not nullified by N_2O emissions from the agricultural sector due to N fertilization of bioenergy crops, increases over time for all scenarios of technological potential. Lowest values of 83% can be found in 2015 in the 400 EJ scenario and highest values of 99.5% in 2095 in the 100 EJ scenario. Until 2095, 4.93% (100 EJ), 6.04% (200 EJ) and 8.07% (400 EJ) of CO_2 emissions reductions from the energy sector due to bioenergy are nullified by N_2O emissions from the agricultural sector due to N fertilization of bioenergy crops (Fig. 7).

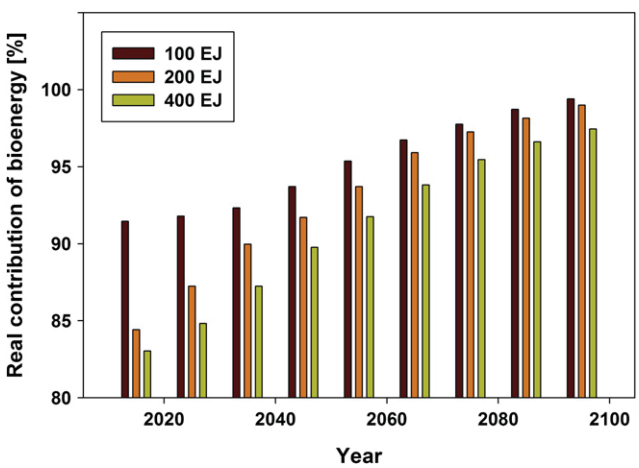

Fig. 7 – The real contribution of bioenergy for low stabilization scenarios if N_2O emissions from bioenergy are taken into account in the 100 EJ (brown), 200 EJ (orange) and 400 EJ (yellow) scenarios. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Fig. 8 shows that N_2O induced by the application of synthetic fertilizers due to bioenergy crop production can have considerable impacts on emissions, as e.g in the 200 EJ and especially in the 400 EJ scenarios former negative GHG emissions convert to positive emissions.

4. Discussion

The main objective of this paper is to discuss the influence of considering co-emissions from bioenergy use for climate change mitigation. Bioenergy is seen as a substantial part of future energy systems, especially if climate policy aims at stabilizing CO_2 concentration at low levels to avoid dangerous climate change. However, among other concerns about the sustainability of bioenergy the potentially unfavorable net greenhouse gas balances have been raised, e.g. [5,6,27]: Like other crops, bioenergy crops require nitrogen inputs for their production as nitrogen fertilizers restock the nutrients removed from the soil by harvest and increase biomass in nutrient poor soils. But nitrogen input inevitably leads to emission of N_2O, which is a byproduct of soil nitrification and denitrification. In order to assess the importance of co-emissions from large-scale cultivation of cellulosic bioenergy crops for very low GHG concentration we estimated future demand for fertilizers and associated N_2O emissions resulting from bioenergy crop production up to 2095. In accordance with other studies that consider N_2O emissions as the most important GHG due to a global biofuels program in the long run [28], our simulation results reveal that additional N_2O emissions rise strongly over time. The impact of biomass application in the energy system for climate change mitigation is slightly but significantly affected by N_2O emissions from the agricultural sector due to N fertilization of bioenergy crops. The real contribution of

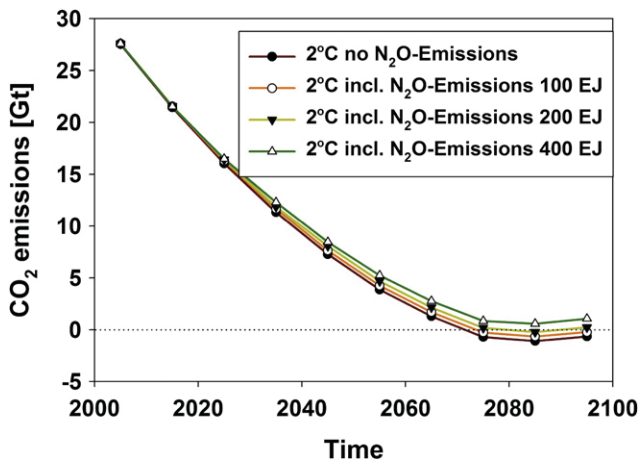

Fig. 8 – Emission trajectory for the low stabilization target without N_2O from dedicated bioenergy crops (brown line), including N_2O emissions in the 100 EJ (orange line), 200 EJ (yellow line) and 400 EJ (green line) scenarios. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

bioenergy to low stabilization scenarios, i.e. CO_2 emissions reductions from the energy sector due to bioenergy that are not nullified by N_2O emissions from the agricultural sector due to N fertilization of bioenergy crops, increases over time. This is mainly due to the fact that, in contrast to 2095, the application of biomass in the energy system is not combined with CCS in the early years. However, in all scenarios CCS does not have the capability to constrain the N_2O emissions due to the fertilization of dedicated bioenergy crops. With respect to the importance of CCS for the contribution of bioenergy to climate change mitigation, one needs to consider that the technology is still uncertain and not proven on a large scale. However, we found that N_2O induced by the application of N in synthetic fertilizers due to bioenergy crop production can have considerable impacts on low stabilization targets in the energy systems. Depending on the amount of bioenergy used in the energy system, the consideration of N_2O in the bioenergy use turned former negative GHG emissions into positive emissions, implying that the contribution of bioenergy use so far suggested by energy system models, e.g. in [29] is probably overestimated.

But nitrogen applied is not always used efficiently by crops [30,31]. Our analysis showed that improving the efficiency of nitrogen application can reduce emissions of N_2O, generated by soil microbes largely from surplus N, but even under optimal efficiency of N-application negative emissions cannot be maintained. Additionally, optimized N-application will indirectly reduce emissions of CO_2 from N fertilizer manufacture. A number of practices that reduce N_2O emissions from the soil exist already today: adjusted application rates based on precise estimation of crop needs (e.g. precision farming), application of slow-release fertilizers or nitrification inhibitors, prevention of time delays between N-application and plant N uptake, placing the N more precisely into the soil to make it more accessible to crops roots, application of denitrification inhibitors or eliminating N-applications where possible, see [32].

Industrial N fixation has increased exponentially from near zero in the 1940s and may play an important role in agriculture required to feed the growing world population with food and bioenergy [33]. In 2005, industrial fixation of N for use as fertilizer totaled approximately 90 Mt N [34], nearly the same amount that would be needed to fulfill the high demand for bioenergy feedstock (400 EJ) in 2100 under constant efficiency of nitrogen application on current levels.

In this study we aimed to assess N_2O emissions from the agricultural sector but large-scale cultivation of cellulosic bioenergy crops is also indirectly responsible for emissions in other sectors. First, we did not consider CO_2 emissions from changes in above- or belowground biomass, soil organic matter, and litter due to the conversion of land. Among all land types, the amount of carbon stored in annual crops for food and feed production is the lowest. Hence, historical land-use change such as deforestation in tropical regions that is mainly driven by agricultural production contributed to about 3.8 Pg of carbon per year from 1990 to 1999 [35,36]. These impacts are likely to be similar for perennial cellulosic bioenergy crops in the case that land with high standing biomass, soil carbon or thick layers of peat [37] is converted into energy plantations or crops and pasture are displaced from current agricultural lands indirectly causing GHG emissions via conversion of native habitat to cropland elsewhere [27].

The net effect of bioenergy production would look different if degraded, abandoned agricultural lands or other land with low carbon content could be used for the cultivation of cellulosic bioenergy crops [38], which could even lead to net carbon uptake as a consequence of root as well as shoot litter. Furthermore, besides soil nitrous oxide emissions, CO_2 emissions associated with agricultural inputs such as the production of fertilizers and farm equipment operation are major sources of GHG fluxes associated with crop production [39,40]. Further enhancement in agricultural productivity needed for fulfilling the future demand for feed- and foodstock will require higher levels of fertilizer application. Today, the production of fertilizers is energy intensive, and contributed between 300 and 600 Mt CO_2-e per year, representing between 0.6 and 1.2% of the world's total GHGs [41].

Large-scale bioenergy production could lead to other conflicts beyond the aspect of GHG balances. Our assessment showed that bioenergy crops will occupy large shares of available cropland and will require high rates of technological change at additional costs. But as land, water, and other inputs are only available in limited amounts, they have been hypothesized to raise the price of food over the long term. Furthermore, concerns about the sustainability of bioenergy are growing, including water use and biodiversity issues. Worldwide, agriculture accounts for roughly 70 percent of global freshwater use, but in the future a growing share will be needed for industrial and household use. As demand for biomass for fuel continues to rise, so too will the pressures on global freshwater supplies for irrigation. Finally, large-scale bioenergy production will have negative consequences for biodiversity. Expansion of agricultural lands for biofuels into natural areas will decrease availability of habitats suitable for many species and reduce the ecosystem services offered by more complex ecological systems [42].

Negative emissions possibly needed in the end of the century for ambitious climate mitigation will probably be more difficult to achieve as proposed by energy system models as so far GHG emissions from bioenergy crop cultivation are not considered in most of the models. In a model comparison of energy-economy models that do not include GHG emissions from land-use mitigation costs more than double with a restriction of the biomass potential in a low stabilization scenario [29]. In addition, they pointed out that the biomass potential stipulated in each case also affects the resulting energy mix: In scenarios with low biomass potential the amount of other renewable sources increases also because restrictions placed on the biomass potential speed development and introduction of the other renewables. Integrated models that take energy, economy, climate and land use into account will have to be developed to learn more about sustainable energy mixes for mitigation of climate change.

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Chapter VI

Food consumption, diet shifts and non-CO2 greenhouse gases from agricultural production

Alexander Popp • Hermann Lotze-Campen • Benjamin Leon Bodirsky

Content of chapter VI: On sustainability of bioenergy production

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Food consumption, diet shifts and associated non-CO_2 greenhouse gases from agricultural production

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ABSTRACT

Today, the agricultural sector accounts for approximately 15% of total global anthropogenic emissions, mainly methane and nitrous oxide. Projecting the future development of agricultural non-CO_2 greenhouse gas (GHG) emissions is important to assess their impacts on the climate system but poses many problems as future demand of agricultural products is highly uncertain. We developed a global land use model (MAgPIE) that is suited to assess future anthropogenic agricultural non-CO_2 GHG emissions from various agricultural activities by combining socio-economic information on population, income, food demand, and production costs with spatially explicit environmental data on potential crop yields. In this article we describe how agricultural non-CO_2 GHG emissions are implemented within MAgPIE and compare our simulation results with other studies. Furthermore, we apply the model up to 2055 to assess the impact of future changes in food consumption and diet shifts, but also of technological mitigation options on agricultural non-CO_2 GHG emissions. As a result, we found that global agricultural non-CO_2 emissions increase significantly until 2055 if food energy consumption and diet preferences remain constant at the level of 1995. Non-CO_2 GHG emissions will rise even more if increasing food energy consumption and changing dietary preferences towards higher value foods, like meat and milk, with increasing income are taken into account. In contrast, under a scenario of reduced meat consumption, non-CO_2 GHG emissions would decrease even compared to 1995. Technological mitigation options in the agricultural sector have also the capability of decreasing non-CO_2 GHG emissions significantly. However, these technological mitigation options are not as effective as changes in food consumption. Highest reduction potentials will be achieved by a combination of both approaches.

1. Introduction

Currently, agriculture accounts for approximately 14% of total global anthropogenic greenhouse gas emissions (5.1–6.1 Gt CO_2-eq/yr in 2005) and is responsible for about 47% and 58% of total anthropogenic emissions of methane (CH_4) and nitrous oxide (N_2O) (IPCC, 2007). CH_4 contributes about 3.3 Gt CO_2-eq/yr and N_2O 2.8 Gt CO_2-eq/yr, whereas the contribution of CO_2 emissions is negligible (US-EPA, 2006; IPCC, 2007; Stern, 2006). CH_4 is produced by anaerobic decomposition of organic matter, mainly associated with enteric fermentation of ruminants. Other sources of CH_4 in agriculture are rice cultivation and manure storage. N_2O emissions from agriculture are basically related to nitrogen fertilizers and manure applied to soils, but also manure storage.

In the past, global agricultural CH_4 and N_2O emissions have increased by nearly 17% from 1990 to 2005, an average annual emission increase of about 60 Mt CO_2-eq/yr (IPCC, 2007). But, the magnitude and relative importance of the different sources and emissions vary widely between regions as developing countries account for the major share (almost three quarters) of agricultural emissions (Stern, 2006).

As a result of ongoing population growth, rising per capita caloric intake and changing dietary preferences, such as an increased consumption of meat and dairy products, demand for agricultural products in the future is anticipated to increase significantly, especially in Asia, Latin America, and Africa. Global land use and associated GHG emissions will change in the future, reacting to these pressures. Increases in consumption have to be met by production gains from increased land use intensity or agricultural expansion. At the international level, shifting certain parts of agricultural production to more productive countries and increasing international trade flows may be an economically efficient way to increase overall productivity (Lotze-Campen et al., in press). While the combined impacts of all these interplays are still highly uncertain, GHG from the agricultural sector will be affected. Projecting their future development is important to assess their impacts on the climate system as well as mitigation potentials and options from the agricultural sector.

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Many studies have been undertaken to assess current and future agricultural GHG emissions. However, the conclusions from these studies differ significantly. The main reason for the differences is that the most crucial drivers for agricultural GHG emissions – supply and demand – are highly uncertain. Resulting global GHG emissions from agriculture range for example from 6700 Mt CO_2-equ (US-EPA, 2006) to 10,150 Mt CO_2-equ (Strengers et al., 2004) for 2020 and differ greatly in geographical location and share of the different agricultural activities.

Moreover, most of the approaches for the estimation of past, current and future GHG emissions from agriculture activities focus on selected single world regions and agricultural activities (e.g. Yamaji et al., 2004; Oenema et al., 2005; Herrero et al., 2008). But agricultural production activities and associated GHG emissions affect each other. For example, CH_4 emissions from manure storage and enteric fermentation due to increased livestock production are complemented by N_2O emissions from the soil due to additional feedstock production (Thorpe, 2009).

In order to simulate these combined effects in a spatially explicit way, we present a Model of Agricultural Production and its Impact on the Environment (MAgPIE) (Lotze-Campen et al., 2008, in press). MAgPIE is a mathematical programming model covering the most important food and feed as well as livestock production types in 10 economic regions worldwide. It takes regional economic conditions as well as spatially explicit data on potential crop yields, land and water constraints from a global process-based vegetation and hydrology model (LPJmL) (Bondeau et al., 2007) into account and derives specific land use patterns. We modified MAgPIE in such a way that each spatially explicit agricultural activity is associated with non-CO_2 emissions.

In this article, we first describe the land use allocation model MAgPIE and how anthropogenic agricultural non-CO_2 GHG emissions are implemented within the modeling system. Secondly, we show regional projections for agricultural non-CO_2 GHG emissions for the business-as-usual scenario and present maps for agricultural CH_4 and N_2O emissions in 1995 and 2055. We then apply the model to assess the impact of future changes in food consumption and contrast the outcome with simulation results that consider technological mitigation options of non-CO_2 GHG emissions in the agricultural sector.

In the next section, we compare our simulation results on agricultural non-CO_2 GHG emissions with other studies. We finally discuss our results and conclude with the identification of our main achievements and recommendations based on our simulation results.

2. Methods

2.1. Model description

MAgPIE (Lotze-Campen et al., 2008, in press) is a global land use allocation model, which is coupled to a grid-based dynamic vegetation model (LPJmL) (Sitch et al., 2003; Bondeau et al., 2007). Hereby, it takes regional economic conditions as well as spatially explicit data on potential crop yields, land and water constraints into account and derives specific land use patterns, yields and total costs of agricultural production for each grid cell. The information flow in our coupled modeling approach is shown in Fig. 1. Since implementation and validation of MAgPIE is presented in detail elsewhere (Lotze-Campen et al., 2008, in press), only a short overview will be provided here.

