Title

Agroecological measures and circular economy strategies to ensure sufficient nitrogen for sustainable farming

Authors

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Highlights

- Organic farming has low impact per unit area but is limited in nitrogen
- Dietary changes are important but insufficient to overcome nitrogen limitations
- Organic farming needs agroecological and circular economy improvements to be feasible globally
- The performance of organic and conventional farming is similar if fully optimized

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Abstract

Sustainable food systems face trade-offs between demands of low environmental pressures per unit area and requirements of increasing production. Organic farming has lower yields than conventional agriculture and requires the introduction of nitrogen (N) fixing legumes in crop rotations. Here we perform an integrated assessment of the feasibility of future food systems in terms of land and N availability and the potential for reducing greenhouse gas (GHG) emissions. Results show that switching to 100% organic farming without additional measures results in N deficiency. Dietary change towards a reduced share of animal products can aggravate N limitations, which can be overcome through the implementation of a combination of agroecological, circular economy and decarbonization strategies. These measures help to recycle and transfer N from grassland. A vegan diet from fully decarbonized conventional production performs similarly as the optimized organic scenario. Sustainable food systems hence require measures beyond the agricultural sector.

Keywords: Nutrient cycling; Food systems; Organic farming; Human diets.

1. Introduction

By 2050, close to 10 billion people will require access to healthy and sustainably produced food (Godfray et al., 2010; Willett et al., 2019). The global food system currently is responsible for ~25% of global greenhouse gas (GHG) emissions (IPCC, 2014; IPCC, 2019). Already now, nitrogen (N) and phosphorus use exceed planetary boundaries by a factor of two (Steffen et al., 2015). These environmental impacts of food production could aggravate further, if global demand for cropland and animal products continue to increase (O'Neill et al., 2018). It is estimated that GHG emissions, synthetic N fertilizer and mineral phosphorus fertilizer (which is a non-renewable resource, and its price has increasing following depletion and scarcity of phosphate rock) use related to the food system could increase by 50–90% in the absence of simultaneous developments of technological changes and other mitigation measures (Springmann et al., 2018), illustrating the massive challenges to improve the sustainability of food systems. A sustainable food system allows all humans a nutritionally adequate diet, sufficient availability of fertile land and nutrients, and ensures that future environmental detriments such as deforestation, biodiversity loss and GHG emissions are kept within planetary boundaries related to land demand and the biogeochemical cycle of phosphorus and N (Erb et al., 2016; Steffen et al., 2015; Theurl et al., 2020).

Owing to systemic trade-offs in the land system, simultaneously achieving targets like GHG reduction, halting deforestation, minimizing biodiversity loss (Newbold et al., 2016; Pereira et al., 2013), and limiting the current N-surplus to levels compatible with the planet's safe operating space presents a daunting challenge. For instance, intensification of agriculture through closure of yield gaps has been found to be a promising strategy due to its area-sparing effect that allows for the restoration of natural habitats. It also reduces net emissions from agriculture and land-use change (Balmford et al., 2018; Lamb et al., 2016). However, intensification in most cases requires increasing amounts of nutrients, particularly reactive N. Expanded use of synthetic N fertilizers (Erisman et al., 2008; Lassaletta et al., 2014; Zhang et al., 2015) is associated with increased GHG emissions and contamination of water bodies (Bodirsky et al., 2014; Carlson et al., 2017; Galloway et al., 2008). Furthermore, intensification alone has not been regarded sufficient as a strategy to avoid cropland expansion. The increasing demand for food and an increasing share of animal products in diets (Bajželj et al., 2014) may require further production enhancement. Therefore, changes in human diet towards reduction of animal products, either in isolation or combined with a sustainable intensification of farming, have been described as viable ways to decrease GHG emissions and land demand (Foley et al., 2011; Mora et al., 2020; Poore and Nemecek, 2018; Tilman and Clark, 2014).

An alternative strategy for a sustainable food system is organic farming. It aims at reducing industrial inputs to agriculture, in particular in the form of synthetically produced pesticides and nitrogen. A keystone of organic farming is to ensure that N withdrawals from farmland is replenished from natural sources, by improving N cycling and including N-fixing legumes in crop rotations. However, crop yields per harvest event in organic agriculture are commonly found to be around 20% lower than in conventional farming systems (de Ponti et al., 2012; Seufert et al., 2012), albeit with large regional heterogeneity depending on the ecosystem, region and management (Barbieri et al., 2019; Li et al., 2020; Seufert et al., 2012). In agroforestry systems herbaceous legume crops or weeds can fix N(Nair, 2011). In rotational systems where land is currently under bare fallow, legume cover crops can be introduced in the rotation. However, for most systems, organic farming usually requires rotational systems with unproductive periods of N-fixing legume/fallow between harvest events (Barbieri et al., 2019; Muller et al., 2017). In those cases, due to the area needed to make reactive N available via legume cultivation, total system area yields over the whole crop rotation are markedly lower. Thus, while per-hectare GHG emissions are lower compared to conventional systems, and higher benefits for ecosystem (Smith et al., 2020) and human health (Reganold and Wachter, 2016; Smith-Spangler et al., 2012) can be expected, exceeding land demand might annihilate these advantages (Connor, 2013, 2008; Meier et al., 2015; Reganold and Wachter, 2016). Changes in diets are therefore deemed necessary as accompanying measures for organic farming.

There are few studies assessing the benefit-cost portfolio of organic farming, accompanied with changes in diets that consider the role of N in assessing and deciding between the alternative paths for a sustainable future. Muller et al. (2017) and Barbieri et al. (2021) used N balance of organic farming and showed there is a deficit of N in full conversion to organic farming. They also concluded that it is small and can be overcome with some additional measures to ensure adequate N-supply on croplands, such as improved legume management, increased N use efficiency (NUE) and recycling of nutrients from various organic wastes. Nevertheless, the effect of agroecological is still missing in the literature.

We here aim for comprehensively assessing various strategies for achieving a sustainable global food system in 2050, including organic farming. For this purpose, we combined a large range of variants of key components in the food system: area availability and demand, diets, yields in cropland and on grazing lands, and feed conversion ratios. We expanded the biophysical balance model BioBaM-GHG (Theurl et al., 2020) by comprehensively computing the N balance and GHG emissions. We explored the effects of strategies aimed at improving the N balance of organic farming, including (a) agroecological strategies for increasing N availability or efficiency of use,

(b) circular economy strategies, i.e. increased nutrient recovery and food waste reduction, and (c) decarbonization strategies through increased use of renewable energy.