The non-linear objective function of the land use model is to minimize total cost of production for a given amount of agricultural demand. Regional food energy demand is defined for an exogenously given population and income growth in ten food energy categories (cereals, rice, vegetable oils, pulses, roots

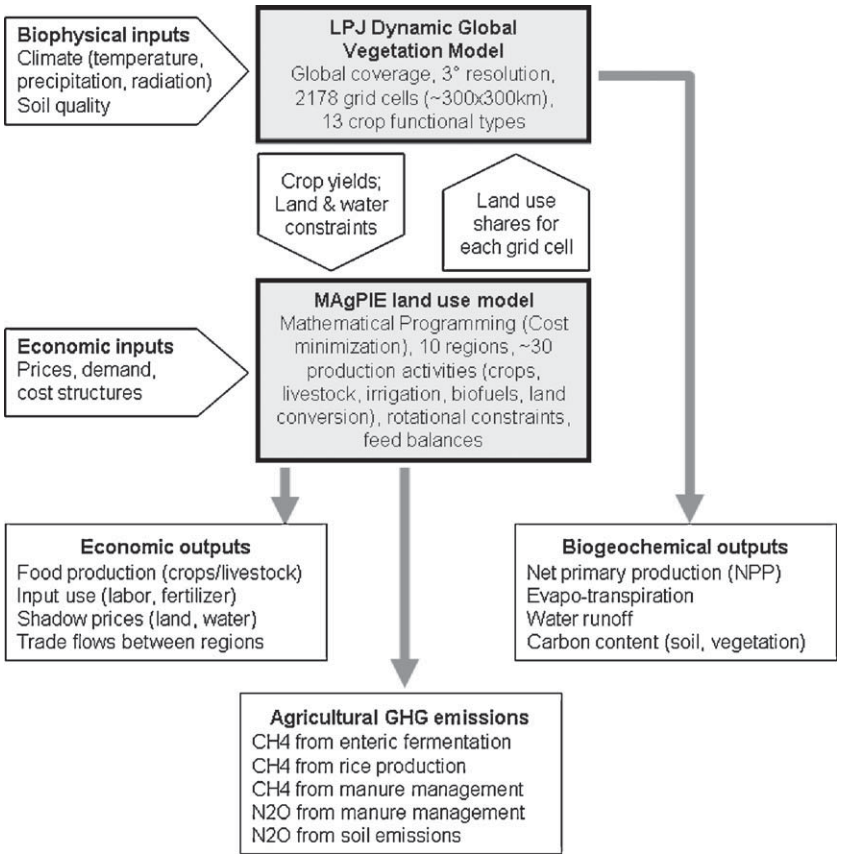

Fig. 1. Information flow within the modeling system.

and tubers, sugar, ruminant meat, non-ruminant meat, and milk), based on regional diets (FAOSTAT, 2008).

Food and feed energy for the ten demand categories can be produced by 20 cropping activities (temperate cereals for food or feed, maize for food or feed, tropical cereals for food or feed, rice, five oil crops, pulses, potatoes, cassava, sugar beets, sugar cane, vegetables/fruits/nuts, two fodder crops) and 3 livestock activities (ruminant meat, non-ruminant meat, milk). Feed for livestock is produced as a mixture of grain, green fodder, and pasture at fixed proportions. Fiber demand is currently fulfilled with one cropping activity (cotton). Cropland, pasture and irrigation water are fixed inputs in limited supply in each grid cell, measured in physical units of hectares (ha) and cubic meters (m^3). Variable inputs of production are labor, chemicals, and other capital (all measured in US$), which are assumed to be in unlimited supply to the agricultural sector at a given price. Moreover, the model can endogenously decide to acquire yield-increasing technological change at additional costs, if otherwise there is no feasible solution (i.e. land use pattern) under a given set of resource constraints.

For future projections the model works on a time step of 10 years in a recursive dynamic mode. The link between two consecutive periods is established through the land use pattern. For the base year 1995, total agricultural land is constrained to the area currently used within each grid cell, according to Ramankutty and Foley (1999). The optimized land use pattern from one period is taken as the initial land constraint in the next. Optionally, additional land from the non-agricultural area can be converted into cropland at additional costs. MAgPIE as an optimization model and based on its current parameterization tends to specialize and concentrate production of single crop types in certain areas. We have introduced upper limits to specific crop shares in specific locations, in line with technological constraints observed in average crop rotations. Trade in food products between regions is simulated endogenously, constrained by minimum self-sufficiency ratios for each region.

Crop yields for each grid cell are supplied by the Lund–Potsdam–Jena dynamic global vegetation model with managed Lands (LPJmL) (Sitch et al., 2003; Bondeau et al., 2007). LPJmL endogenously models potential crop yields by linking climate and soil conditions, water availability, and plant growth, and takes the impacts of CO_2, temperature and radiation directly into account. However, the model does not include soil degradation and management factors (plant breeding, fertilizer and pesticides, machinery) apart from irrigation into the calculation of potential yield levels. Actual yields are derived from potential ones through a 'management factor' based on FAO statistics. LPJmL also covers the full hydrological cycle on a global scale, which is especially useful as carbon and water-related processes are closely linked in plant physiology (Gerten et al., 2004; Rost et al., 2008). The suitability of LPJmL for crop and water studies has been demonstrated by validating simulated phenology and yields (Bondeau et al., 2007; Fader et al., 2010), river discharge (Biemans et al., 2009), evapotranspiration (Sitch et al., 2003; Gerten et al., 2004) and irrigation water requirements (Rost et al., 2008).

Crop yields for MAgPIE are computed as a weighted average of irrigated and non-irrigated production, if part of the grid cell is equipped for irrigation according to the global map of irrigated areas (Döll and Siebert, 2000). In case of pure rain-fed production, no additional water is required, but yields are generally lower than under irrigation. If a certain area share is irrigated, additional water for agriculture is taken from available water discharge in the grid cell. Water discharge is computed as the runoff generated under natural vegetation within the grid cells and its downstream movement according to the river routing scheme implemented in LPJmL.

Spatially explicit data on yield levels and freshwater availability for irrigation is provided to MAgPIE on a regular geographic grid, with a resolution of three by three degrees, dividing the terrestrial land area into 2178 discrete grid cells of an approximate size of 300 km by 300 km at the equator. Towards higher latitudes the grid cells become smaller. Each cell of the geographic grid is assigned to one of ten economic world regions (Fig. 2): Sub-Saharan Africa (AFR), Centrally-planned Asia including China (CPA), Europe including Turkey (EUR), the Newly Independent States of the Former Soviet Union (FSU), Latin America (LAM), Middle East/North Africa (MEA), North America (NAM), Pacific OECD including Japan, Australia, New Zealand (PAO), Pacific (or Southeast) Asia (PAS), and South Asia including India (SAS). The regions are initially characterized by data for the year 1995 on population (CIESIN et al., 2000), gross domestic product (GDP) (World Bank, 2001), food energy demand (FAOSTAT, 2008), average production costs for different production activities (McDougall et al., 1998), and current self-sufficiency ratios for food (FAOSTAT, 2008). While all supply-side activities in the model are grid-cell specific, the demand side is aggregated at the regional level. There are 12 demand categories, compared to 23 cropping and livestock activities. Certain demand types can thus be satisfied with different crop types, which allows for substitution between supply categories. Aggregate demand within each region, defined by total population, average income and net trade, is being met by the sum of production from all grid cells within the region.

Trade in food products between regions is simulated endogenously, constrained by minimum self-sufficiency ratios for each region. This is to say that some minimum level of domestic demand has to be produced within the region, while the rest can be allocated to other regions according to comparative advantages.

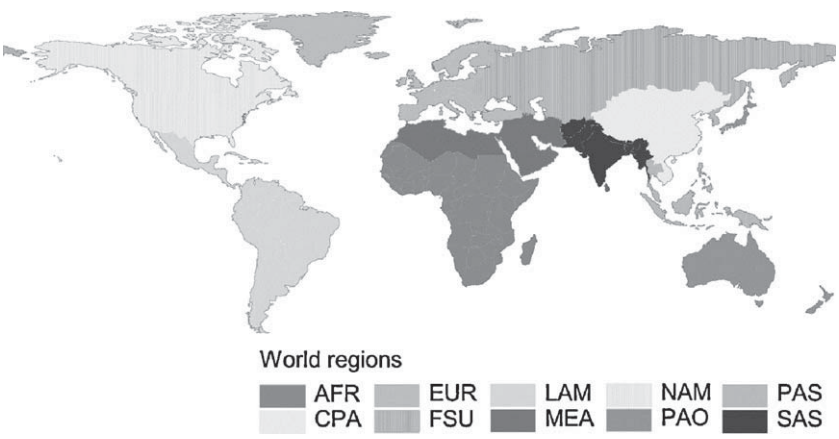

Fig. 2. Model regions.

The version of MAgPIE presented here incorporates a representation of the dominant greenhouse gas (GHG) emissions from different agricultural activities. We focus on N_2O emissions from the soil and manure storage as well as CH_4 emissions from rice cultivation, enteric fermentation and manure storage that add up to 87% of total agricultural (land use) emissions in the year 2000 (US-EPA, 2006). As agricultural emissions arise from multiple causes, they depend on the type of agricultural activity. Their extent is heavily influenced by crop or animal type, fertilizer input, climate, soil quality or farm management. In the following we give an overview of the simulated agricultural emissions, a detailed description is attached as Appendix A.

2.2. Overview of simulated agricultural emissions

We calculate anthropogenic *N_2O emissions from agricultural soils* by including direct as well as indirect emissions. In our approach, direct N_2O emissions are affected by nitrogen input due to synthetic fertilizers, crop residues, N-fixing crops, and manure application. Indirect N_2O emissions enter the atmosphere by one of two pathways: (1) atmospheric deposition of NOx and NH3 (originating from fertilizer use and livestock excretion of nitrogen), and (2) leaching and runoff of nitrogen from fertilizer applied to agricultural fields and from livestock excretion.

Anthropogenic N_2O *from animal waste management systems (AWMSs)* is produced by the nitrification and denitrification of the organic nitrogen content in livestock manure and urine. In our modeling approach N_2O emissions from AWMS are affected by the amount of livestock products, livestock product specific nitrogen excretion and specific AWMS for animal products.

Anthropogenic *CH_4 emissions from AWMSs* are produced during the anaerobic decomposition of manure. In our model, CH_4 emissions from AWMS are influenced by livestock species and temperature. We furthermore differentiate between developed and developing countries.

Anthropogenic CH_4 *emissions from enteric fermentation* occur when microbes in an animal's digestive system ferment food. Methane is produced as a byproduct and is exhaled by the animal. Emissions factors are specific for the model regions as the share of domesticated ruminants such as cattle, buffalo, sheep, goats, and camels differs.

The anaerobic decomposition of organic matter in flooded rice fields produces CH_4. We model anthropogenic *CH_4 emissions from rice cultivation* to depend on water management practices and regional specific emission factors.

All grid cell specific emission factors are consistent with the Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 1997) and the IPCC Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories (IPCC, 2000). All IPCC national parameters, livestock and crop types are aggregated to the MAgPIE regions, animal (ruminant meat, non-ruminant meat and milk) and crop production types. In line with international greenhouse accounting practice (IPCC, 2007), emission factors are expressed as carbon dioxide equivalents. CO_2, N_2O and CH_4 emissions were converted and summed together to CO_2 equivalents (CO_2-e) using the 'global warming potential' (GWP), which determines the relative contribution of a gas to the greenhouse effect. The GWP (with a time span of 100 years) of CO_2, CH_4 and N_2O is 1, 25 and 298, respectively (IPCC, 2007).

2.3. Scenario analysis

Agricultural GHG emissions will depend on future lifestyles, i.e. the demand for agricultural products, affected by population growth, per capita caloric intake and changing dietary preferences, such as an increased consumption of meat and dairy products.

First, we apply the MAgPIE model under different scenarios to assess the global impact of future changes in food consumption on agricultural non-CO_2 GHG emissions.

In the '*baseline scenario*' a conversion rate of initially non-agricultural land to cropland was chosen to display the historical cropland expansion (1995–2005) of about 3.5% (FAO, 2008) per decade. Regional trade balances, food energy demand and the share of livestock products in total caloric intake are kept constant at 1995 levels. In the *'Increased meat scenario'* the link between GDP and food energy demand as well as the share of livestock products in total caloric intake are given by a regression equation (Fig. 3). In the *'Decreased meat scenario'* food energy increases with increasing GDP comparable to the *'Increased meat scenario'* but the share of livestock products in total caloric intake is reduced in each time step, starting in 2005, by 25% compared to the previous time step.

Additionally, we implemented scenarios ('*Increased meat scenario + technical mitigation*' and '*Decreased meat scenario + technical mitigation*') that consider technological mitigation options for all agricultural activities. Regionally specific emission reduction factors are based on a study that investigates agricultural non-CO_2 reduction potential at different carbon prices (Lucas et al., 2007). In these scenarios we applied maximum attainable emission reduction factors for each time step at highest carbon price of 1000 $ per ton of carbon. In general, we identified highest GHG reduction potentials for CH_4 emissions from rice production, mainly due to better water management, followed by N_2O and CH_4 emissions from manure management by e.g. covering of storages or adoption of biogas plants (Cole et al., 1997; Thorpe, 2009), N_2O emissions

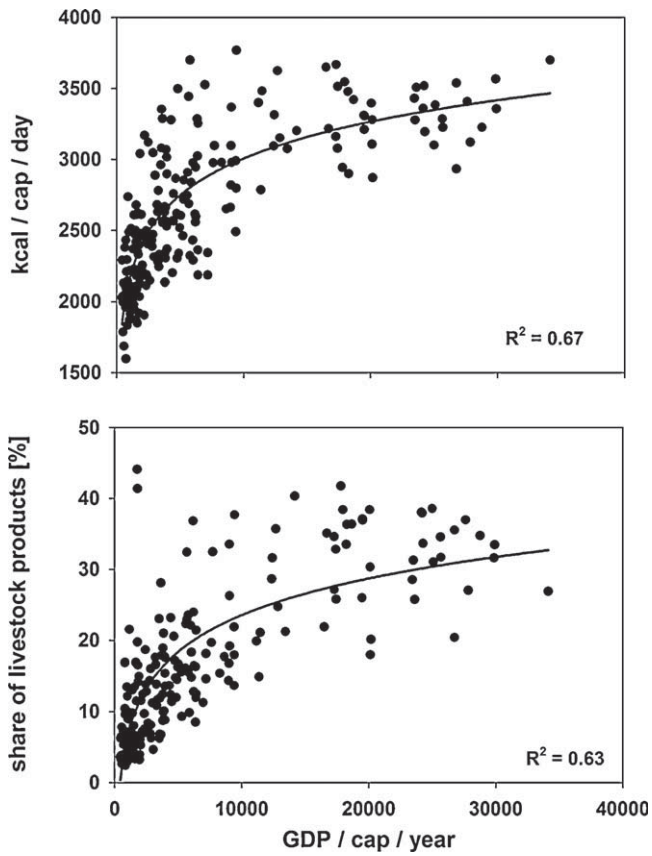

Fig. 3. (A) Regression of food energy demand and (B) share of livestock products in total caloric intake against income (gross domestic product per capita) based on data for 105 countries (in 1990 and 2000).

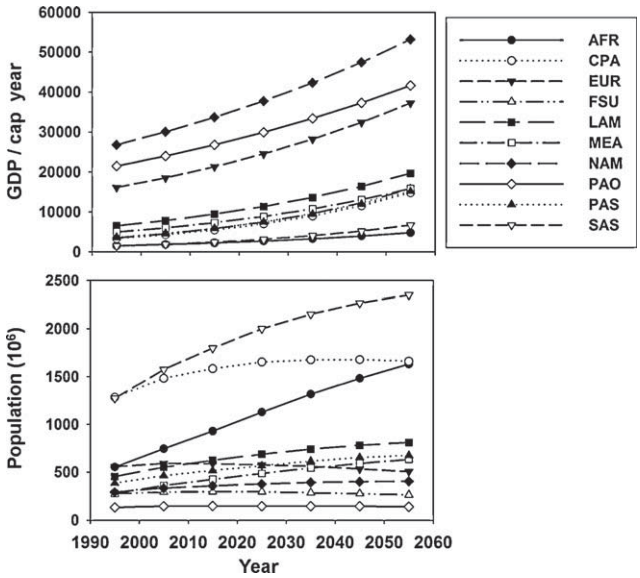

Fig. 4. Exogenous scenario inputs on regional population and GDP growth for all model regions.

from soils due to improvement of N-efficiency (Smil, 1999) and CH_4 emissions from enteric fermentation due to better herd management and ameliorated quality of livestock through breeding and improved feeding practices (Smith et al., 2008).

For each scenario, we run the MAgPIE model in six 10-year time steps from 1995 until 2055 in a recursive dynamic manner. The model is driven by external scenarios on population growth and GDP growth (Fig. 4) taken from the SRES A2 scenario (IPCC, 2000). Global population increases up to about 9 billion in the year 2055, and average world income per capita reaches about 15,000 US$ (in 1995 purchasing power parity terms). There are no climate impacts on future yields, i.e. relative yield variability between grid cells is constant at 1995 levels. Scenario specific calorie intake per person per day and share of livestock products are displayed in Fig. 5.

Additionally, we apply the MAgPIE model to investigate regional projections for agricultural non-CO_2 GHG emissions for the '*baseline scenario*' and present maps for agricultural CH_4 and N_2O emissions in 1995 and 2055.

3. Results

3.1. Regional projections

MAgPIE projects global agricultural non-CO_2 emissions (CO_2-e) to increase (compared by 1995) by 63% until 2055 (Fig. 6). The contribution of different sources and emissions varies widely between regions. Global CH_4 emissions will increase by 57% and global N_2O emissions by 71%. Fig. 7 shows the related global maps of CH_4 and N_2O emissions.

Global CH_4 emissions from enteric fermentation will increase by 70%, N_2O emissions from soils by 62% and total emissions from

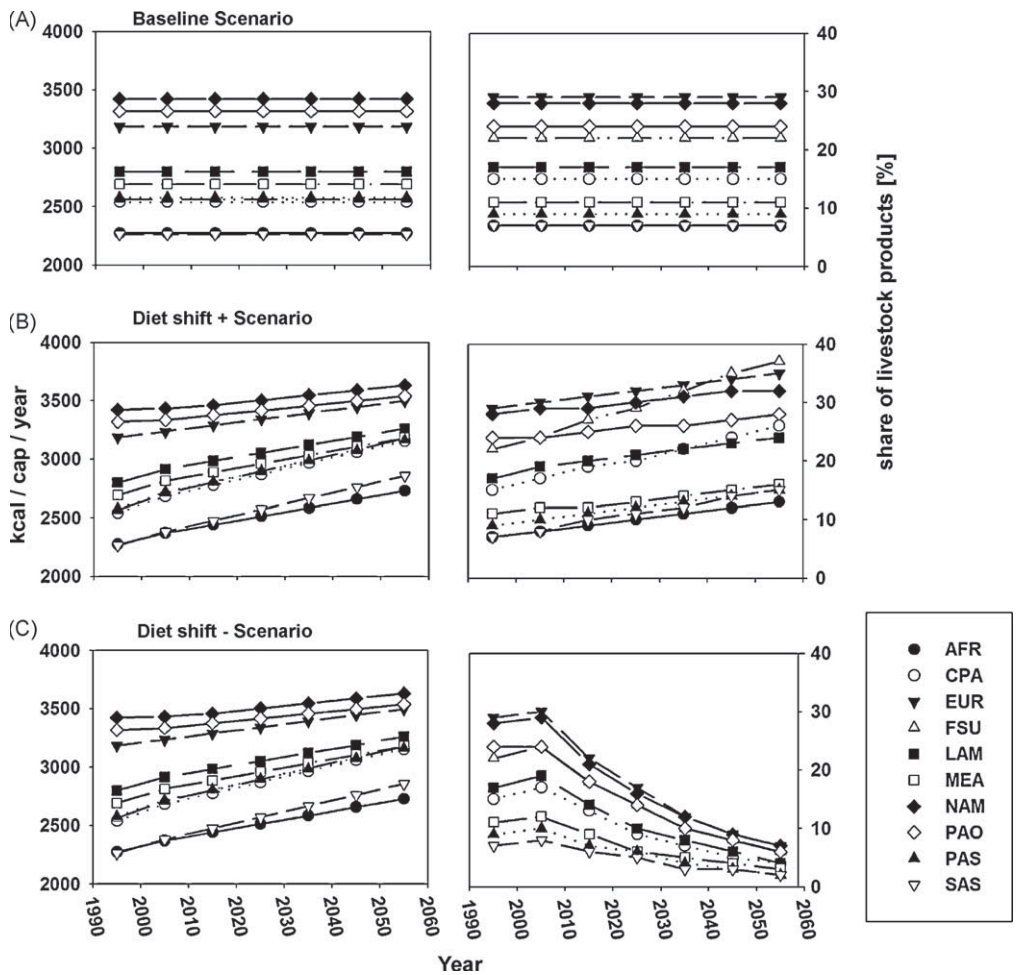

Fig. 5. Calorie intake per person per day and share of livestock products for all world regions and all scenarios.

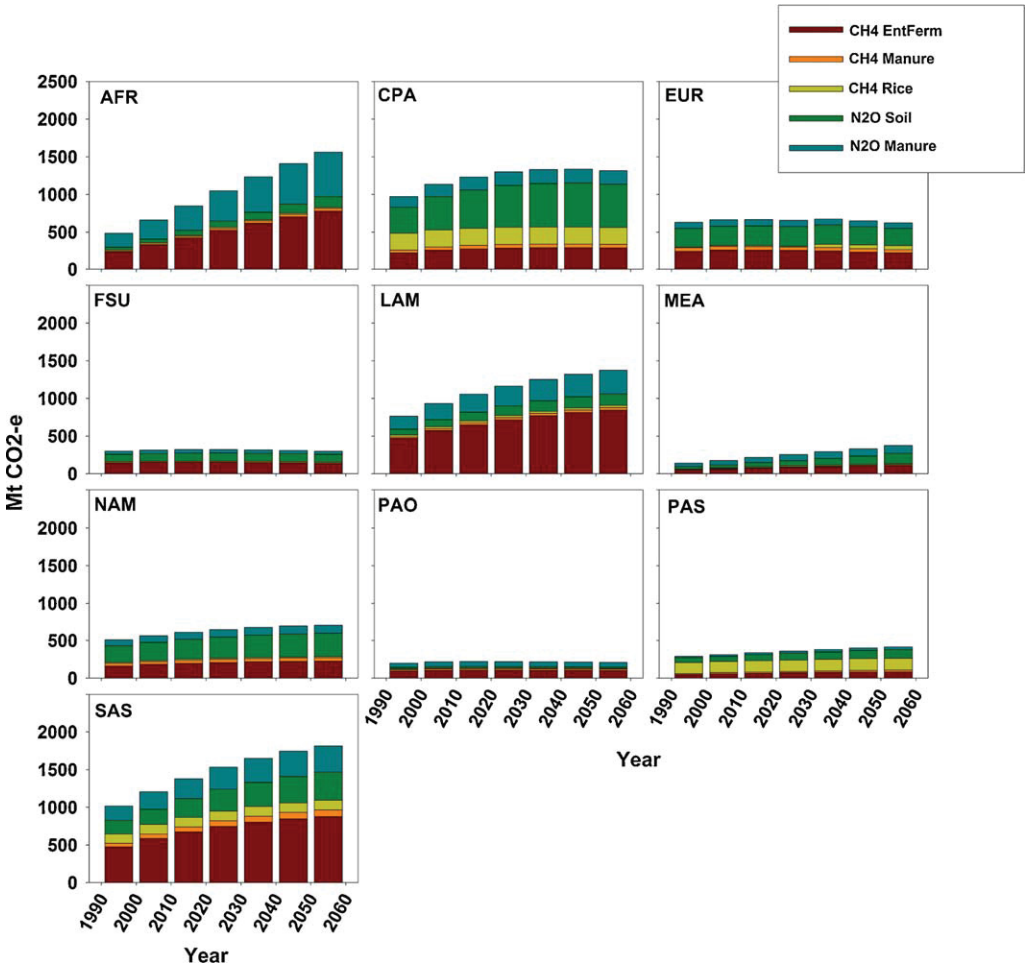

Fig. 6. Regional predictions for agricultural greenhouse gas emissions (BAU).

manure management by 75% (CH_4: 48%; N_2O: 82%). CH_4 emissions from rice production are far less affected (4%).

Less developed regions like AFR (221%), MEA (167%), SAS (78%), LAM (79%), PAS (42%) and CPA (35%), where population numbers are projected to rise most, show the highest increase in total non-CO_2 emissions until 2055. In contrast, developed regions like EUR (−1%) and PAO (4%) with least projected population growth rates show the lowest increase in total non-CO_2 emissions until 2055 (Fig. 8).

3.2. Scenario analysis

Our '*baseline scenario*' (Fig. 9), i.e. constant share of animal products in human diets, reveals that global agricultural non-CO_2 emissions will increase from 5314 CO_2-e in 1995 to 8690 CO_2-e until 2055. The '*Increased meat scenario*' (Fig. 10A) shows that changing dietary preferences towards more livestock products with increasing income does enhance global agricultural greenhouse gases by 76% until 2055 compared to the baseline scenario. Global CH_4 emissions from enteric fermentation rise by 101%, CH_4 emissions from manure management by 104%, N_2O soil emissions by 11% and N_2O from manure management by 114%. In contrast, CH_4 emissions from rice cultivation and N_2O soil emissions decrease by 11%. The '*Decreased meat scenario*' (Fig. 10B), i.e. a decadal reduction in the demand for meat products by 25%, decreases global non-CO_2 emissions in 2055 by 51% compared to our baseline model. High decreases from enteric fermentation (−68%) and manure management (−69%) are contrasted by lower decreases in N_2O soil emissions (−24%) and increases in CH_4 emissions from rice production (6%).

In the '*Increased meat scenario + technical mitigation*', technological mitigation options in the agricultural sector decrease global agricultural non-CO_2 emissions by 36% in 2055, compared to the '*Increased meat scenario*', but still increase emissions by 13%, compared to the '*baseline scenario*'. Lowest non-CO_2 emissions can be found in the '*Decreased meat scenario + technical mitigation*' where reduced demand for meat products combined with mitigation options in the agricultural sector will lead to global agricultural non-CO_2 emissions of 2519 CO_2-e in 2055.

3.3. Comparison with other studies

Simulation results from the MAgPIE model for CH_4 emissions from enteric fermentation, manure and rice as well as for N_2O emissions from soil and manure agree with previous global and regional studies for non-CO_2 emissions from the agricultural sector.

To test the quality of our simulation model we first compare simulated regional agricultural non-CO_2 GHG emissions from MAgPIE for 2005 with a study on agricultural non-CO_2 GHG emissions conducted by US-EPA (2006) that is based on reported national emissions. The EPA study covers all non-CO_2 GHG emissions globally.

We aggregate national data from US-EPA to the 10 MAgPIE regions and check how total regional CH_4 (enteric fermentation, manure management and rice production) as well as total regional

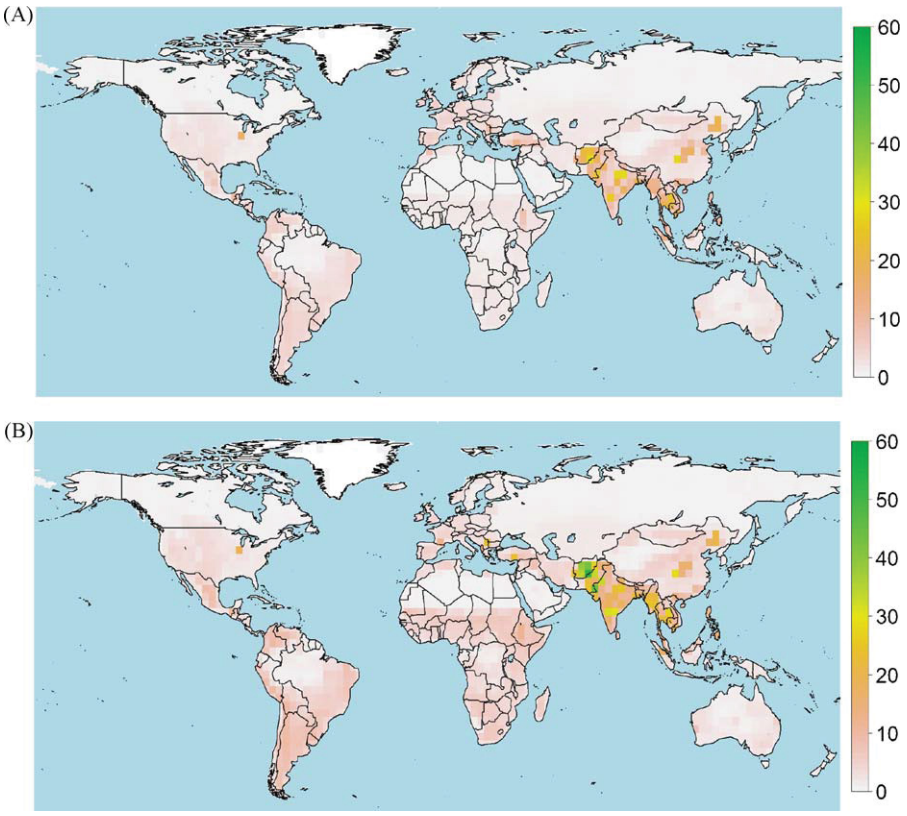

Fig. 7. Cell-specific agricultural greenhouse gas emissions for total agricultural CH_4 emissions in Mt CO2-equ for 1995 and (B) 2055.

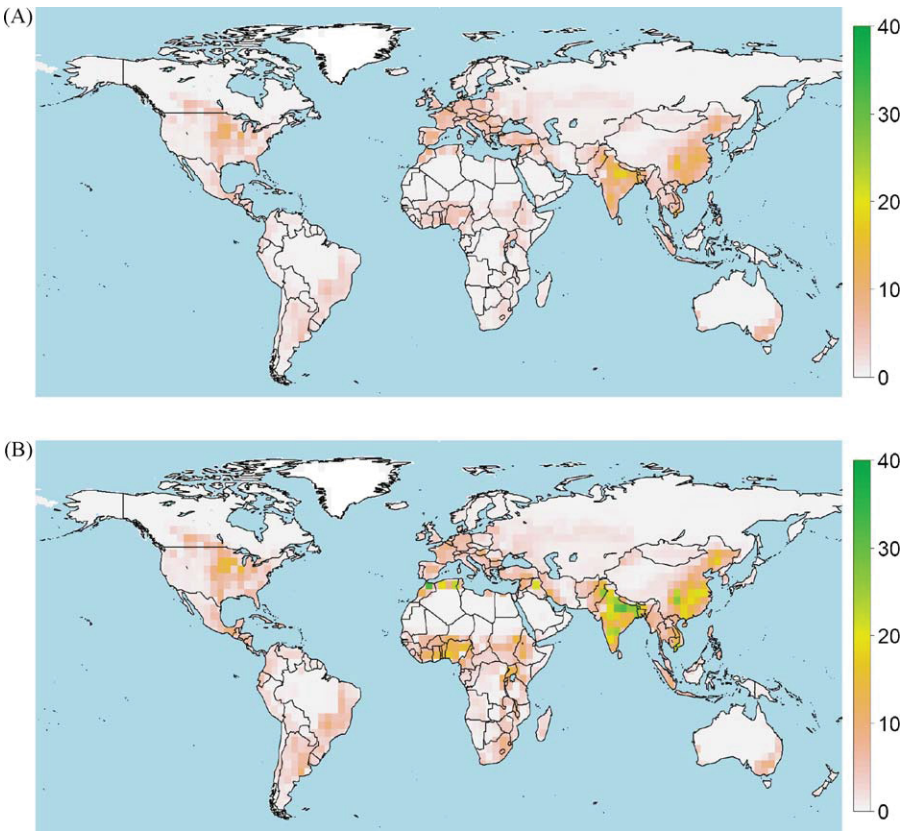

Fig. 8. Cell-specific agricultural greenhouse gas emissions for total agricultural N_2O emissions in Mt CO_2-equ for 1995 and (B) 2055.

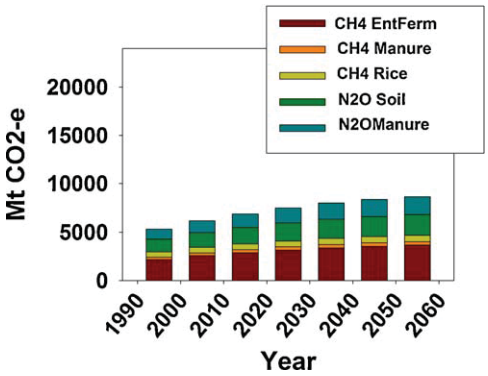

Fig. 9. Global emissions for 'baseline scenario'.