2. Methods

In this paper, we extended the BioBaM modelling framework (Erb et al., 2016) to include nitrogen restriction in an integrated feasibility assessment of diets for 2050. In this section, we present an abridged description of the model. Then, we present the improvements included in the standard model, i.e. the carbon (C) and nitrogen (N) balance. With the C and N balances, we also calculated the direct greenhouse gases (GHG) emissions (i.e. emissions that occur on agricultural land). The indirect GHG emissions were considered using a life cycle assessment perspective based on Theurl et al. (2020). GHG emissions of synthetic N fertilizer production and agricultural operations thus include also indirect emissions. Finally, we studied potential new sources of N (namely to improve organic scenarios) and also innovations that could potentially reduce GHG emissions of diets (e.g. decarbonization of synthetic N fertilizer production process).

2.1. The BioBaM model

The BioBaM model, which served as basis of this work, was proposed by Erb et al. (2009) and has since been used in a large array of applications (Erb et al., 2012; Haberl et al., 2013, 2011, 2010). BioBaM is a biophysical model that calculates the balance between the supply and demand of biomass on a global scale, differentiating 11 regions and allowing for trade between regions. A scenario is feasible when global biomass demand is matched by supply by at least 95% (considering a 5% uncertainty range; cropland constraints). A scenario is unfeasible if for a given level of maximum cropland expansion the global biomass production (in C and N) does not match demand. There are two reasons why this may happen: in conventional production scenarios due to lack of area for food production, and in organic scenarios due to lack of area for food production or N fixation (after the inclusion of the N balance explained in section 2.2). In its current version BioBaM considers 11 primary commodities (listed in the Supplementary Table S1). All feasible scenarios in BioBaM must have zero-deforestation until the year 2050. This assumption is not a constraint built into the implementation of the model, but instead it is a constraint in the results. This means that scenarios where forestland decreases (measured as an increase in agricultural land) are calculated and discarded. The model was built using extensive and consistent databases on ecological and socioeconomic biomass flows and land use. The year 2000 was used as baseline and biophysical scenarios of the global agro-food system for

2050 were built based on 5 key parameters and according variants (human diet, origin of meat in human diets, livestock feed composition, crop yields and cropland area). The baseline year has no influence in the results for the year 2050, which can be evaluated without resorting to the baseline year (the year 2000 is only an optional comparison point). Our choice of the year 2000 for comparison is more demanding on future food systems. As forest land globally decreased between 2000 and 2020 and GHG emissions increased in the same period, if we had chosen a more recent baseline year more scenarios would be feasible but those would be insufficient to at least revert to 2000 conditions.

2.2. Model framework

In this paper we extended the BioBaM model with a carbon (C) and nitrogen (N) mass balance model that enabled the assessment of the C and N budgets of each diet and updated the calculation method of greenhouse gas (GHG) emissions in BioBaM (a graphical representation of the C and N balances can be found in the SI file – Section 1.1, Figure S1).

The C demand wa s fully based on the biomass balance performed in the previous version of the BioBaM model. Biomass was converted into C flows using a factor of 0.5 kg C/kg dry matter. For the N balance, we built a parallel N balance. Biomass demand was converted into N requirements (considering the N content of the crops and that a fraction of the N inputs is not taken up by the plants), and the N supply was calculated (details in the sections below).

For each scenario, we calculated the N balance, which is zero when demand and supply are exactly matched in a given region. There is an N surplus when supply exceeds demand and a deficit otherwise. Therefore, this extension of the BioBaM-GHG model considered both the role of N as a resource in the feasibility of diets and its role as a driver of environmental degradation (there is a surplus of N if the supply is higher than the actual N taken up in food products). In the model, we calculate both the N use and the N emissions to the environment (as an environmental impact). We considered all relevant N sources, namely mineral fertilizer, manure, biological N fixation (BNF) and atmospheric N deposition. N soil surplus was defined as all N that is used in croplands and grasslands but is not taken up by the plants or emitted as a GHG. Surplus N in ecosystems generates emissions to the atmosphere, soil and water (Zhang et al., 2015).

We specifically distinguish two production systems, conventional and organic production, which typically correspond to two very distinct sources of N fertilization. We assumed that the sources of N fertilization in “conventional” production are mineral fertilizer, manure from livestock production (in scenarios that include animal products) and atmospheric N deposition; in “organic” production, all N fertilization must be biological (manure, BNF) or atmospheric N deposition.

The baseline assessment was carried out for the most likely scenarios for the future evolution of each relevant variable, without implementing particularly disruptive/innovative strategies for the improvement for food systems, applied to 2050 in comparison to 2000. Regarding production systems, we used conventional production assuming 4 cropland yield estimates for 2050: (1) crop yields in 2050 equal to crop yields in 2000 (2) three different scenarios for crop yields' increase between 2000 and 2050 (considering that crop yields increase in 95% of the regions); and organic production, considering that crop yield is 80% of the conventional yields in 2050 and 25% of the area is used for biological nitrogen fixation (BNF) while the remaining area is used for cropland production, which is equivalent to a rotation of 3:1 (crop:legume). The crop types used are indicated in Section 1.3 of the SI file. Two different livestock diets, one with an increased fraction of roughage and another with an increased fraction of grain compared to the year 2000. Five potential cropland expansion scenarios, from no expansion compared to the year 2000 to 70% cropland expansion. In the dietary options, we considered a business-as-usual (BAU) diet, a balanced diet with meat where the fraction of animal products is corrected to a healthy level in comparison with the BAU diet (based on USDA and HHS, 2010), three vegetarian alternatives (lacto, ovo and ovo-lacto) and a vegan diet. All diets are described in the SI File (Table S2). In the diets with animal products, three different variants were considered, (1) keeping the current proportion of ruminant and monogastric products, (2) animal products exclusively from ruminants, (3) animal products exclusively from monogastric animals.

To reduce the influence of uncertainty in the analysis, first we studied a multitude of scenarios in terms of dietary options, cropland expansion and yields, and the main conclusions are valid for all cases assessed. Then, we also considered a dietary choice and production system unfeasible only if the limitation in area or N was higher than 5% to accommodate for uncertainty in the data (Erb et al., 2016; Theurl et al., 2020). This analysis took an "extreme worlds" perspective for the food system of the future. The analysis was also oriented towards diets. We therefore considered full global conversion of each production system and dietary option separately. The food system in 2050 is more likely to be mixed, and therefore the analysis here should be understood as setting an option space for satisfying food demand.

2.3. Nitrogen requirements and soil loss

Biomass demand was calculated for each scenario in BioBaM as the sum of (a) demand of primary crops for food and feed from cropland and (b) roughage demand for the production of meat and milk from grassland. Cropland demand included 11 categories corresponding to primary commodities and 1 class of roughage from grassland. Here, we also assessed the N removal with harvested crops, as any N removed from soil needs to, on the long run, be replenished with some kind of fertilizer or reactive N. For this purpose, we use the N content of each crop category (Table S4 in the SI file).

It is, however, not sufficient to supply fertilizer as withdrawn with the harvest. N requirements were calculated using N production curves (i.e. N yield measured as kg N output/ha as a function of the N inputs measured in kg N input/ha). N production curves are used to calculate N output as function of the applied N, typically they are assumed to follow a Michaelis-Menten relationship (e.g., see Bodirsky and Müller (2014)). In our case, N yields are known (Y) and assumed constant, we inverted the calculation to obtain the N requirements (F) as a function of the theorical maximum yield (Y_{max}), according to

$$F = \frac{Y_{max} \cdot Y}{Y_{max} - Y} \cdot A_{req} \cdot \qquad (1)$$

where A_{req} is the area required, as calculated in the BioBaM model.

The N production curves have a hyperbolic shape and therefore N use efficiency, the return in the harvested matter of any N applied, decreases as the yield increases (diminishing marginal returns). At the same time, as also shown by Eq. (1), N that was not used by the plants is the difference between N input and N output is excess N, which is lost to the environment. Because N inputs increase at a proportionally higher relative rate than yields, the equation implies that an increase of N input also increases the fraction of N lost to soil. Production curves for cropland were obtained from Mueller et al. (2017), and N production curves for grasslands were obtained from Niedertscheider et al. (2016).

2.4. Nitrogen supply

Manure used in croplands was calculated in the livestock sub-model of the BioBaM-GHG model proposed by Theurl et al. (2020). Atmospheric N deposition was calculated by multiplying the required area by the deposition rate in the year 2000 and 2050 from Dentener (2006). Here, as a simplification, we assumed that atmospheric nitrogen deposition varies in space and time. The rate of deposition is a function of the region of the World and its different between the year 2000 and 2050.

For conventional production, mineral fertilizer used was calculated by subtracting manure and atmospheric N deposition from F (calculated using Eq. (1)). We considered that there is no limit to mineral fertilizer production, and hence conventional production is never limited by N. In a scenario with higher N requirements, the extra requirements are supplied by extra portions of mineral fertilizer production – with excess N eventually being released to the environment.

In organic production, mineral fertilizer is replaced by BNF. BNF is limited by BNF rate and the area allocated for it. In each region, the total BNF ($TBNF$) was calculated by multiplying the total available area (A_{avail} – obtained from the cropland expansion scenarios in BioBaM) and the fraction of the area reserve for BNF (F_{BNF}) and the BNF rate (BNF_{rate}). In the baseline assessment, F_{BNF} was 25%, but we changed the regional length of rotations in the optimization of N availability when we tested strategies for improving food systems. The BNF rate was estimated based on Herridge et al. (2008). It is calculated using the yield (Y), harvest index (HI), root to shoot ratio (RS), nitrogen content (NC), and percentage of plant N derived from N_2 fixation ($\%Ndfa$)

$$TBNF = BNF_{rate} \cdot Area_{avail} \cdot F_{BNF} = \left[Y \cdot \left(\frac{1}{HI}\right) \cdot (RS + 1) \cdot NC \cdot \%Ndfa\right] \cdot Area_{avail} \cdot F_{BNF}. \quad (2)$$

Finally, the N requirements and potential N supply (sum of manure, TBNF and atmospheric N deposition) were compared. If the N required was equal or lower than the N supply, total N used was calculated by multiplying the potential N supply by the ratio between N requirement and potential N supply. If N requirement was higher than potential N supply, all the N available is used, and the non-supplied N was supplied by another region of the world with surplus N unused to supply local production. A scenario is not N limited if there is at least one feasible (non-limited in N) distribution of food production in the World after using this procedure, i.e. global N requirement is lower than the potential global N supply.

2.5. Carbon and nitrogen inputs/outputs

Above- and belowground plant litter and manure are the main flows of C and N into the soil organic pools. C/N ratios (Chen et al., 2003) were applied to calculate both C and N excretion of manure. Animal excretion can be used as organic N fertilizer promoting the closure of N cycle (van Zanten et al., 2016; Van Zanten et al., 2018). For example, Van Zanten et al. (2018) demonstrated that livestock raised under the circular economy approach could provide a considerable amount of human daily protein without animals consuming any human-edible biomass.

For the soil N pool, the inputs are N fertilization (synthetic and organic), N from atmospheric deposition, N captured in root nodules by rhizobia and/or free-living N-fixing microorganisms and, in grazing areas only, excreta decomposition. The outputs of the N pool are the amounts of N taken up by plants determined in the plant balance, plus atmospheric emissions of N_2O and NH_3 from soils and N leaching. The N emission in the form of N_2O, resulting from nitrification/denitrification processes was calculated using the Williams et al. (1992) method, which uses empirical parameters adapted to land use types. The direct and indirect N_2O emissions due to fertilizer application (synthetic and/or organic fertilizer) were estimated using the IPCC (2006) method. The NH_3 emissions from soil due to fertilizer application were estimated using the EMEP/EEA (2019) Tier 1 approach. We considered only one soil pool (including organic and inorganic pools). Inorganic C was excluded due to its small interaction with the organic pool that generates the most important N and GHG emissions (Lugato et al., 2018).

Plant residues (litter and belowground biomass) can either enter the soil organic pool or lead to an emission to the atmosphere of C (as CO_2) and N (as N_2O) due to the organic decomposition and nitrification/denitrification processes. The C emissions from litter were calculated neglecting the change in C:N ratio of the litter due to N_2O emissions. Emissions from burning residues were also included (IPCC, 2006).

The livestock C and N balances were based on the metabolic activity of monogastric (pigs, poultry) and ruminants (bovine, caprine and ovine). C and N accumulation in livestock were calculated based on the intake (grain, crop residues and roughage) and outputs (meat, milk and eggs). C emissions (as CO_2 and CH_4) and N emissions (as N_2O and NH_3) from livestock production and manure management were also considered using the IPCC (2006) Tier 2.