N_2O emissions (agricultural soils and manure management) correspond. Fig. 11 shows a scatter plot that combines total CH_4 and N_2O emissions. Total regional CH_4 ($R^2 = 0.88$) and N_2O ($R^2 = 0.70$) emissions agree between MAgPIE and US-EPA (2006). The only exception is SAS where MAgPIE projects higher animal numbers and associated GHG emissions. This discrepancy originates from the fact that consumption per capita of meat is lower in South Asia than income alone would explain.

Future trends for agricultural emissions are difficult to project as the most crucial drivers for agricultural GHG emissions – supply and demand – are highly uncertain due to impacts of world markets and consumption patterns on national livestock production patterns. The study of US-EPA (2006) assumes that dietary preferences remain constant. For the year 2015 US-EPA reports increases in global agricultural CH_4 emissions by 21% and N_2O emissions by 32% (relative to 1995). These results are well comparable to simulation data from the MAgPIE model with constant diet preferences that suggest increases in agricultural CH_4 by 27% and N_2O by 33% for this period.

Finally, we compare our model results for the '*Increased meat scenario*' with other studies that cover only single world regions or focus on single emission sources for agricultural non-CO_2 GHG emissions (Table 1). Overall our simulation results are in good agreement with these studies.

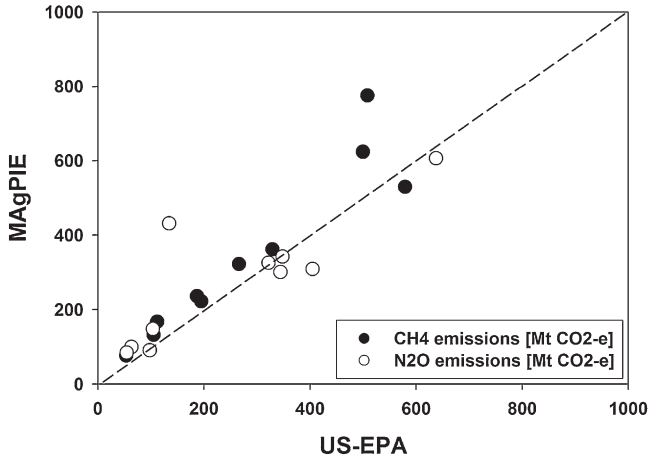

Fig. 11. Comparison of MAgPIE model results with GHG emissions from US-EPA for the year 2005.

Table 1
Comparison of GHG emissions simulated by MAgPIE with other studies.

MAgPIE	Stehfest et al. (2009)	Steinfeld et al. (2006)
Total global CH_4 emissions from livestock production [Mt CO_2-eq]		
2895 (2005)	2500 (2000)	2578 (2004)
4029 (2055)	4250 (2050)	
MAgPIE		**Herrero et al. (2008)**
Total CH_4 emissions from livestock production in Africa [Mt CO_2-eq]		
235 (1995)		195 (1995)

4. Discussion

The main objectives of this paper were to present the implementation of agricultural non-CO_2 GHG emissions into the global land use allocation model MAgPIE and to apply this modeling approach for testing the impact of changing consumption patterns and technical mitigation options in the agricultural sector on global GHG emissions from agricultural production in the future.

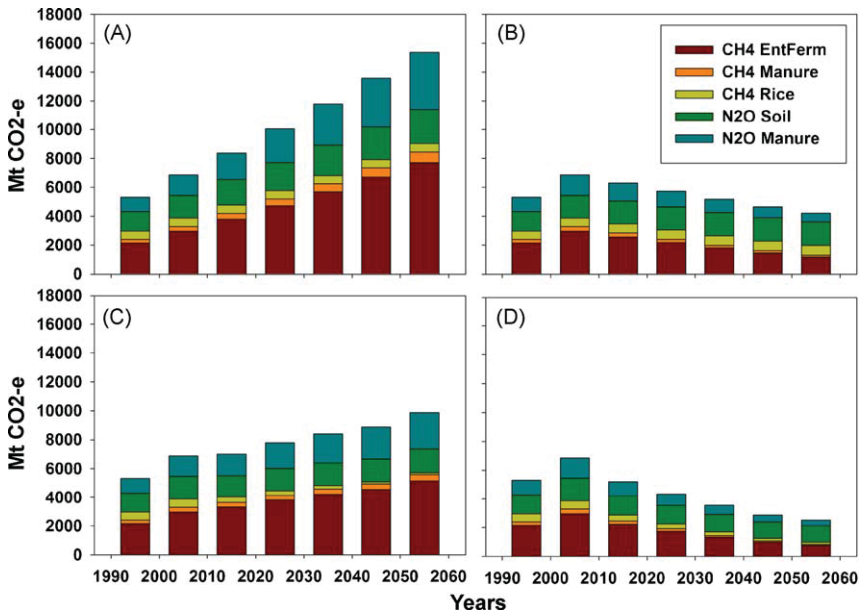

Fig. 10. Global emissions for (A) diet shift + scenario, (B) diet shift − scenario, (C) diet shift + scenario M and (D) diet shift − scenario M.

Agricultural non-CO_2 GHG emissions depend mainly on the type of agricultural activity but projecting land use patterns poses many problems as future supply and demand of agricultural products are highly uncertain. A proper treatment of these interactions is therefore an asset to investigate future GHG emissions from agriculture. MAgPIE is suited to investigate potential pathways for the agricultural sector by combining socio-economic information on population, income, food demand, and production costs with spatially explicit environmental data on potential crop yields, and water availability for irrigation. As a consequence, agricultural non-CO_2 GHG emissions simulated by the MAgPIE model agree with previous studies (e.g. US-EPA, 2006; IPCC, 2007; Stern, 2006) for non-CO_2 emissions from the agricultural sector.

Annual GHG emissions from agriculture are expected to increase in coming decades (IPCC, 2007). But future agricultural GHG emissions will depend on (i) future consumption patterns, i.e. the demand for agricultural products, affected by population growth, per capita caloric intake and changing dietary preferences, such as an increased consumption of meat and dairy products but also on (ii) future technical mitigation options in the agricultural sector. In order to analyze the impact of different consumption patterns and mitigation potentials, we compared agricultural non-CO_2 GHG emissions until 2055 from an '*Increased meat*', '*Decreased meat*' and two mitigation scenarios ('*Increased meat scenario + technical mitigation*' *and* '*Decreased meat scenario + technical mitigation*') with our '*baseline*' scenario.

In our *baseline* scenario, where food energy consumption and dietary preferences remain constant on the level of 1995, total non-CO_2 GHG emissions climb up to 8690 Mt CO_2-e in 2055.

Increases in GHG emissions are mainly associated with world population increases from currently 6.5 billion to 9 billion by 2050 (UN, 2005). While GHG emissions in the developed regions are close to stagnant, the lion's share of increase in population numbers and GHG emissions is occurring in developing and emerging regions.

But livestock herds across much of the developing world will grow even more if increasing food energy consumption and changing dietary preferences towards more consumption of meat and dairy products with increasing income are taken into account. Increased production of beef, poultry, and pork will lead to higher CH_4 emissions from enteric fermentation, increases in manure with consequent increases in GHG emissions, and higher demand for feedstock associated with higher levels of fertilizer application and N_2O emissions. In contrast, under a scenario of reduced meat consumption ('*Decreased meat scenario*'), global agricultural non-CO_2 GHG emissions would decrease even compared to 1995. Yet, high decreases from enteric fermentation, manure management and soil emissions due to fodder cropping are balanced by modest increases in N_2O soil emissions from food cropping and CH_4 emissions from rice production.

However, strategies that lead to shifts in consumption patterns and promote reduction of livestock numbers raise further considerations.

First, livestock food products are very valuable for nutrition as they contributed globally an average of 33% of protein to dietary intakes in 2003 (Steinfeld et al., 2006). For many poor and undernourished people in the developing world who frequently suffer from protein deficiencies livestock products are important parts of food consumption. In contrast, less meat-oriented diets in the developed regions would have positive health effects (Popkins et al., 2001). In general, while shifts away from meat-oriented diets lead to reduced GHG emission levels, they might have negative impacts on already stressed fish populations (Thorpe, 2009). Second, on average, the livestock sector's GDP currently accounts for 40% of agricultural GDP (Steinfeld et al., 2006). And specifically in developing countries the livestock sector provides livelihood for many poor people.

Applying the mitigation scenarios indicated that technological mitigation options in the agricultural sector have also the capability of decreasing non-CO_2 GHG emissions significantly. However, these technological mitigation options are not as effective as changes in food consumption. Highest reduction potentials will be achieved by a combination of both approaches.

For both, technical and behavioral GHG reduction options, incentives are needed to guarantee that the agricultural sector contributes its share to climate change mitigation. Putting a price on GHG emissions via a tax or emission trading scheme is an economically sound prescription for addressing the negative externalities of greenhouse gases. On the one hand, a price on non-CO_2 GHG emissions from the agricultural sector will lead to less application of GHG-intensive practices and will set an incentive for developing new technologies and implementing existing ones. On the other hand, as it seems very unlikely that behavioral changes will happen voluntarily, GHG pricing or regulation will also modify consumer prices and therefore affect changes in consumption patterns.

In this article, the future contribution of agriculture to global non-CO_2 GHG emissions has been investigated. In addition to direct agricultural emissions such as CH_4 from enteric fermentation of livestock or N_2O from soils, the agricultural sector is also indirectly responsible for emissions in other sectors (Stern, 2006).

First of all, among all land types, the amount of carbon stored in croplands is the lowest. Land use change such as deforestation in tropical regions that is mainly driven by agricultural production contributed about 3.8 PgC per year from 1990 to 1999 (Houghton et al., 2001). Furthermore, the production of fertilizers is another important source of greenhouse gas emissions. Further enhancement in agricultural productivity needed for fulfilling the future demand for feed and food will require higher levels of fertilizer application. The production of fertilizers is energy intensive, and contributed between 300 and 600 Mt CO_2-eq/yr, representing between 0.6 and 1.2% of the world's total GHG emissions (Wood and Cowie, 2004).

Knowledge on potential future GHG emissions from the agricultural sector is of huge importance for the assessment of mitigation potentials and options in the agricultural sector (Smith et al., 2008). But our model does not only provide the opportunity to investigate potential future non-CO_2 GHG emissions from the agricultural sector under different conditions such as growing food demand. The model structure of MAgPIE also supports the inclusion of other land-intensive goods such as bio-energy carriers (Lotze-Campen et al., in press). Combining these land use pressures with our approach to estimate agricultural non-CO_2 GHG emissions will help to assess the net-contribution of bio-energy to climate change mitigation by including additional N_2O emissions from the soil due to agricultural intensification for food, feed and bio-energy crops.

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Appendix A

A.1. N_2O emissions from the soil

Nitrous oxide (N_2O) is produced naturally in soils through the microbial process of denitrification and nitrification. Anthropogenic activities such as fertilization, production of nitrogen-fixing crops add nitrogen to the soil, thereby increasing the amount of nitrogen available for nitrification and denitrification.

We calculate N_2O emissions from agricultural soils ($N2O_{SOIL}$) in time t within a grid cell c by including direct ($N2O_D$) as well as indirect emissions ($N2O_I$):

$$N2O_{SOIL_{t,c}} = N2O_{D_{t,c}} + N2O_{I_{t,c}} \tag{1}$$

A.1.1. Direct N_2O emissions from agricultural soils

We estimate direct N_2O emissions ($N2O_D$) from the N input by synthetic fertilizer (NS), crop residues (NR), N-fixing crops (NF), and manure application (NM):

$$N2O_{D_{t,c}} = (NS_{t,c} + NF_{t,c} + NR_{t,c} + NM_{t,c}) \times e_f \tag{2}$$

The emission factor e_f is estimated by Bouwman (1996) to lie between 0.0025 and 0.0225 kg N_2O–N kg N. We apply the default value 0.0125 kg N_2O–N kg N as used in the IPCC guidelines (1997).

A.1.2. Calculation of total nitrogen input by synthetic fertilizer

We assume that nitrogen input by synthetic fertilizer is correlated with the yield level of a crop: Higher yields need higher nitrogen input, as lack of reactive nitrogen is a limiting constraint to plant growth. To obtain the correlation coefficients for each crop type, we used expert guesses for average fertilizer application per crop (IFA, 2002) of 80 countries, and the corresponding average yield per country from FAOSTAT (FAO, 2008). Assuming a linear relationship between yield (P) and nitrogen fertilizer input (NS), we derive the relationship:

with a being the intercept, m the slope and Cr the crop type.

$$NS_{t,c} = \sum_{Cr} a_{Cr} + m_{Cr} \times P_{t,cCr} \tag{3}$$

A.1.3. Calculation of total nitrogen input by N-fixing crops

Nitrogen input from N-fixing crops (NF) within cell c at time t is calculated from dry yield production (P) of all N-fixing crops (Cr_F) within this cell multiplied by the fraction of nitrogen (n_F) in this respective crop:

$$NF_{t,c} = \sum_{Cr_F} 2 \times P_{F,t,c} \times n_F \tag{4}$$

The factor 2 refers to the harvest index.

A.1.4. Calculation of total nitrogen input from crop residues

Nitrogen input from crop residues (NR) are calculated as:

$$NR_{t,c} = 2 \times \left(\sum_{Cr_F} P_{F_{t,c}} \times n_F + \sum_{Cr_{NF}} P_{NF_{t,c}} \times n_{NF} \right) \times (1 - f_r) \times (1 - f_b) \tag{5}$$

where P_F and P_{NF} describe the total annual biomass production of N-fixing (Cr_F) and non-N-fixing (Cr_{NF}) crops and n_F and n_{NF} the respective fraction of nitrogen in these crops. fr stands for the fraction of crop residues that is removed from the field as crop and fb the fraction of crop residue that is burned. The factor 2 converts edible crop production to total crop biomass.

A.1.5. Calculation of total nitrogen input from manure application

Direct N_2O emissions result from livestock manure that is applied to soils either through daily spread operations (all applied manure) or direct deposition on pastures, range, and paddocks (PRP) by grazing livestock.

Manure nitrogen used as fertilizer (NM) is estimated by:

$$NM_{t,c} = Nex_{t,c} \times (1 - (f_b + f_g)) \times (1 - f_e) \tag{6}$$

where Nex describes the total N excretion by animals. f_b stands for the fraction of livestock N contained in excrements burned for fuel, f_g for the fraction of livestock N deposited onto soil during grazing and f_e for the fraction of total N excretion that is emitted as NOx or NH_3.

To estimate N excretion Nex, livestock L products (P_L) within a cell c for each time step t are multiplied by livestock product specific N excretion (ex_{PL}):

$$Nex_{t,c} = \sum_L P_{L,t,c} \times ex_{P_L} \tag{7}$$

A.1.6. Indirect N_2O emissions from agricultural soils

Indirect N_2O emissions ($N2O_I$) enter the atmosphere by one of two pathways: (1) atmospheric deposition of NOx and NH_3 (originating from fertilizer use and livestock excretion of nitrogen), and (2) leaching and runoff of nitrogen from fertilizer applied to agricultural fields and from livestock excretion.

$$N2O_{I_{t,c}} + N2O_{V_{t,c}} + N2O_{L_{t,c}} \tag{8}$$

Votalization of NH_3 and NOx during fertilization – Only a specific fraction of applied Nitrogen from synthetic fertilizer (NS) and manure (NM) turns into NH_3 and NOx. The emission factor e_v of 10% is the share of NH_3 and NOx turning into N_2O due to votalization proposed by the IPCC Guidelines (1997):

$$N2O_{V_{t,c}} = ((NS_{t,c} \times f_{V_s}) + (NM_{t,c} \times f_{V_M})) \times e_v \tag{9}$$

Leaching of N during fertilization – We use the IPCC recommendation (1997) that 30% of both, synthetic fertilizer (NS) and manure (NM) applied is lost to leaching and surface run-off (f_l) and 3% of this lost N is emitted as N_2O (e_l):

$$N2O_{L_{t,c}} = (NS_{t,c} + NM_{t,c}) \times f_1 \times e_l \tag{10}$$

A.2. N_2O emissions from animal waste management systems

N_2O from animal waste management systems (AWMSs) is produced by the nitrification and denitrification of the organic nitrogen content in livestock manure and urine. The basic equation to estimate N_2O emissions ($N2O_{AWMS}$) from $AWMS$ within a cell c at time t is as follows:

$$N2O_{AWMS_{t,c}} \sum_{AWMS} Nex_{AWMS_{t,c}} \times e_{AWMS_c} \tag{11}$$

where e describes the MAgPIE region-specific N_2O emission for the various $AWMS$ and Nex stands for the N excretion per $AWMS$ within a cell c.