2.6. Strategies to increase the sustainability of food system

Here, we considered three types of improvements possible: agroecological improvements, circular economy and decarbonization strategies. In the agroecological strategies, we considered improvements of NUE (Lassaletta et al., 2016), improvement of the feed conversion efficiency of livestock production (Herrero et al., 2016) and biological N fixation (BNF) in combination with optimization of crop rotation in organic production. Circular economy strategies included the performance of strategies such as reduction of food waste (Muller et al., 2017; Springmann et al., 2018) and the closure of nutrient cycles through use of organic municipal solid waste and wastewater treatment plant (municipal and human waste) sludges as fertilizer. Finally, we assessed the effects of a “decarbonization strategy” (Springmann et al., 2018) where in 2050 mineral N fertilizer production (in conventional farming) has zero GHG emissions due to the elimination of N_2O emissions during production through the use of catalysts and the replacement of fossil fuels with renewable energy and methane biogas from manure management. For details on data and the implementation of improvement strategies, see section 1.5 in the SI file.

Here, we considered multiple agroecological strategies for improving food systems. These practices are drawn from the field of agroecology, which is a wider holistic approach that uses a variety of agroecological practices and also refers to other aspects of food production besides agronomic factors that were not considered in this study. The first agroecological strategy considered here was the improvement of NUE of crop production, obtained by a higher theoretical maximum yield (Y_{max}) for the year 2050. Data was obtained from Lassaletta et al. (2016). For the improvement of livestock production efficiency, we considered that the feed conversion ratios (FCRs) in the different world regions were improved to the level of the region with lower FCR s, according the used FCR in the model (based on Bouwman et al. (2005) – statistical approach based on FAO data). Finally, we considered an improvement of BNF rate in combination with optimization of crop rotation in organic production. For this, we considered a doubling in the BNF rate in relation with the baseline assessment (baseline BNF rate is ~70 kg N ha^{-1}). Regarding the optimization of crop rotation, first, we increase the fraction of time with legumes in rotations from 25% to 50%. Second, with this change, we calculate the potential N available globally and compare it with N demand in each scenario. If potential available N is lower than N demand, the scenario is limited by N as there would be no global distribution of production that would ensure sufficient N to respond to food demand. Otherwise the scenario is non-limited by N and there is at least one way of distributing food production globally that is feasible. Third, in the non-limited scenarios we optimized the rotation, i.e. changed the fraction of legumes in the rotation in all regions. All cases were assessed for possible limitations in terms of availability of land. In the regions that are limited in N, the fraction of legumes in the rotation was increased up to a maximum level of 75% of legumes, using all land available (note that this is always possible, because this increase in legume fraction does not change the total amount of land used). Food production in those regions is therefore reduced (down to the maximum that can be produced with the N available in the region) and moved to the regions with available N. The fraction of legumes in the rotation in those regions was then reduced in order to produce more food, compensating for the extra food demand from the limited regions.

In the circular economy strategies, the first measure studied was an elimination of food waste during agricultural production at the farm level. We did not consider reductions of wasted food during transportation, retail and consumption. This was due to the fact that the second measure was the development of new sources of N fertilizer from organic municipal and human waste as organic fertilizer. If we considered reductions in food waste at retail or consumer stages, organic residues would decrease and they would be no longer available for producing N. The total municipal and human waste production per region was calculated using the World Bank projection of municipal solid waste generation for the year 2050 and the organic fraction of municipal solid waste (World Bank, 2018) (in Supplementary Table S2).

For decarbonization strategies, we considered the use of alternative production processes or energy sources (e.g. use of renewable energy) in the Haber–Bosch process. i.e. substitution of fossil fuels with renewable energy. As a simplification, we assumed that the GHG emissions of the renewable energy used will be zero in the year 2050. The impact of renewable energy production is very low compared with other energy sources, and in the future, the GHG emissions of energy consumption can be even lower as result of expanded electrification and technological developments (e.g. carbon storage technology). We also assessed the potential for reducing GHG emissions by using CH_4 from manure management to produce biogas. Using biogas leads to oxidation of CH_4 to CO_2 and consequently reduces global warming potential (as the global warming potential of CH_4 is 34 kg CO_2e/kg CH_4 with carbon feedbacks and for CO_2 it is 1 kg CO_2e/kg CO_2) (Holm-Nielsen et al., 2009; Nasir et al., 2012).; The avoided CO_2 impacts from replacing natural gas with biogas were neglected because they represent a 1 for 1 substitution in CO_2 from natural gas, and hence have a comparative effect of only 1/12. However, we considered additional CH_4 emissions due to leakages of 2.4% of the CH_4 produced (Scheutz and Fredenslund, 2019). Further, we also considered that N_2O emissions that usually occur during manure management are avoided (equal to zero) when manure is used for biogas production (Scherson et al., 2014). This strategy improves the GHG balance of the global food system without affecting the feasibility of organic farming due to area or N availability. Methane is converted to CO_2 during manure management and treatment, and manure is used in cropland, meaning that the N is conserved.

Table 1 describes the individually modelled strategies and the production systems affected by them. Details about the implementation of these strategies can be found in the Methods section.

Table 1. Description of the strategies considered in this study. BNF – Biological nitrogen fixation, GHG – Greenhouse gases, N – Nitrogen, NUE – Nitrogen use efficiency.

Type of improvement strategies	Individual strategies	Description	Affected production types
Agroecological	Improvement of NUE	Reduction of N requirements per unit of area of cropland and grassland	Both (conventional and organic)
	Improvement of livestock production efficiency	Reduction of cropland and grassland use for animal feed and reduction of enteric fermentation emissions and animal excretion	Both (conventional and organic)
	Improvement of BNF rate and optimization of crop rotation	Increase of total fixed N and supplied for cropland production	Only organic production
Circular economy	Reduction of food waste during production to zero	Reduction of food demand that reduces required cropland production	Both (conventional and organic)

	Use of all organic municipal solid waste municipal and human waste as fertilizer	Additional source of fertilizer that increases of total N available for cropland production	Both (conventional and organic)
Decarbonization	Decarbonization of mineral fertilizer production and elimination of N_2O emissions	Elimination of GHG emissions from fertilizer production	Only conventional production
	Production and use of methane from manure management	Reduction of GHG emissions due conversion of CH_4 to CO_2 through combustion	Both (conventional and organic)

Here, we considered a “snapshot” of the food system in the year 2050. With this assumption, we removed all transient emissions or sinks from C and N mass balances, namely carbon sequestration. Further, C sequestration in soil and biomass is not a permanent sink and can be transformed into a C source with land use changes in the food systems.