We estimated Nex by:

$$Nex_{AWMS_{,t,c}} = \sum_L P_{L,t,c} \times Nex_{P,c} \times AWMS_{P_{L,c}} \tag{12}$$

Here, livestock products (P_L) (ruminant meat, non-ruminant meat, milk) within a cell, calculated by MAgPIE for each time step t are multiplied by livestock product specific N excretion (*Nex*) and a fraction of *Nex* that is managed in one of the different AWMS for animal products.

A.3. CH_4 emissions from animal waste management systems

CH_4 emissions from Animal Waste Management Systems ($CH4_{AWMS}$) are produced during the anaerobic decomposition of manure. To estimate $CH4_{AWMS}$, livestock products (P_L) are multiplied by specific N emission factors (e_P):

$$CH4_{AWMS_{t,c}} = \sum_L P_{L,t,c} \times e_{P_{L,c}} \tag{13}$$

Default values for e_{PL} differentiate between animal species, temperature, as well as developed and developing countries. We used the PIK Global Climate Dataset (2007) to assign the appropriate emission factors to the MAgPIE regions.

A.4. CH_4 emissions from enteric fermentation

Enteric fermentation refers to a fermentation process whereby microbes in an animal's digestive system ferment food. Methane is produced as a byproduct and is exhaled by the animal. Domesticated ruminants such as cattle, buffalo, sheep, goats, and camels account for the majority of methane emissions in this sector. The basic equation to estimate CH_4 emissions from enteric fermentation ($CH4_{FERM}$) in time t within cell c is as follows:

$$CH4_{FERM_{t,c}} = \sum_L P_{L,t,c} \times e_{PL} \tag{14}$$

where livestock products P_L within a cell are multiplied by a livestock product specific emission factor (e_{PL}).

A.5. Spatial distribution of livestock activities

In general, livestock production tends to concentrate in areas favored by cheap input supplies (particularly feed), and by good market outlets for livestock products such as urban areas and infrastructure (Steinfeld et al., 2006). However, as we do simulate demand patterns on a regional level and do not account for local markets such as urbanization and infrastructure, our spatial distribution of animals is only related to input supplies. Our main feature for allocating livestock production P_L (ruminant meat, milk and non-ruminant meat) is livestock's demand for feedstock F. We assume that ruminant meat (*rum*) and milk production (*milk*) are mainly associated with pasture land and non-ruminant meat (*non-rum*) with crop production. Therefore, we calculate cell-specific c livestock production in time t by multiplying regional livestock production with the proportion of area A for the respective feedstock in each cell c to the region r:

$$P_{L,c,t} = \frac{A_{F,c,t}}{A_{F,R,t}} \times P_{L,R,t} \tag{15}$$

A.6. CH_4 emissions from rice cultivation

The anaerobic decomposition of organic matter in flooded rice fields produces methane. Several factors influence the amount of methane produced, including water management practices and the quantity of organic material available to decompose. To estimate CH_4 emissions from rice production ($CH4_{RICE}$) in time t within a grid cell c we take a seasonally integrated emission factor (f_R) for different ecosystem water management regimes and organic amendments into account:

$$CH4_{RICE_{t,c}} = f_R \times A_{R_{t,c}} \times 10^{-12} \tag{16}$$

Here, A_R stands for the area under rice cultivation under m and o conditions. In practice, it is difficult to obtain specific emission factors for each commonly occurring set of rice production conditions in a country, so the IPCC Guidelines instruct countries to first obtain a baseline emission factor (*bEF*) for continuously flooded fields without organic amendments. Different scaling factors are then applied to obtain an adjusted seasonally integrated emission factor for the harvested area as follows:

$$f_R = bEF \times s_m \times s_o \tag{17}$$

where s_m is a scaling factor that accounts for the differences in ecosystem and water management regime and s_o is a scaling factor for organic amendments.

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Chapter VII

Stuck in the Anthropocene:

The case of reactive nitrogen

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Content of chapter VII: Stuck in the Anthropocene

Stuck in the Anthropocene: The case of reactive nitrogen.

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Abstract

Reactive nitrogen (Nr) is an indispensable nutrient for agricultural production and human alimentation. At the same time, terrestrial, aquatic, air and atmospheric Nr pollution is estimated to cause damage in the magnitude of 0.3% to 3% of global GDP, mainly to human health and ecosystem services[1].

An influential study[2] suggests to reduce anthropogenic Nr sources below a 'planetary boundary' of 35 Teragram (Tg) Nr in order to keep the earth system within the stable conditions of the Holocene, which is about one sixth of the current level[3]. As agriculture is responsible for three quarters of current anthropogenic Nr sources[3], this sector is central to Nr mitigation. Key mitigation options to reduce agricultural requirements of Nr include the improvement of Nr efficiency in crop and animal production systems, the reduction of food waste in households, and lower consumption of Nr-intensive products like meat and milk[1].

However, the mitigation potential of these measures remains unclear, especially under the added pressure of population growth and changes in food consumption.

We show that, with regard to Nr pollution, the return to Holocene conditions is out of reach for decades to come.

If current trends persist, crop production alone will require Nr sources of 220 Tg Nr by the year 2050. Even if all key mitigation options[1] were implemented, crop production still requires 86 Tg Nr per year by the year 2050, more than twice the level of the 'planetary boundary'.

Our findings stress the urgent need for ambitious mitigation policies, targeting both farmers and consumers. As Nr pollution will also persist under mitigation, a strategy is required that protects vulnerable systems from Nr pollution.

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Terrestrial sources of reactive nitrogen (Nr) before the industrial revolution are estimated at 40-100 Teragram (Tg) Nr per year, stemming mostly from micro-organisms that fix Nr biologically from unreactive dinitrogen (N_2) in the atmosphere[4]. Since industrialisation, human activity raised the yearly terrestrial Nr sources by 210 Tg Nr[3]. Earth's nitrogen cycle left the Holocene and entered human-made Anthropocene[2]. Today, the largest anthropogenic source of Nr is the inorganic fixation of industrial nitrogen fertilizer by the Haber-Bosch Synthesis, followed by the biological fixation of Nr through the cultivation of leguminous plants and sugarcane. Additional to biological and inorganic fixation for agriculture, Nr is also fixed for industrial purposes[5], and released by the combustion of biomass and fossil fuels. Parts of it reach croplands via atmospheric deposition[6]. Moreover, considerable amounts of Nr stored in cropland soils are released when soil organic matter depletes through soil management[7].

While passing through the agricultural supply chain, the Nr from these sources is subsequently lost through denitrification, leaching and volatilisation on the field, in manure management, waste and sewage[1,8–11]. As there is little long-term storage of Nr in the agricultural sector, Nr sources correspond approximately to Nr losses, and are an easily communicable indicator for the disturbance of environmental systems by Nr pollution[2]. Lost Nr can have a cascade of effects on the environment[11,12]. For instance, in combination with other nutrients, excessive Nr can lead to the disturbance of aquatic and terrestrial ecosystems through eutrophication. In the air, it leads to the formation of ozone and particular matter. Furthermore, Nr affects different atmospheric processes influencing the climate and ozone layer. Negative impacts on human welfare[1,11] include direct health damages like cancer and asthma, as well as the loss of ecosystem service, impairing for instance fish provision.

In this study, we apply a land-use and nitrogen-budget model[7,13–15] to estimate the Nr sources required to settle a given demand for agricultural products in the future and to analyse the potential of different Nr mitigation measures.

Our reference scenario without Nr mitigation is parameterized according to the "SSP2" storyline of the shared socio-economic pathways[16]. This middle-of-the-road scenario assumes a world population of 9 billion people in 2050 and a global GDP that more than triples to 230 trillion US-$Dollar_{05}$ in purchasing power parity[17]. Population groups which gain in prosperity demand a more affluent diet with a higher share of livestock products[18]. Therefore, crop production for food and feed increases by more than 50% compared to 2010, mainly by intensifying existing cropland. Over time, the developing countries adopt the industrialized livestock management of the Western

regions. Efficiency of Nr application on cropland improves in all world regions, but predominantly in the least efficient ones (see Methods, Table A1).

In the year 2050, according to this scenario, Nr sources of 220 Tg Nr are required for crop production. This Nr stems mostly from inorganically fixed fertilizer (F) and biological fixation, but also from atmospheric deposition (D) and the release of Nr by the depletion of soil organic matter (SOM) (Figure 1). While passing through the food supply chain, most of these nutrients are lost on the fields (LF), in household waste and sewage (LH, LS), and in animal waste management (LA). Other losses (O) include post-harvest and processing losses, slaughter waste, the application of manure on pasture land, and material use of agricultural products.

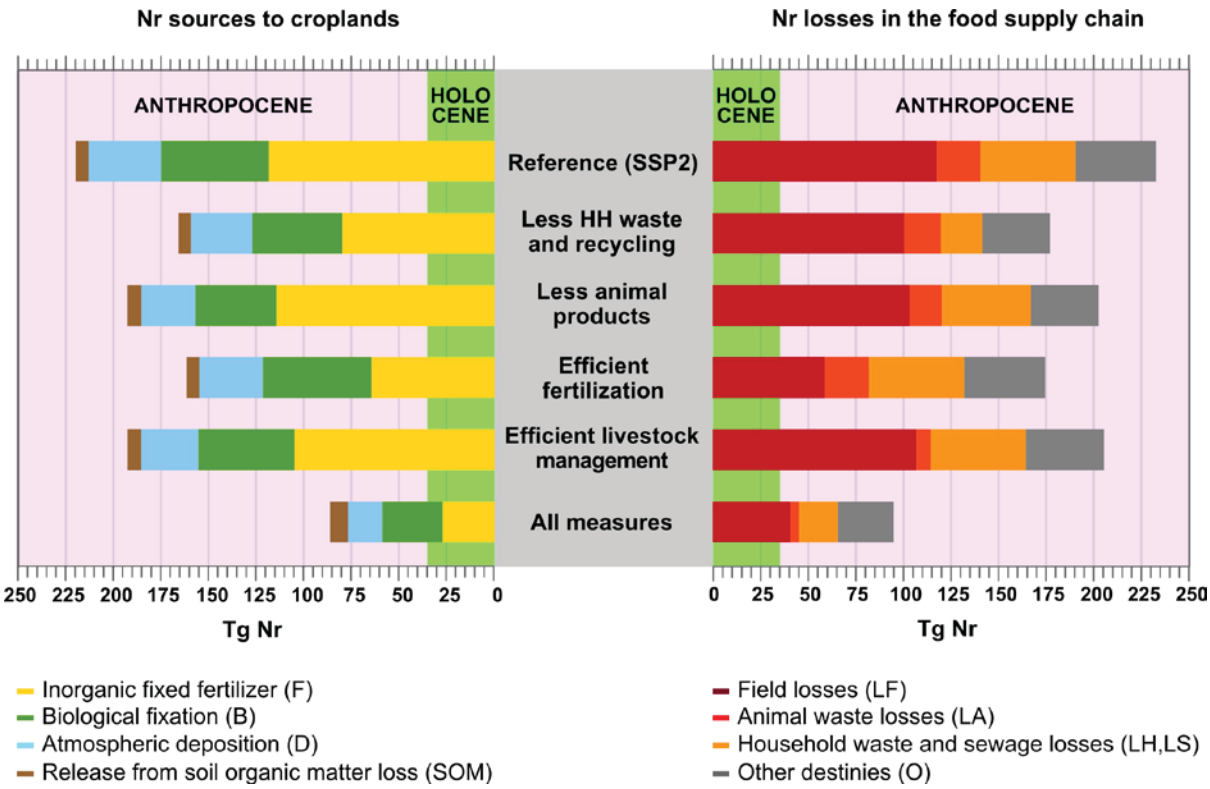

Figure 1: Reference and mitigation scenarios of Nr sources and the subsequent loss of Nr throughout the food supply chain in the year 2050. As the Nr losses from household (HH) waste and sewage include the nutrients from fish and animal products from pasture-fed livestock, losses are slightly higher than cropland Nr sources (see Supplementary Information).

Subsequently, we estimate the effect of four Nr mitigation options on Nr sources and losses (see Fig. 1). These options (see Methods for details) comprise the key mitigation actions proposed by the assessment "Our Nutrient World"[1] for cropland production, the livestock sector, societal consumption, and sewage. "Less household (HH) waste and recycling" could lower Nr sources (F+B+SOM+D) by 54 Tg Nr, removing the Nr losses that accrued in the production of wasted food,

and recycling the nutrients from sewage and household waste to the fields. “Less consumption of animal products” reduces Nr requirements by 27 Tg Nr, lowering losses in animal waste management as well as field losses by rendering parts of the feedstock production obsolete. “Efficient fertilization” can reduce Nr losses on the field by 58 Tg Nr. “Efficient livestock management” can lower required Nr sources by 27 Tg Nr through reduced losses in animal waste management and lower feedstock production.

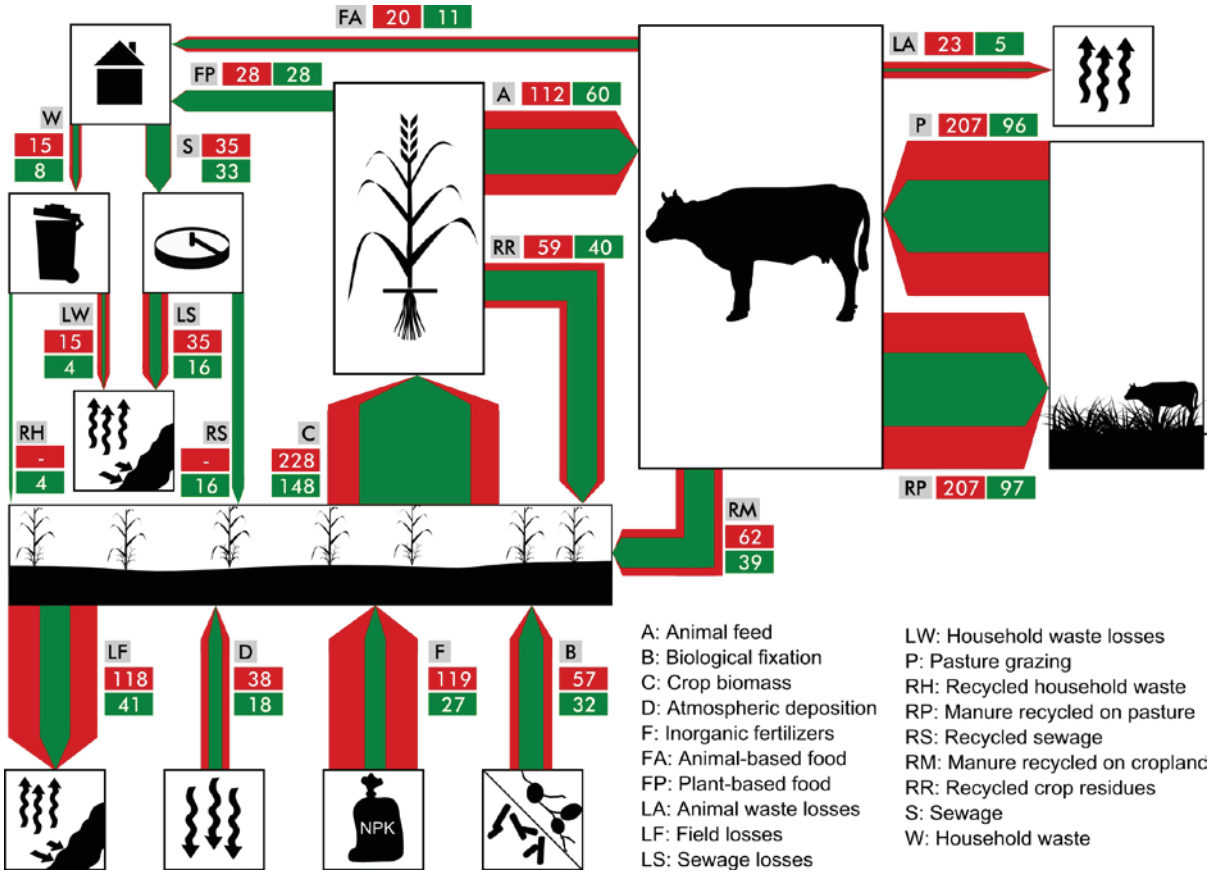

Figure 2: Nr flows in agriculture in the year 2050 in Tg Nr. Arrows describe flows in in the reference SSP2 scenario (red) and after applying all Nr mitigation measures (green). Only major flows are presented in this figure (see Supplementary Information for an overview of all simulated flows).

Combining all mitigation measures fundamentally alters the flows in the nitrogen cycle by 2050 (Fig 2). Crop biomass and grazed pasture are strongly reduced (C, P), and the dominant role of the livestock sector declines (A, P). Moreover, the absolute amount of recycled manure (RM) and crop residues (RR) as well as biological fixation (B) is reduced, but much less in relation to the savings of industrial fertilizer (F). The “full-chain efficiency”[1] expressing the share of newly fixed Nr arriving at the consumer is increased to 69%, considerably higher than the current level (22%) as well as the

level projected in SSP2 for 2050 (29%), and it exceeds even the proposed long-term target of 50%[1]. In total, the required Nr sources are reduced by 134 Tg. Nevertheless, crop production alone still requires Nr sources of 86 Tg Nr. This is more than twice the amount of 35 Tg Nr proposed as planetary boundary[2] that would allow to return from Anthropocene to Holocene conditions (see Fig 1). Disregarding Nr from atmospheric deposition, which is only partly an anthropogenic Nr source, the remaining Nr sources of 68 Tg Nr constitute still twice of the proposed planetary boundary. Despite the mitigation efforts, the 95 Tg Nr lost in the agricultural supply chain therefore continue to be a substantial environmental threat.

Beyond the analysed mitigation measures, few further options remain. These include lowering material use of agricultural products, abandoning the burning of crop residues, or reducing and recycling processing waste (Item O in figure 1). Our scenarios did also not consider morally problematic options like reducing population growth[19], profound behavioural changes like diets solely based on leguminous proteins[20], or technological breakthroughs until 2050 like cereals that fix nitrogen[21].

The scope of this analysis was focused on Nr requirements of cropland and livestock production for food and material use. The cultivation of bioenergy crops to replace fossil fuels or to sequester CO2 was not considered in this study. Producing 100 Exajoule (one fifth of current primary energy) of nutrient-efficient second generation bioenergy crops requires approximately 10 Tg additional Nr sources even under best practice[14]. Beyond agriculture, further Nr sources stem from NO_x and NH_3 emissions in the transport, industry, power and residential sectors. Some of these sources are difficult to circumvent, and even in the climate change mitigation scenario RCP 2.6, they can only be reduced from 37 Tg in 2010 to 27 Tg in 2050[22]. This shifts the nitrogen cycle even further away from Holocene conditions.

Nr pollution is already today far too high from an economic perspective – upper estimates of monetized damage caused by agricultural Nr pollution are as high as the agricultural value added[1,23]. If no paradigm change takes place, Nr pollution may still rise[7,10,24]. Our results affirm that strong mitigation actions have to be undertaken simultaneously in crop farming, animal husbandry, food processing, food consumption, sewage, and the recycling links between sectors[1]. In developed regions, improving food consumption patterns by reducing household waste and by lowering the share of animal products is probably the largest challenge, yet governments are reluctant to intervene in dietary habits[25]. For less developed regions, the main objective should be to reach best practice in nutrient management on farms.

Unfortunately, even in the case of a significant structural change of agriculture towards sustainability as described in this article, the nitrogen cycle is irretrievably stuck in the Anthropocene. A production system feeding 9 billion people will continuously be a source of dangerous pollution, directly harming people and perturbing the earth system via its linkage to biodiversity loss, global warming and ozone depletion. Next to mitigation, it is therefore crucial to identify core vulnerable systems that have to be protected from nutrient pollution[26] and to shift or denitrify the responsible pollution sources.

Methods Summary

The projections were made using the Model of Agricultural Production and its Impact on the Environment (MAgPIE)[7,13–15], revision 7294. For a given food demand[18] the model estimates cost-optimal production patterns under the limitations of scarce land and water. The model distinguishes 10 world regions, 18 crop groups and 5 livestock groups.

The model includes a nitrogen-budget module[8] that covers the agricultural Nr flows in cropland and livestock management, processing of agricultural products and food consumption. The closed budget approach guarantees that the sum of Nr fixation, Nr release and inflows from other sectors corresponds to the sum of Nr losses and Nr flows to other sectors. Similarly, intermediary closed budgets are also used on a regional level for cropland soils, distribution and processing of agricultural products, livestock feeding, manure management, and the household sector. Taking the example of cropland soils, the Nr withdrawn from the soil by crop biomass and lost to the environment has to equal Nr inputs like inorganic fertilizers, manure and atmospheric deposition.

A1 Extended Method Section

A1.1: Land-use model

The *Model of Agricultural Production and its Impact on the Environment,* abbreviated MAgPIE, is a mathematical programming model that is used to create long-term scenarios and assessments of global land use[7,13–15]. It features the major dynamics of the agricultural sector, like trade, technological progress and land allocation according to the scarcity of suitable soil, water and economic resources. As it treats agricultural products not only as economic values but also as physical good, MAgPIE can perform analysis of material flows. The products in the model comprise 17 crop groups, each with individual above- and belowground crop residues, 5 livestock production types, 8 types of conversion by-products originating from food processing, grazed pasture and scavenging. Products can be used for food, feed, other use (comprising material use and waste in the production chain) and seed, where applicable. Crop residues can also be recycled to soils or burned in the fields. The demand for food enters the model as an exogenous trajectory. The demand for feed depends on the level of livestock production, with each livestock category having individual regional feed baskets. Demand for material consumption and production waste are assumed to grow in proportion to food demand, while the demand for seed is a fixed share of crop production. The production of crops requires financial resources as well as land and water. Cropland expansion induces additional costs and is limited by biophysical conditions as well as by competing land use activities. Instead of increasing the agricultural area, the model can also invest into yield-increasing research and technology. Crop growth functions connect crop harvest to the production of above- and belowground residues. Similarly, the production of conversion by-products depends with a fixed conversion factor on the regional crop supply. Finally, livestock production requires financial resources, feed and water.

To match global demand with supply, MAgPIE optimizes global land use patterns and trade flows to minimize financial costs. To account for trade restrictions and distortions, the socio-economic world regions are forced to produce a certain share of their internal demand. The optimization problem is solved in consecutive time steps from 1995 to 2050, whereby the cropland area and the level of technology are passed on from one time step as input data to the consecutive time step.

A1.2: Nr budget module

MAgPIE was recently extended by a Nr flow module[7] that transforms all biomass flows in the model into Nr flows. The flows are consistently connected such that the Nr being fixed or entering the agricultural sector in the model corresponds to the Nr lost to the environment or flowing to sectors

outside of the scope of the model. Also every sub-budget of the model is balanced. Central to the model are regional nutrient budgets for cropland soils. Within each time step, all withdrawals from soils have to be balanced out by Nr inputs (Eq. A1.1).

Eq. A1.1: H+RA+RB-S-B1= SNUPE*(F+RM+RRA+RRB+D+B2+SOM+RH+RS)

Nr inputs covered by the model include inorganic fertilizer (F), manure recycled to croplands (RM), above and belowground crop residues remaining on the field (RRA, RRB), atmospheric deposition (D), biological fixation by free living microorganisms (B2), the Nr released when soil organic matter depletes after the opening of new cropland (SOM), as well as recycled household waste (RH) and sewage (RS) in the mitigation scenario. As only a share of Nr inputs to cropland soils is withdrawn by the crops, Nr inputs are multiplied by a regional soil Nr uptake efficiency (SNUPE). Soil withdrawals are calculated as Nr in harvest (H), above- and belowground residues (RA, RB) minus seed (S) and minus biological fixation within the plant (B1). The latter two Nr inputs are not taken up from the soil and are therefore not subject to fertilization losses. The difference between inputs and withdrawals represents the losses by denitrification, leaching and volatilisation (Eq. A1.2).

Eq. A1.2: LF = (1-SNUPE)*(F+M+RRA+RRB+D+B2+SOM+RH+RS)

Similarly, a budget is used to estimate the availability of manure. The Nr in the bodies of slaughtered animals as well as the Nr in milk and eggs is subtracted from the Nr in livestock feed. The remainder is assumed to be excreted. While the feed from pasture is assumed to be excreted back on pastureland, other excrements are distributed between different animal waste management systems with individual loss and recycling rates to cropland soils.

For this study, the model was extended for a more detailed representation of food intake, food waste and sewage. Total food demand (intake + waste) and the share of animal based products are estimated based on regression models[7,18]. The contribution of aquatic products to the protein supply by animal based products is held constant at the level of year 2005[27]. To estimate household food waste, we assumed that the minimum waste share in all countries is 15%, accounting for inevitable losses. Moreover, we assumed based on plausible regional intake estimates for developed regions[29] that the per capita demand exceeding 2200 kcal per capita per day is probably wasted. The resulting regional waste shares match well current estimates[28], but are dynamic for the future when per capita food demand rises. The Nr in sewage was estimated as the non-wasted share of food supply, not accounting for the relatively small share of Nr that is accumulated in human bodies or lost in

sweat, hears and nails[9]. In our nutrient flow model, recycled sewage and organic waste becomes an input for the soil nutrient balance comparable to manure, recycled residues or inorganic fertilizers.

A1.3 Description of mitigation scenarios

The scenario "Less household (HH) waste and recycling" assumes a reduction of waste share to 20%, resulting in a cut of per capita demand of each country at 2750 kcal (2200/0.8) in the year 2050. This is lower than the daily global per capita average in 2005 of 2787 kcal, and considerably lower than the OECD average of 3444 kcal[27]. Furthermore, we assume that half of the nutrients in household waste and sewage are recycled as fertilizers.

The scenario "Less animal products" assumes that no country derives more than 15% of its calories (29% of proteins) from animal based products. This corresponds to a "demitarian" western diet[1], in which the share of animal-based calories is half of the current level in Western regions, and reverts the trend of the reference scenario (Table 1).

The scenario "Efficient fertilization" estimates the effects of increasing SNUPE. High SNUPEs are difficult to reach, as this requires fertilizing the right amount of the right fertilizer at the right time and place, but also a better spatial integration of inhomogeneous Nr inputs like atmospheric deposition, manure and crop residues. We assume SNUPE increases globally from currently 53% to 75% in 2050 (Table 1), approximately halving relative losses compared to today. A SNUPE of 75% is 15 percentage points higher than the level of the best-performing world region, Europe[7], and also higher than the level in the most efficient agroecosystems[29].

The scenario "Efficient livestock management" consists of two components. Firstly, we assume that feed energy requirements of animals can be reduced by 25% relative to the reference scenario through better breeds and an improved deployment of existing feedstock. This increase in livestock productivity relative to the reference scenario corresponds well with the estimates of another study[30] if the trends are prolonged from 2030 to 2050. Lower feed intake also reduces manure excretion in the model. Secondly, the share of animal manure which is recycled from the stables to agricultural land is set to 90% in 2050, the highest plausible recycling share of the most efficient animal waste management systems[8].

The scenario "All measures" combines all above measures (Table 1).

Table A1: Scenario parameters

Year	1995	2010 [a]	2050	2050
Scenario		SSP2	SSP2	All measures
Population (bio)	5.7	6.8	9.1	9.1
GDP (trillion USD 05 in PPP)	39	67	230	230
Per capita demand (kcal per capita per day)	2677	2841	3178	2731 [b]
HH-Waste (share of demand)	22%	25%	31%	19% [b]
Animal-based calories in diet (share of demand)	16%	18%	22%	15% [b]
Soil Nr uptake efficiency (SNUPE)	51%	53%	60%	75%
Feed efficiency [c] (J animal product per J feed)	3.3%	3.6%	4.3%	5.7%
Recycling quota of animal manure in confinement (share)	61%	64%	76%	90%

[a]: Values for 2010 are already scenario parameters. [b]: Some countries do not reach the limit of 2750 kcal per capita per day or the livestock share of 15% in the economic conditions of SSP2. [c]: Average over all livestock products and regions weighted with the livestock production of 2010.

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Supplementary Information is linked to the online version of the paper at www. …

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Chapter VIII

Synthesis and outlook

Benjamin Leon Bodirsky

Content of chapter VIII: Synthesis and outlook

1 Overview

The first section of this chapter answers the research questions that were set out in chapter I and thereby summarizes the key findings of this dissertation. The second section proposes how the existing model could be extended into an integrated model of agriculture, nitrogen and human welfare. The last section finally gives a brief outlook on how the agricultural sector may change when the welfare aspects of Nr pollution were considered.

2 Key findings

2.1 What are plausible scenarios for global food demand in the 21st century?

The rising demand for food through population growth and more affluent diets has been in the past the major driver of land use change (Huber et al. 2014) as well as of nitrogen pollution (Neher 2013). Out of this reason, an elaborated representation of food demand in agricultural long-term models is of crucial importance.

Food demand can be expected to rise strongly until at least the middle of the 21st century.

In chapter II we developed a slim and transparent model to forecast global food demand. While this food demand model is tailored for the requirements of the MAgPIE model, it was designed in a way that method and data can be used easily also by other research groups. Next to total caloric consumption, it also projects the share of animal calories in the diet. Moreover, it is the first study that estimates the future development of household waste on a global scale.

The results of all demand projections indicate that food demand can be expected to rise strongly until at least the middle of the 21st century. This implies also a large increase in food waste, as the food purchased by households exceeds the plausible intake levels already today. At least for the first half of the century, also the demand for livestock products strongly rises. Our food demand projections also indicate that while undernourishment decreases in both absolute and relative terms in the coming decades, it will remain a major problem, especially in some African countries.

In chapter III, our modelling approach for forecasting food demand was compared with other approaches. Most other models were more complex, taking into account regional income- , price- and cross-price-elasticities. However, of all 10 demand models listed in chapter III, only our food demand model reproduced the trend of a falling share of animal-based products that can be observed in developed regions (FAOSTAT 2013). Also, our analysis found strong and significant non-income related trends in all regression models, which indicate that some drivers or processes important for long-term projections might not be included in standard demand models. Apart from striking differences in regard to the share of animal products in the diets, the models come to rather similar results for the long-term projections. At the same time, the model comparison revealed also the short-comings in our approach: In contrast to other models, our model does not explicitly include prices as independent variable. Therefore, it cannot simulate the effect of supply shocks (e.g.

climate impacts, taxes, shifting trade regimes) on demand quantities. The inelasticity of demand at the same time overestimates the effect these shocks have on MAgPIE's shadow prices, as a reduction of demand does not take pressure out of the system.

Each model having its advantages and disadvantages, it is therefore important to sustain the plurality of modelling approaches, and to even think of innovative new modelling approaches. Such an approach is presented for instance by Prajal et al (2012), who cluster global food demand into diet archetypes, and analyse how countries switch from one archetype to another over time. They observe clear path-dependencies in these transformation, with some paths having a more sustainable and healthy diet than others.

Model comparisons that include such different models can be full of insights, whereby the design of such inter-comparison studies is of importance. Firstly, higher transparency is required, especially in regard to parameterisation and calibration, as many models are little documented. Secondly, participating model developers should resist and actively counteract a "race to the middle", in which divergent estimates are corrected or "improved" to meet the mainstream result. Finding the processes which lead to such deviations might be more insightful than a joint best-guess projection. One step to achieve this could be to prohibit corrections after result-submission, or to provide also the history of results, describing the learning process and the changes made.

2.2 What is the current state of the global agricultural nitrogen cycle?

As the current knowledge on the global nitrogen cycle is still dominated by large uncertainties, chapters IV and VII provide a new set of estimates for the major Nr flows, using for most estimates an independent methodological approach or a different parameterization than previous studies.

The human influence on the global nitrogen cycle is far more profound than previously thought.

Assembling a comprehensive overview over the agricultural Nr flows, this study estimates that current Nr losses of agriculture alone are 185 Tg Nr in the year 2010 (see SI of Chapter VII). Adding 40 Tg Nr from combustion and 20 Tg Nr for industrial use (Fowler et al. 2013), total anthropogenic Nr losses of 245 Tg Nr by far exceed the 140 Tg Nr estimated as the human influence by Rockström et al (2009). At the same time, estimates for pre-industrial terrestrial Nr sources were recently corrected downwards to 58 (40-100) Tg Nr (Vitousek et al. 2013), much less than the previously estimated 128 Tg Nr (Galloway et al. 2004). Hence, the human influence on the global nitrogen cycle is far more profound than previously thought.