3. Results

3.1. Full scale conversion to organic farming without additional measures

Full conversion to organic farming by 2050, under future scenarios of yield and efficiency without additional measures would be infeasible for any dietary choice due to N deficits (Figure 1). While all conventional production scenarios show a minimum N surplus comparable to the year 2000, with maxima ranging up to 120 Tg N yr-1 (1 Tg = 10^{12}g = 1 million metric tons), the deficit of organic farming scenarios ranks between 37 – 93 Tg N yr^{-1}. In general, scenarios with increased conventional crop yields are associated with larger N surplus because NUE marginally declines with increasing yield levels, and therefore each additional unit of N used by the crops to grow increases the N applied and lost to ecosystems (Lassaletta et al., 2016; Mueller et al., 2017). For organic farming, N limitations are smaller in diets that involve non-meat animal products (lacto and ovo-lacto diets), which is due to the additional availability of livestock N in the production system. In these scenarios, N contained in biomass from grassland is transferred to cropland via livestock manure. The vegan and ovo diet use less area than diets that include milk and meat, as farmland for ruminant feed production is unnecessary. Those two diets, however, are associated with the largest N deficits of all scenarios (~80-93 Tg N yr^{-1}). Diets with meat have higher N deficits than lacto or ovo-lacto diets.

The difference between scenarios involving products from ruminants and monogastrics are small because of two opposite effects. On the one hand, ruminants require more agricultural area (cropland and grassland) and are less efficient in converting inputs into products than monogastric animals. On the other hand, despite the lower feed demand of monogastric animals, they consume mostly crops and therefore require more cropland area. Ruminants consume mostly crop residues/by-products from food manufacturing and graze and therefore do not contribute as much to increasing cropland demand, requiring grazing land instead. For example, considering FAO 2050 yields, a cropland expansion of 11%, “grain” variant of animal feedstuff composition, and BAU diet, and comparing the source of the animal products in diet between only ruminant products with only monogastric products, the first requires about 5,900 Mha where only about 1,850 Mha are cropland, and the second requires about 1,800 Mha but about 90% of the agricultural area required is cropland.

Organic production systems are, in consequence of lower yields, more limited than conventional production systems, and always require substantial cropland expansion (70% in the meat diets and 11% in the non-meat diets, results shown in Figure S3 in the SI file). This additional cropland demand is both caused by lower harvest yields and the integration of N-fixing legumes within the crop rotation. Total GHG emissions of diets that use organic farming are also higher than emissions in 2000, except for diets relying on non-meat or monogastric animal products. GHG emissions are reduced in those diets due the avoidance of enteric fermentation emissions and due to the reduction of cropland use for feed production, while at the same time reaping the benefits of N fertilization from non-ruminant manure when compared to vegan alternatives. In general, the maximum GHG emissions of organic farming, associated with diets containing meat, are smaller than the maximum emissions from corresponding conventional systems. Organic scenarios have lower GHG emissions than conventional production scenarios because GHG emissions from BNF are lower than total GHG emissions of synthetic fertilizer (production plus application). GHG emissions from BNF are due to the decomposition of biomass. Part of the N incorporated in aboveground biomass is left on the soil, causing direct N_2O emissions similar to emissions from crop residues left on the field. Nevertheless, organic scenarios require more area for BNF, which leads to more total soil N_2O emissions. In diets that include meat (BAU and balanced meat diets), organic scenarios have lower GHG emissions due to the N fertilizer from manure, but in diets without meat, due to the lower quantity of manure, organic scenarios have higher GHG emissions.

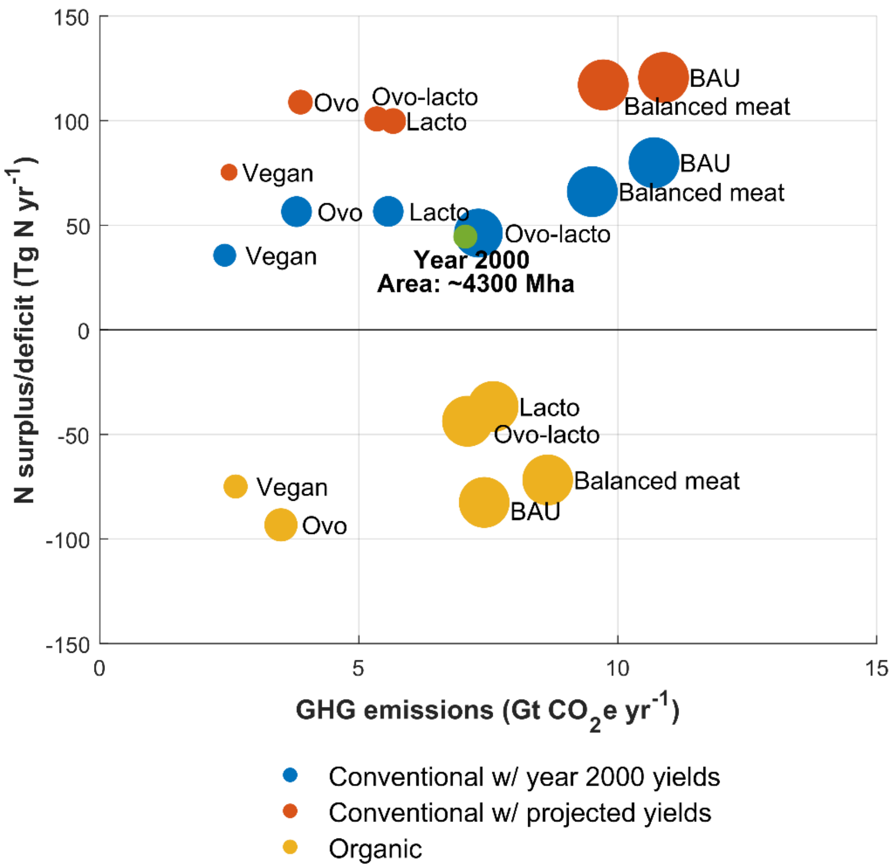


Figure 1. Effect of conversion to conventional production (with projected yield and year 2000 yields) and conversion to organic farming without additional measures on N surplus/deficits, GHG emissions and used farmland (cropland plus grassland) in the year 2050. Among the five cropland expansion variants and two livestock feed variants, only the scenario with lowest cropland use is shown (all scenarios with higher cropland expansion are feasible as well), GHG emissions and N surplus/deficit (surpluses are located in the top half of the graph and deficits in the lower half, and where N surplus is considered the potential for N pollution; scenarios with N deficit are not viable; the optimal value for N surplus/deficit is 0). The size of the circles indicates the used farmland. The starting point is the Year 2000, represented as a green dot.

3.2. Impact of additional measures in the organic food production system

So far we showed that organic farming is infeasible without additional measures. However, a bundle of additional measures exist to be tested if diets based on organic farming could be rendered feasible.