Chapter IV did not only newly quantify the major Nr flows, but also assembled a literature overview over other estimates for these flows. Based on this research, the data quality of various flows can be categorized as follows:

Good data quality exists for estimates of Nr flows that are listed in public statistics. Errors of +- 10% are reported for the consumption of inorganic fertilizers (Fowler et al. 2013). However, as was pointed out in chapter IV, estimates of the two major databases FAOSTAT and IFADATA diverge

often drastically, indicating that uncertainty might actually be higher. Also the estimates for protein consumption of households can be regarded to be of relatively high data quality, even though Smil also reports inconsistencies in FAOSTAT protein estimates (Smil 2000). Of similar quality are the estimates for Nr in crop production and aboveground crop residues, the biological fixation of leguminous crops, as well as the excretion of manure by animals. These Nr flows were estimated by various studies, coming to similar results while using different methods (e.g. bottom-up and top-down) and parameterizations (e.g. independent sources for Nr contents of harvested crops).

Medium quality can be attributed to the estimates for crop residue use, manure recycling and manure losses. Even though these Nr flows were estimated by a broad range of studies, these studies often use the same parameters of expert-guess quality from Smil (1999), IPCC (1996) and Eggleston (2006). Also, the estimates for crop processing and distribution are rather uncertain. The data quality of the Food Balance Sheets which is used for these estimates is of lower quality than the FAO production statistics (Smil 2000), and the complex network of processing steps complicates a clean separation of flows. Finally, all Nr loss estimates have to be considered also of medium quality. So far, global estimates of losses were only executed top-down, estimating Nr losses based on Nr inputs like inorganic fertilizers, instead of measuring nutrient losses bottom up, e.g. upscaling from nutrients measured in rivers. The quality of top-down estimates depends strongly on the comprehensiveness of considered Nr flows. E.g. not considering Nr released by soil organic matter loss automatically leads to lower losses. Finally, the emission parameters splitting Nr losses into leached, volatilized, and denitrified Nr are very uncertain (Eggleston et al. 2006; Smil 1999), especially in regard to denitrification (See supplementary online material of Bouwman et al. 2013). For N_2O emissions, chapter IV showed that estimates for global N_2O emissions may vary by +-25% (90% confidence), taking only in account the uncertainty of emission parameters and not the uncertainty of the underlying Nr inputs.

Poor data quality has to be attested firstly to the estimates of Nr fixation by free-living Nr-fixing microorganisms in cropland soils and biological fixation on savannahs, as current estimates lack empirical foundation (Herridge, Peoples, and Boddey 2008). Secondly, Nr flows connected to the cultivation of fodder crops have to be considered also as highly uncertain, as the area and production of fodder crops have deliberately not been published by FAOSTAT (2013), even though they have been collected. As fodder crops may make up one fifth of global Nr harvest (chapter IV), reliable data is strongly required. Thirdly, Nr in grazed biomass is very uncertain. It was so far only estimated top-down, calculating grazed biomass as the gap between animal feeding requirements and available feed stock. Especially for South-Asia, this approach leads to results that are little plausible (Wirsenius 2000). Fourthly, the changes of soil Nr pools are neglected by most inventories, even though the Nr stored in soils could be as much as 95000 Tg (Vitousek and Matson 1993). The similar estimates of our study and of Vitousek and Matson (1993) both indicate that the loss of soil organic matter after land conversion may be a major source of Nr and contribute more Nr to the agricultural nitrogen cycle than leguminous Nr fixation. Soil organic matter loss may also be the missing piece of the jigsaw to explain the low Nr fertilizer use in Sub-Saharan Africa. Similar estimates of losses from wetland drainage as well as changes of Nr stocks in continuously managed soils are yet still little investigated.

2.3 How will the nitrogen cycle evolve into the future?

To estimate scenarios of the future global agricultural nitrogen cycle, this study proceeds in two steps. Firstly, scenarios are formulated for the general development of the agricultural sector which drives the agricultural nitrogen cycle. Secondly, the interactions between the agricultural sector and the nitrogen sector are modelled.

The general dynamics of the agricultural sector are simulated with the land-use model MAgPIE. The early research done with the MAgPIE model was based on a single, best guess baseline scenario (Lotze-Campen et al. 2008; Lotze-Campen et al. 2009; Popp, Lotze-Campen, and Bodirsky 2010; Popp et al. 2011). To provide instead a larger range of plausible scenarios, the MAgPIE model was extended by different sets of model drivers like population, per-capita income, technological progress in the livestock sector, fertilizing efficiency, as well as forest and trade policies. In chapter IV, four scenarios are parameterized according to the storylines of the Special Report on Emission Scenarios (SRES) (Nakicenovic et al. 2000), using amongst other drivers the food demand projections from chapter II that are created consistently based on the SRES storylines. The scenario framework provided by this dissertation can now easily be updated to represent new storylines in MAgPIE, as it was shown for the Shared Socio-Economics Pathways (SSPs) (Kriegler et al. 2012) that are used in chapters III and VII.

The business-as-usual scenarios simulated in this study show a broad range of possible outcomes for the global agricultural nitrogen cycle.

In the coming decades Nr pollution rises in all scenarios. Pollution in the end of the 21st century may return to current levels or more than double depending on the scenario.

The interactions between the agricultural sector and the nitrogen cycle are modelled with a newly designed mass balance model for Nr flows. Nr flows are linked to numerous dynamic parameters in the model, for example translating agricultural production into nutrient withdrawals, or accounting for the alteration of manure excretion when livestock feed mixes change. Furthermore, the model also captures the feedback of biophysical flows on agricultural activity, for example simulating the fertilization activity where organic and inorganic fertilizers can be substituted. The model is unique in the respect that it connects gapless all Nr flows within the agricultural nitrogen cycle, from the fixation of Nr through its various recycling paths up to its release to the environment. Therewith the approach guarantees an overall consistency of the simulated flows at any time.

The four business-as-usual scenarios presented in chapter IV show very heterogeneous developments of the future agricultural nitrogen cycle. The levels of Nr losses into the environment however rise in all scenarios at least until the middle of the century. In the end of the 21st century, Nr losses may return to current levels or rise up to more than twofold its current state, with N_2O emissions rising to more than the threefold. Given the severe impacts that Nr pollution has already today (Sutton, Howard, et al. 2013; Brink et al. 2011), the expectable further increase of pollution reinforces the need for strong mitigation action.

The resulting projections are in line with past trends and the range of estimates compares well to various other studies (chapter IV, Winiwarter et al 2013). Still, our estimates tend to project a stronger rise of the nitrogen cycle than most other estimates, even though efficiency improvements are considered in all baseline scenarios. Reasons for these higher estimates are the following: (a) we consider the dietary transition of developing countries in all scenarios, (b) the adoption of intensified livestock production increases the pressure on the cropland sector in our model compared to other models (c) the closed budget approach of MAgPIE helps to circumvent infeasibly low Nr inputs.

In a sensitivity analysis we showed that the uncertainty of emission parameters may alter the estimates of N_2O emissions by +/- 25%. However, there are good reasons that the actual spread of these emission scenarios should be even larger (both upwards and downwards), as the IPCC emission factors (Eggleston et al. 2006) cannot be regarded as constant over time. Being linked to Nr inputs instead of losses, these emission factors underestimate the emission-savings of improved fertilizing efficiency by approximately factor 2[1]. This leads to an overestimation of N_2O emission in low-emission scenarios. In high-emission scenarios, however, emission factors might be too low as there is a positive feedback between global warming and the emission parameters which are temperature and moisture-dependent (Sutton, Reis, et al. 2013). A 5° warming of the climate may lead to an increase of NH3 emission factors by one third to two thirds (Sutton, Reis, et al. 2013).

2.4 Which additional pressure on the nitrogen cycle might evolve from the production of 2nd generation bioenergy?

In is still highly uncertain which role bioenergy in general, as well as specific types of bioenergy carriers (e.g. residues, manure, conventional or lignocellulosic crops) may play in the future energy mix (IPCC 2012).

Within potential bioenergy carriers, 2nd generation bioenergy crops (most importantly fast growing grasses like miscanthus and trees like poplar) have promising attributes both in regard to potential energy provision and environmental sustainability. Compared to 1st generation bioenergy crops like soybeans, maize or sugarcane, 2nd generation crops have higher energy yields. At the same time, they require less Nr fertilization as nutrients are drawn into the roots in the harvest period, leading to low Nr withdrawals by the harvested biomass. (Röhricht 2008; Bhattacharya et al. 2003)

Bioenergy from lignocellulosic grasses is comparably nutrient efficient. Still, its cultivation on large scales can counteract low Nr pollution targets.

In some high-bioenergy scenarios, 2nd generation bioenergy crops are projected to play a major role, replacing scarce mineral oil for transportation in the 2nd half of the 21st century and providing a climate change mitigation option, especially when

[1] While Eggleston et al. (2006) link emission factors to Nr inputs (e.g. inorganic fertilizers), the quantity of emissions actually depend on the Nr losses (the amount of Nr not withdrawn by the crop). As any improvement of Nr uptake efficiency only reduces losses while not reducing crop withdrawal, Nr losses are stronger reduced than Nr inputs. Assuming a global Nr uptake efficiency of 50%, and N2O emissions being linearly related to Nr losses, this leads to a bias of approximately factor 2. This bias does not affect the results of the low-pollution scenarios of chapter VII, as this analysis focusses exclusively on Nr losses and not on N2O emissions.

combined with carbon capture and sequestration (Klein et al. 2013). According to Neher (2013), the changes of the nitrogen cycle after 2050 may be shaped predominantly by bioenergy demand, as population growth stagnates or reverses, and diets saturate.

According to the analysis of chapter V, the production of 100 EJ would require 15-20 Tg of additional fixed Nr, or about 10 Tg Nr under very efficient nutrient management. Depending on the deployment of bioenergy (scenarios ranging from less than 100 to more than 400 EJ (Berndes, Hoogwijk, and van den Broek 2003)), pressure might be substantial. As bioenergy is often used especially in sustainable scenario, it may counteract low Nr pollution outcomes (Winiwarter et al. 2013). Taking the example of the low-pollution scenario of chapter VII, 300 EJ of efficiently fertilized bioenergy (30 Tg) would increase the Nr requirements of agriculture (86 Tg) by about 35%. While Nr mitigation and climate change mitigation usually go hand in hand, bioenergy production is an example of a trade-off between both sustainability targets.

The appropriate parameterization of 2nd generation bioenergy crops is still subject to uncertainty. Our estimates of nutrient requirements from chapter V take only into account the nutrient removals by harvested biomass. Newer versions of MAgPIE (e.g. Klein et al. 2013) also account that Nr has to be re-fertilized which is lost when crop residues decay on the field. However, the used methodology (IPCC 2006) was created for crops with yearly turnover, assuming that all residues decay within one year. Miscanthus and Poplar are however perennial plants; even though plant parts like fine roots and leaves also have a frequent turnover, certain plant parts can keep the nutrients for more than one year, and should therefore have lower Nr losses and fertilizer requirements.

2.5 How would the adoption of nitrogen mitigation measures change the nitrogen cycle and Nr pollution?

Projections for the nitrogen cycle so far mainly concentrated on baseline scenarios (Tilman et al. 2001; Tubiello and Fischer 2007; Erisman et al. 2008; Bouwman, Beusen, and Billen 2009; Davidson 2009; Winiwarter et al. 2013). Global mitigation scenarios specifically concerned for nitrogen pollution were focused on the livestock sector (Pelletier and Tyedmers 2010; Bouwman et al. 2011), or on simple, non-model-based estimates (Sutton, Howard, et al. 2013; de Vries et al. 2013). Chapter VII presents the so far first low-target mitigation scenarios for the agricultural nitrogen cycle that cover mitigation measures along the whole agricultural supply chain.

The effects of demand-side mitigation can be simulated comprehensively with the MAgPIE model, as MAgPIE captures complex dynamics in the food supply chain like changed trade patterns or the substitution of manure against fertilizer when the livestock sector is reduced. In this dissertation, different scenarios of demand-side mitigation are presented in chapters VI and VII. Chapter VI focusses on a reduction of the share of animal-based calories in the diet, and assumes that consumption is reduced by 25% per decade in all regions. In contrast, chapter VII has a less ambitious scenario, where livestock consumption is only reduced in developed countries, and only up to a certain threshold (15% of calories) to guarantee developing countries the right to diversify their diet. However, chapter VII amends reduced livestock consumption also by a scenario of household waste reduction. While reduced livestock consumption and diminished household waste have probably the highest demand-side mitigation potential, there are also a number of other demand-side mitigation measures which were not regarded by this thesis, e.g. a shift of

consumption within animal products from cattle to chicken (Wirsenius, Azar, and Berndes 2010), the boycott of products from inefficient production regions (Bellarby et al. 2013), or the shift towards organic and locally produced products (Billen, Garnier, and Lassaletta 2013). As a reduction of livestock consumption also reduces the mitigation potential from these mitigation measures, the negligence of these measures should however have no large influence at the minimum Nr requirements estimated in chapter VII.

Ambitious mitigation measures could strongly reduce Nr pollution. However, a return of the nitrogen cycle into Holocene conditions is out of reach for the coming decades.

To integrate diverse supply-side mitigation measures into the model, chapters VI and VII provide two different approaches. Chapter VI estimates supply side mitigation based on a bottom-up marginal abatement cost-curve (Lucas et al. 2007), which has the advantage that mitigation action can also be connected to costs. At the same time, this approach has the disadvantages of a lacking representation of the biophysical changes within the model, the negligence of other Nr pollutants beside N_2O, as well as very poor data quality (Bodirsky et al. 2009). In contrast, chapter VII uses a systemic approach, where the efficiency of whole systems (like fertilization efficiency or feeding efficiency) is regarded without making explicit which technologies lead to these efficiencies. This approach has the advantage that it can be integrated consistently into MAgPIEs biophysical Nr module such that indirect up- and downstream effects of mitigation measures are considered.

Both approaches show that a strong reduction of pollution would be possible with a combination of demand and supply-side measures, indicating that pollution level could be approximately halved. However, it is also stated that a focus of mitigation action on only one sector is insufficient to reduce emissions below current levels.

Despite the large mitigation potential, the results of chapter VII also show that Nr pollution will remain far beyond Holocene levels even under very optimistic assumptions for technological advances and consumption behaviour. This can be explained on the one hand by the continuous population growth and the diet improvements in the least developed countries, two drivers of the nitrogen cycle that should not be impeded by Nr mitigation activity. On the other hand, mitigation costs of non-point sources of pollution like field Nr losses are high and increase over-proportionally with increasing efficiency. Moreover, maximum conversion efficiencies exist for many processes (e.g. for the transformation of natural gas to reactive Nr in the ammonia synthesis, for the transformation of plant to animal proteins, or for the harvest index).

Therefore, agriculture cannot feed 9 billion people without causing considerable Nr pollution that harms people and disturbs the Earth system.

3 Towards an integrated model of agriculture, nitrogen and human welfare

A vision for future research is the development of an integrated model of agriculture, nitrogen and human welfare, as described in Figure 1. Such a model could be used to evaluate the welfare effects of different human behaviour, taking into account its complex biophysical consequences.
A large part of such a model is already represented by the current version of MAgPIE: The consumption and production of agricultural products, the pollutants (such as N_2O) that emerge out of agricultural activity, as well as the costs and benefits connected to agricultural production. As will be discussed in the following, missing or incomplete parts include most importantly the costs or co-benefits of mitigation or adaptation measures (Figure 1, 1), the estimation of impacts of Nr pollution (Figure 1, 2), and a conceptual approach to the representation of welfare (Figure 1, 3). The integration of all these aspects into one model could be of large scientific value.