If all innovations applicable to organic farming were implemented, organic farming would not only have more N at its disposal than needed, but also greatly reduce GHG emissions and farmland used compared to 2000 (Figure 2). The feasibility of organic farming in terms of N constraints is aided by the presence of some animal products in the diet. A moderate amount of animal products in the diet can help the N budget. This is particularly true of the ovo-lacto diet, which is the most efficient in providing sufficient N while minimizing N surplus (15 Tg N yr^{-1}). This diet involves non-ruminants (the “ovo” component) fed on crops, thus requiring more N than vegan diets. However, the effect is counteracted by ruminants (the “lacto” component), which transferring N from grassland to cropland via manure, reducing the need of additional land for biological N fixation. The two effects are complementary and produce the most well-balanced diet in terms of N use. Vegan and lacto diets minimize land use (~1,600 Mha yr^{-1}), while the vegan diet is the one with the lowest GHG emissions possible (1.9 Gt CO_2e yr^{-1}).

N limitations in organic farming can only be overcome through the joint implementation of agroecological innovations (Figure 2). Under organic farming, all diets with ruminant products result in increased farmland area demand in relation to 2000 due to the need for area set aside for BNF. Nevertheless, GHG emissions are lower in all diets than in 2000, which means that conversion to organic with all improvements is capable of overcompensating for the increased demand for food from a growing population with lower emissions per unit of food produced. The application of any agroecological innovation in isolation is insufficient to overcome the N limitation in organic farming. Results are shown for each individual innovation in Figure 2 and in detail in SI. Improving NUE has the largest individual contribution and, on average, is capable of decreasing 52% of the global N deficit.

Circular economy strategies reduce the N surplus generated by the application of agroecological improvement measures because reducing food waste and using N from wastes reduces the area required for cropland production and its N_2O emissions from soil.. These strategies can reduce organic N deficits by about 17%. Used land and GHG are not significantly affected (Figure 2). The decarbonization strategy applicable to organic farming, use of biogas from manure management, only affects diets including animal products. GHG emissions reductions are the largest (on average 4% - 8%) in monogastric systems because they produce more managed manure than ruminants per head. The highest reduction is in the BAU diet with monogastric products (11%).

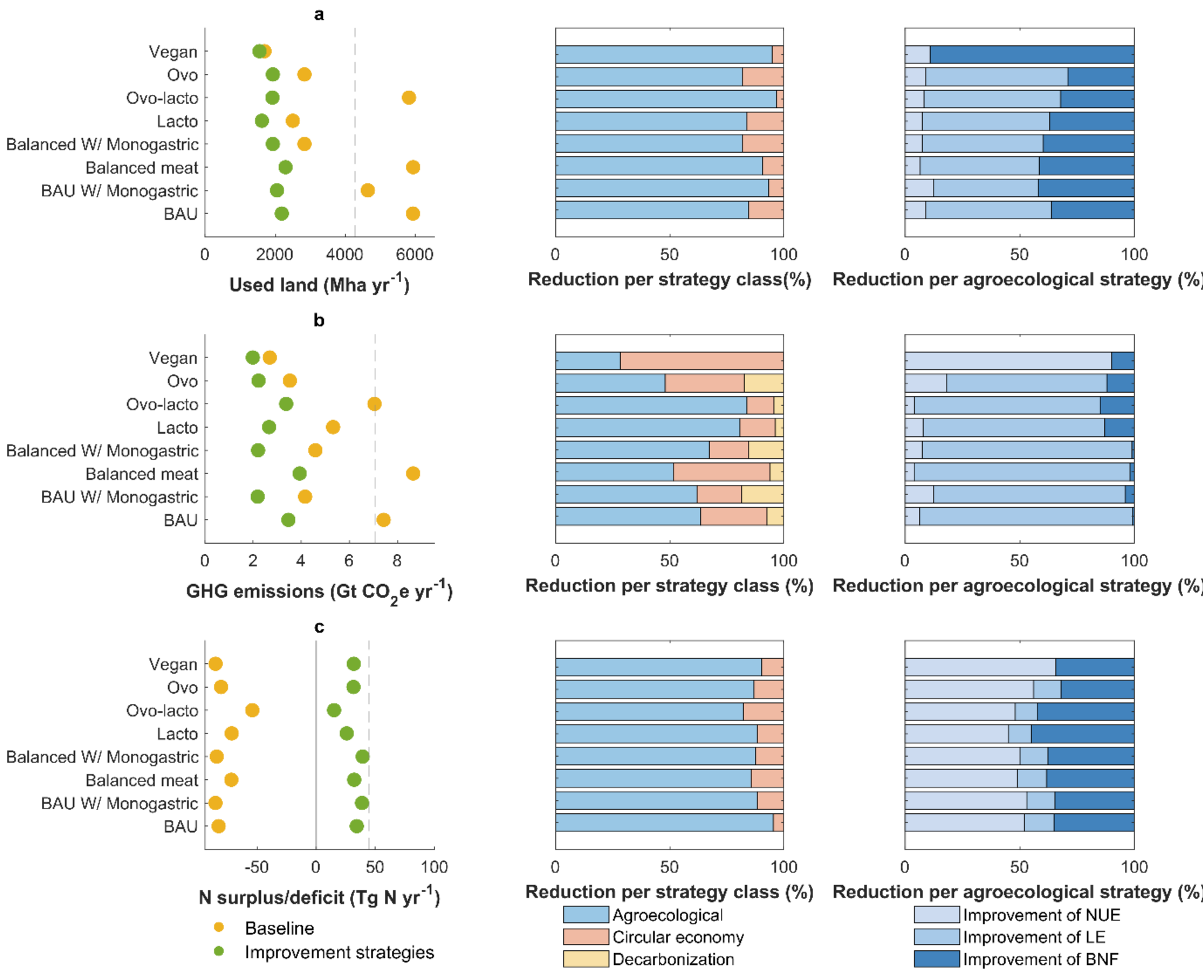


Figure 2. Effect of different strategies on used land (a), GHG emissions (b) and N surplus/deficit (c). Results for the baseline are displayed as orange dots. 'Baseline' is the analysis conducted for full conversion to organic farming by 2050, under future scenarios of yield and efficiency without additional measures. The results for improvement strategies are represented with green dots. In each subplot, the vertical dashed line represents the value for the year 2000. BNF – Biological nitrogen fixation, GHG – Greenhouse gases, LE – Livestock efficiency, N – Nitrogen, NUE – Nitrogen use efficiency.

3.3. Is fully optimized organic production more sustainable than conventional farming?

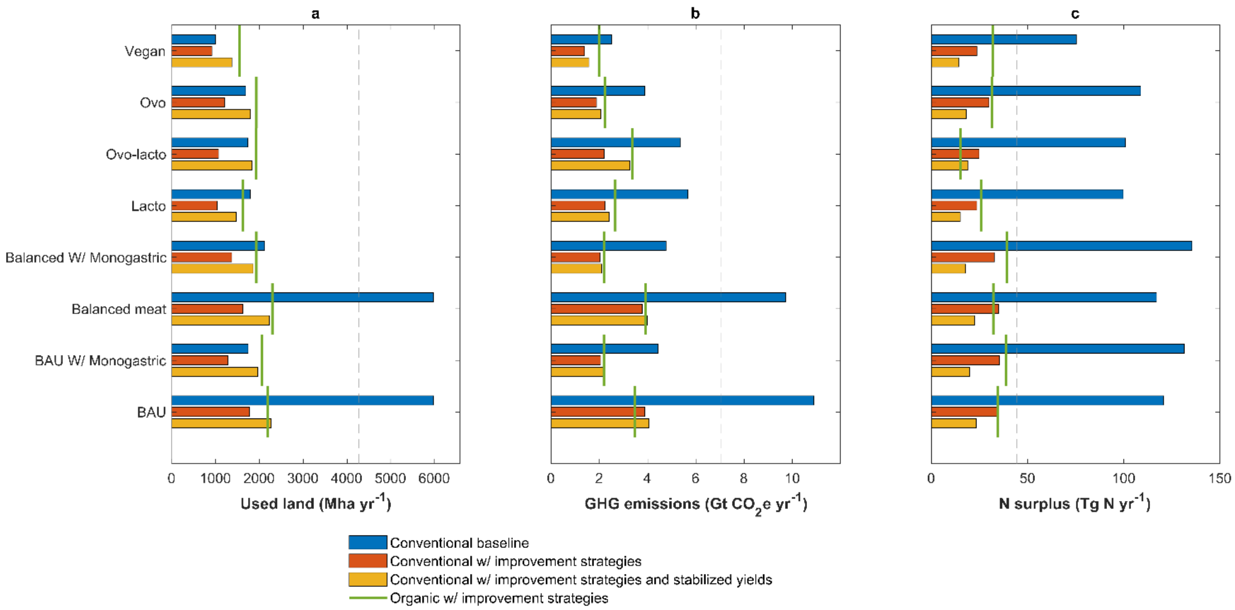
Fully optimized organic production, i.e. after implementation of all improvement strategies, is associated with less GHG emissions and a lower N surplus than conventional production, regardless of dietary choices, but it uses more farmland due to the need of land set aside for BNF for scenarios without ruminants, including vegan and vegetarian scenarios (Figure 3, comparing "baseline conventional" and "organic with improvement strategies"). Analysing the same dietary options studied for organic farming, but now for conventional farming (full results shown in SI file

– Section 2), showed that, when yields are expected to grow, the area used for food production is smaller. Higher yields in conventional systems, as for organic systems, save land for natural restoration but generate higher GHG emissions and N surplus due to the greatly reduced NUE.
Conventional production can be optimized through the same strategies as the ones applied for organic farming described previously. This results in a reduced mineral fertilizer demand and thus reduced GHG emissions from upstream energy requirements of fertilizer production. The agroecological strategies that influence conventional production are the improvements of NUE and FCRs. Circular economy and decarbonization strategies also apply, where for the latter we also considered additionally the full decarbonization of N mineral fertilizer production through use of renewable energy. Results show that, if yields are higher in 2050 than in 2000, then all diets produced conventionally spare much more land for nature than organic (Figure 3 comparing conventional and organic with improvement strategies). Due to our assumption that BNF occurs in a fraction of the required land for food production, organic scenarios, even if yields increase significantly, require additional land for biological N fixation and therefore, despite improvements, are still relatively worse than conventional farming in this regard. Organic production would, however, have similar or slightly lower N surpluses and GHG emissions than optimized conventional production for comparable diets.

If, however, we assume an alternative future where conventional yields remain at the level in the year 2000 (low-intensity optimized conventional farming), conventional production improves its performance over more intensive farming (i.e. with increased use of fertilizers and yields). The vegan diet with conventional production systems in particular shows a better performance in terms of GHG emissions, N surplus and land demand compared to any diet combined with organic production (Figure 3, comparing "conventional with improvement strategies and stabilized yields" and "organic with improvement strategies").

The significance of these differences, however, is difficult to interpret given the inherent uncertainty of the data. The conventional production vegan diet uses 40% less farmland (conventional: 914 Mha; organic: 1551 Mha), is responsible for 30% less GHG emissions (conventional: 1.4 Gt CO_2e yr^{-1}; organic: 2.0 Gt CO_2e yr^{-1}) and has a surplus 5% lower than the best diet produced using organic farming (conventional: 14.4 Tg N yr^{-1}; organic: 15.1 Tg N yr^{-1}), i.e. the diet that minimized the impacts of organic farming in each dimension. To put these numbers in context, in 2050 the conventional vegan diet and the ovo-lacto organic use 70/55% less farmland, are responsible for 80/50% less GHG emissions and have an N surplus 50/45% lower than the diet with the production mix and the food demand of the year 2000 (also represented in Figure 3), despite a growing food demand. Both production systems and multiple diets produced according to each of them have the capacity to significantly decrease the environmental burdens of food production in the future.

Overall, there is no combination of diet variant and production variant that is significantly better than all the others in all indicators (land used, GHG emissions and N surplus). An improved vegan diet with increased conventional yields is the alternative that maximizes spared land for nature, but it has a higher N surplus than the organic ovo-lacto diet. The vegan diet from conventional production with stabilized yields uses more farmland and minimizes GHG emissions, but also has the same N surplus as the organic ovo-lacto diet. Organic production in diets, particular vegetarian, is in general a good option for reducing N surpluses, but it requires using more land than their conventional counterparts. Such diets are also better aligned with the current view on healthy diets, which include limited amounts of animal products (Godfray et al., 2010; Willett et al., 2019).