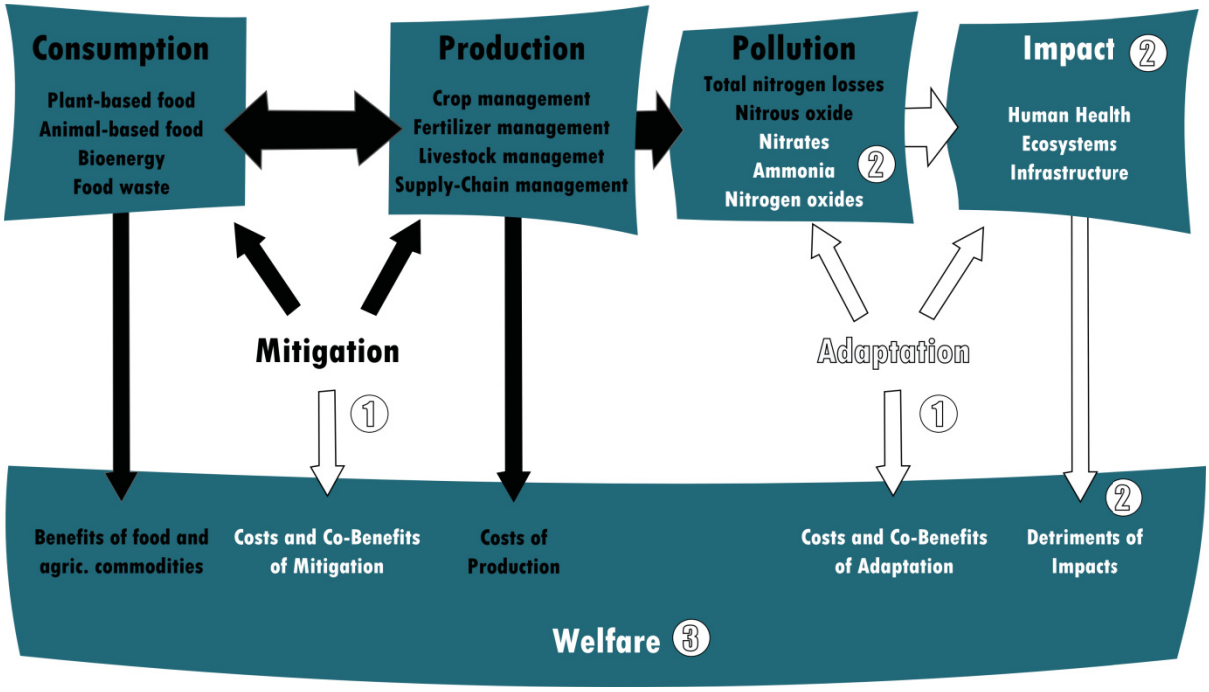

Figure 1: A possible framework for an integrated model of agriculture, nitrogen and human welfare. Loosely adapted from Brink et al. (2011). Black objects are already part of MAgPIE; white objects indicate proposed future model developments.

3.1 Mitigation and adaptation: Linking change to costs

In chapters VI and VII of this thesis, we already investigated how a number of key mitigation measures targeting agricultural consumption and production could alter Nr pollution. However the link connecting these measures to mitigation costs is so far not represented sufficiently in the model.

On the supply side, central parameters like the fertilizer efficiency or like the use of animal waste management systems could be made price-elastic. This would allow to estimate how agricultural production practices change once Nr pollution is priced. However, the available data on mitigation costs is unfortunately of poor quality (Bodirsky et al. 2009): Available studies on mitigation costs mostly cover low-cost options which are fully exploited even in less ambitious mitigation scenarios. Almost all the literature is based on the same, few, methodologically-poor back on the envelope

calculations and expert estimates. Often they are generalized from European examples to global estimates or from few current practices to long-term potentials.

On the demand side, central parameters of food demand like total calories, the share of animal calories, or the composition of feed baskets could be made price-elastic. Taking into account Nr pollution as additional cost factor would then alter food demand patterns and consumer surplus. However, the available short-term elasticities (USDA 2009) would probably overestimate the impact of a reduced consumption on consumer surplus, as food demand is strongly shaped by habits that may change over longer time-spans, and by preferences that jump from one state into another (Allais and Nichèle 2007). Rozin et al. (1997) argue that some preference shifts, like becoming vegetarian, can turn into durable values if they are connected to a moralization process, and that these values can even be transmitted cross generations. While a dietary change may be connected to welfare losses in the short-term, a new diet may not have additional costs once it is established. As MAgPIE operates on long time-scales, it should consider such effects.

Additional to the analysed mitigation measures, the impact of adaptation measures should be evaluated. Adaptation could either remove existing pollutants, e.g. by establishing wetlands to denitrify Nr in aquatic systems (Jansson et al. 1994), or it could reduce the impacts of pollutants, e.g. by moving emission sources of short ranged air pollutants like intensive piggeries away from densely populated areas (Sutton, Howard, et al. 2013). The latter adaptation activity can be easily implemented into MAgPIE, as the spatial allocation of agricultural activity is one of the core dynamics in MAgPIE.

There are numerous environmental and social co-benefits of supply and demand-side mitigation as well as adaptation. For example, lower consumption of livestock products and a reduction of food waste reduce greenhouse gas emissions, improve public health, diminish the pressure on deforestation, soil degradation and biodiversity loss, and decrease phosphorus as well as water use (Steinfeld et al. 2006; Stehfest et al. 2009; Wirsenius 2000; Dwyer 1988). Precise nitrogen fertilization can reduce costs and goes hand in hand with better dosing and lower environmental footprint of other nutrients (e.g. phosphorus) and agrochemicals. Improved livestock diets also decrease methane emissions, the number of slaughtered animals, and further externalities related to feedstock cultivation (Wirsenius 2000). MAgPIE can provide an excellent framework for the simulation of co-benefits, as it represents already a broad range of environmental indicators next to nitrogen, for example water consumption, phosphorus fertilization, greenhouse gas emissions or deforestation. As these co-benefits reduce the costs of mitigation, their inclusion can change the evaluation of the mitigation costs and the intended mitigation level substantially.

3.2 Pollution impacts: Linking nitrogen pollution to human detriment

To estimate the detriments Nr pollution has on humans, MAgPIE would need to be extended in three steps:

In a first step, environmental pollution has to be modelled in more detail. Currently, MAgPIE already estimates total Nr losses and N_2O emissions, while nitrate leaching and volatilisation of NO_x and NH_3 are only represented very simplistic according to the IPCC emission factors (Eggleston et al. 2006). Bouwman et al. (2013) offer a good example on how Nr pollutants can be represented in detail by a global model. As the pollutants NO_x, NH_3, and leached nitrates operate on a local scale, their spatial

explicit modelling is of importance to estimate pollution impacts. MAgPIE's spatial resolution, simulating agricultural production on clustered 0.5° cells (Dietrich, Popp, and Lotze-Campen 2013), provides a suitable framework for this.

In a second step, Nr pollution has to be connected to physical impacts, e.g. by linking ozone pollution to crop yield damages, or by connecting NO_x air pollution to disability-adjusted life years (DALYs) lost. First estimates of such kind were made in the recent past on a global scale (Van Grinsven et al. 2013; Brink et al. 2011; Sutton, Howard, et al. 2013), but only refer to the present state.

In a third step, a value has to be attributed to the physical impacts. Impacts on agricultural production like yield-damages could be endogenously considered within the MAgPIE model. The valuation of non-marketed goods (e.g. public health or recreational value of intact ecosystems) is more complex as there is no market price. Estimates have to be based instead on indirect approaches, using methods like the travel-distance approach or surveys on the willingness to pay for environmental goods (Van Grinsven et al. 2013; Brink et al. 2011; Sutton, Howard, et al. 2013). However, as will be pointed out in the following section, valuing Nr impacts only according to (approximated) market values might be problematic.

3.3 Welfare: Aggregating social welfare

Estimating social welfare requires the definition of a social welfare function that aggregates welfare between individuals. An often used approach is to define social welfare as the sum of consumer surplus (the difference between consumers' willingness to pay and the market price) and producer surplus (the difference between the market price and production costs). In principle, the MAgPIE model can be used to analyse economic surpluses (Stevanovic et al. 2013).

While maximizing the sum of economic surpluses simulates a pareto-efficient market outcome, it cannot be shown that this outcome is optimal from a social welfare perspective. Taking into account most importantly the obstacle that individuals preferences cannot be compared ordinally, it can be shown that such a welfare function cannot be generated based on the individuals preferences. Any social welfare function is therefore a subjective assumption. (Deaton 1980; Arrow 1988).

As the economic surpluses are estimated based on the willingness to pay and sell, preferences enter the aggregation weighted with the purchase power of the respective individual. Especially in a global society where the initial wealth distribution ranges over orders of magnitude (WORLDBANK 2013), it is highly questionable whether economic surplus optimization comes close to social welfare optimization. For example, a world without hunger could have a lower economic surplus than a world with hunger but with high bioenergy consumption in the developed world. Similarly, if health impacts are valued according to the willingness and capability to pay for a healthy life, economic surplus maximisation would lead to a shift of polluting production from rich countries to poor countries, improving surpluses while eventually increasing casualties. Economic surplus is therefore a questionable indicator for policy recommendations in unequal societies.

Besides the concerns about the inter-personal aggregation of surpluses in unequal societies, one may also raise the paternalistic argument that the consumer surplus does not necessarily reflect the welfare consumers gain from products; in other words that a consumers expressed preferences do not reflect his actual welfare (Verbeke et al. 2007; Tirole 2002; Thaler and Sunnstein 2008). For example, when subjective emotional well-being and overall life evaluation are estimated, obese

persons perform significantly worse (Kahneman and Deaton 2010), even though their financial budget would probably allow for a healthy diet.

Alternative (yet no less subjective) welfare indicators have been developed, for instance the Human Development Index (McGillivray 1991) or the index of Subjective Well-Being (Diener 2000). Yet there are only few complex models that take make exemplary use of such indicators (e.g. Llavador, Roemer, and Silvestre 2011). As such indicators would allow for a more differentiated debate about desired future developments, their inclusion into MAgPIE or even into the REMIND-MAgPIE framework (see chapter I) should be an aim for future model-development.

In such a framework, MAgPIE could provide a number of welfare-relevant environmental indicators, but also MAgPIE's economic parameters are highly relevant social indicators. As 70% of the poor world population live in rural areas, agriculture is their main source of income (WORLDBANK 2013). At the same time, the purchase of food is still the main post of expenditure for the poorest households (NSS 2008).

The macro-economic and energy model REMIND could complement these indicators by parameters for general wealth distribution, education or leisure time (parameters that are not yet represented in this model, but are closely connected to modelled parameters). As one of the strengths of REMIND is the inter-temporal foresight, also question of distribution between generations could be discussed.

4 The hope for the next agricultural revolution

This thesis showed that population growth, opulent diets and inefficient agricultural production systems have driven the agricultural nitrogen cycle out of the stable conditions of the Holocene, and that Nr pollution can be expected to rise to unprecedented levels within the 21st century. The thesis also showed that while a return into Holocene conditions is out of reach, an ambitious abatement of Nr pollution is possible. However, this requires a thorough restructuring of the agricultural market. In contrast to its current state, agriculture would be characterized by efficient production and supply chains, by more sustainable consumption, and by lower pollution. This transition would be no less than a new agricultural revolution, comparable to the green revolution in the last century that boosted agricultural production but also environmental impacts.

While such an efficiency revolution sounds utopian at first glance, there is a solid economic basis for such a transition. Damage caused by agricultural reactive Nr pollution is in the same order of magnitude as the agricultural value added: In Europe alone, Nr causes damages in the order of 0.4-2.7% of European GDP (van Grinsven 2013). About half of these damages stem from agriculture, while the agricultural value added of the European Union in 2008 is 1.6% of the GDP (Worldbank, 2013). Similarily, global costs caused by Nr pollution are estimated to be 200 to 2000 billion USD (Sutton, 2013), compared to the global agricultural value added of 1700 billion USD (Worldbank, 2013).

It is evident that Nr pollution is substantially under-regulated. Even in Europe, the world region with the most progressive Nr regulations, mitigation costs are often by orders of magnitude lower than pollution costs (Holland et al. 2011). However, Nr pollution is in principle a type of externality that could be regulated comparatively well. In contrast to climate change, ozone depletion, or ocean

acidification, the impacts mostly occur close to the polluter (Dragosits et al. 2002), which provides incentives for regulations on local, sub-national or national level. At the same time, the large environmental and social co-benefits of Nr mitigation measures could work as door-opener to the regulation of environmental problems like climate change that are suffering from the free-rider problematic.

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Statement of Contribution

Introduction and conclusion were written by Benjamin Leon Bodirsky.
The structure and content of the dissertation were discussed with Prof Dr Ottmar Edenhofer, Dr Hermann Lotze-Campen and Dr Alexander Popp.

Chapter 2: Food demand projections for the 21st century
State: Under review in "Food Security"

Susanne Rolinski and Benjamin L. Bodirsky designed the study together and are main authors at equal shares. Susanne Rolinski performed the regression analysis. Benjamin L. Bodirsky collected data and applied the regressions to the SRES storylines. Anne Biewald collected and processed data for the comparison section and wrote the respective parts of the article. Hermann Lotze-Campen and Alexander Popp discussed the study design and the set-up of the article. All authors discussed the results and commented on the manuscript.

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Chapter 3: The future of food demand: understanding differences in global economic models
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Hugo Valin is corresponding author and coordinated the research. Hugo Valin, Ronald D. Sands, Dominque van der Mensbrugghe and Gerald D. Nelson are the main authors of the study.
Hugo Valin, Ronald D. Sands, Dominique van der Mensbrugghe, Gerald D. Nelson, Helal Ahammad, Elodie Blanc, Benjamin L. Bodirsky, Shinichiro Fujimori, Tomoko Hasegawa, Petr Havlík, Edwina Heyhoe, Page Kyle, Daniel Mason-D'Croz, Sergey Paltsev, Susanne Rolinski, Andrzej Tabeau, Hans van Meijl, Martin von Lampe and Dirk Willenbockel contributed model results, explained their methodology, discussed the results and commented on the manuscript.

Chapter 4: N2O emissions from the global agricultural nitrogen cycle – current state and future scenarios
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Benjamin L. Bodirsky is the main author of this study and wrote the manuscript, with important contributions from Alexander Popp. Benjamin L. Bodirsky, Hermann Lotze-Campen, and Alexander Popp designed the study and analysed data. Benjamin L. Bodirsky extended MAgPIE by Nr flows and Nr budgets, crop residues, conversion by-products. Susanne Rolinski and Benjamin L. Bodirsky provided food demand scenarios. Hermann Lotze-Campen, Jan P. Dietrich, Christoph Schmitz, Isabelle Weindl, Alexander Popp, Lena Scheiffele, Benjamin L. Bodirsky developed and improved MAgPIE.

Chapter 5: On sustainability of bioenergy production: Integrating co-emissions from agricultural intensification
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Alexander Popp is the main author of this study, created the study design with Hermann Lotze-Campen and wrote the manuscript. Hermann Lotze-Campen, Alexander Popp, Benjamin L. Bodirsky developed and improved MAgPIE. Marian Leimbach, Brigitte Knopf, Nico Bauer provided the bioenergy scenarios. Tim Beringer provided LPJml results for bioenergy crops. Benjamin L. Bodirsky extended MAgPIE by GHG emissions as well as nitrogen fertilizer use, and helped to write the respective parts of the article. All authors discussed the results and commented on the manuscript.

Chapter 6: Food consumption, diet shifts and associated non-CO2 greenhouse gases from agricultural production
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Alexander Popp designed the study, made the model-runs, and wrote the article. Hermann Lotze-Campen and Alexander Popp developed the MAgPIE model. Benjamin L. Bodirsky calculated the emission factors and implemented emissions into MAgPIE. All authors discussed the results and commented on the manuscript.

Chapter 7: Stuck in the Anthropocene: The case of reactive nitrogen.
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Benjamin L. Bodirsky is the main author of this study. Benjamin L. Bodirsky, Alexander Popp designed the study and analysed data. Benjamin L. Bodirsky wrote the paper, with important contributions from Alexander Popp, Christoph Müller, Hermann Lotze-Campen. Benjamin L. Bodirsky, Hermann Lotze-Campen, Jan P. Dietrich, Christoph Schmitz, Isabelle Weindl, Alexander Popp, Markus Bonsch, Florian Humpenöder, Miodrag Stevanovic, Susanne Rolinski, Anne Biewald developed and improved the model. All authors discussed the results and commented on the manuscript.

Tools and Ressources

The following tools were used for the modelling exercise and typesetting:

- GAMS (Brooke et al. 1988), using the CONOPT solver (Drud, 1994)
- R (R 2013), including most importantly the "Landuse library" developed at Potsdam Institute of Climate Impact Research (Dietrich et al. 2013)
- Latex (Lamport, 1994).
- Open Office
- Inkscape
- Microsoft Office
- Zotero

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