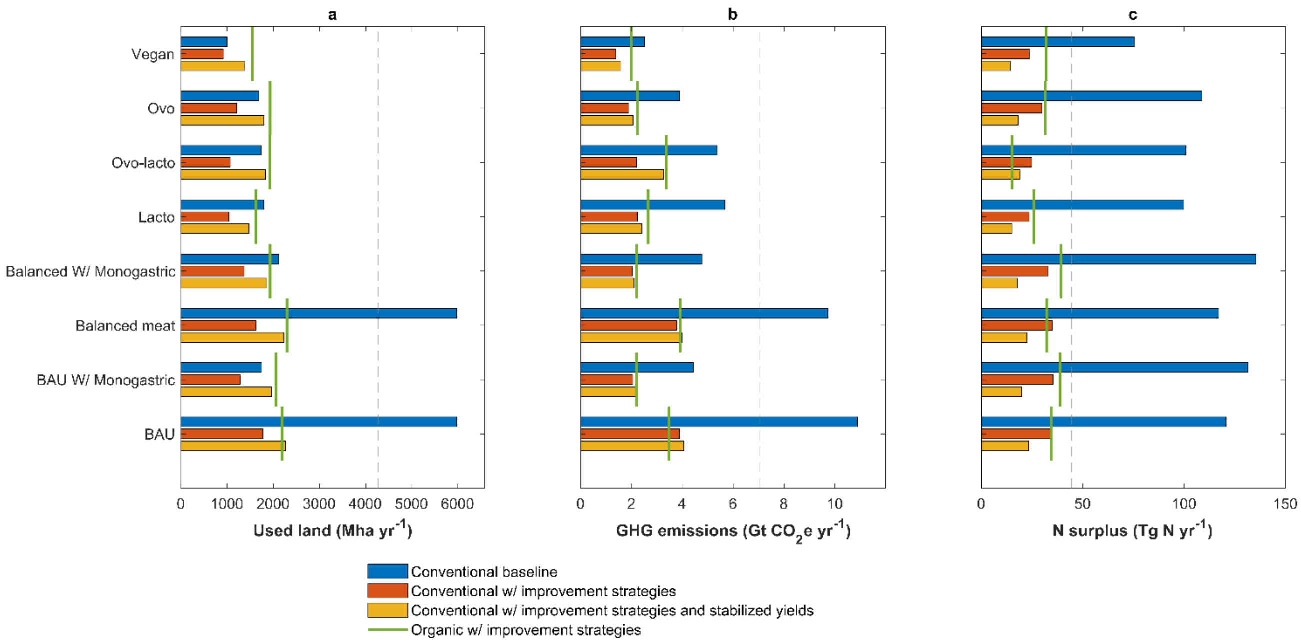


Figure 3. Comparison of production systems on the performance of diets on used land (a), GHG emissions (b) and N surplus (c). In each subplot, the vertical dashed line represents the value for the year 2000.

4. Discussion and conclusion

Our results highlight the trade-offs in the food system between diets, land use, GHG emissions and the N balance. The main question related to future sustainable food systems should therefore be if and how these trade-offs can be mitigated.

An often-voiced suggestion for feeding a growing population is to close crop yield gaps (Foley et al., 2011; Lamb et al., 2016). Our results showed that this option, taken to its attainable maximum, would save area for natural restoration but would also increase GHG emissions and N surplus (Erb et al., 2016; Theurl et al., 2020). Maintaining crop yields at current levels would generate less GHG emissions and N surpluses due to higher NUE but it would also require more farmland. Another frequent suggestion for improving the food system is dietary change (Bajželj et al., 2014; Erb et al., 2016; Springmann et al., 2018; Theurl et al., 2020; Tilman and Clark, 2014). Dietary changes are particularly suited to alleviate some of the trade-offs by enlarging the option space (Erb et al., 2016; Theurl et al., 2020) and reduce GHG emissions. However, as we show here, any switch between ideal-types of diets carries new trade-offs. Less animal products reduces GHG emissions due to less CH_4 from enteric fermentation, but it also reduces an important source of N in the food system, i.e. N in animal manure, that must be compensated for. Ultimately, universal adoption of some diets such as veganism, while beneficial in some respects, can also create other problems such as that of the N balance, as previously observed by Theurl et al. (2020).

Besides the trade-off of organic farming, which is a reduced per-area pressure with increased area demand (Balmford et al., 2018; Smith et al., 2020), we find an additional, virulent trade-off. The strategy to keep area demand at bay by reducing the share of animal products in diets (Barbieri et al., 2019; Muller et al., 2017) results in an N deficit of the production system that cannot be closed within rotational systems. We calculate a deficit of 9 kg N $(ha.yr)^{-1}$ for the ovo-lacto diet, similar to the result by Mueller et al. (3-7 kg N $(ha.yr)^{-1}$). Our results are also inline with Barbieri et al. (2021) considering organic production without improvement strategies. We found a N deficit of about 75 Tg N in BAU scenario and Barbieri et al. (2021) estimated about 36 Tg N. With improvement strategies, our results showed that organic farming is not N limited, but Barbieri et al. (2021) is still limited. Notice that Barbieri et al. did not consider the improvement of NUE, which is one of the most important strategies to overcome N limitation in organic farming. The difference between the two estimations can be attributed to different assumptions of crop yields, and different rates of food waste generation. A detailed comparison of our study with the literature (including the studies cited here and others) was performed in this work, and is shown in the Supplementary file S1, section 2.5 and in particular Table S11.

Because the high potential of organic farming to reduce GHG emissions, even if area or N availability are limiting factors, it is worth exploring ways to surpass these shortcomings. A strategy could be to allow for some level of mineral N fertilization – at levels about 40% of current average uses. Such strategies would, however, not comply with the current standards of organic farming and thus present challenges of good practice and governance (Muller et al., 2017) and not abate the problem of increased area demand. Additionally, this strategy would fail to fully avoid GHG emissions, even if mineral fertilizer is produced in fully decarbonized industrial systems, due to the N_2O emissions related to N application. It should be noted that organic production requires more area for BNF, which leads to extra soil N_2O emissions that can compensate for the lower emissions per unit area from BNF.

Our study demonstrates the necessity of integrated and multi-dimensional approaches to meet the challenges of the global food system. There are options that render organic farming feasible in terms of land and N availability while minimizing GHG emissions and sparing farmland. These options require a combination of ambitious agroecological innovations, circular economy and decarbonization strategies. No individual measure alone is sufficient to overcome the N gap. The implementation of these strategies presents formidable technical, managerial as well as governance challenges, as discussed below.

Improvement of NUE is frequently suggested as one plausible solution to reduce N surplus (soil and GHG emissions) (Lassaletta et al., 2016; Mogollón et al., 2018; Mueller et al., 2017; Zhang et al., 2015). In short, improvement of NUE requires the so-called fertilization with the right rate, with the right timing, in the right form and right placement (Bowles et al., 2018; Zhang et al., 2015). The implementation of the concept varies geographically. In some regions, large gains can sometimes be obtained without further technological developments. Regions such as Africa or Central Asia have low crop yield due to low N applications and depletion of N soil reserves. In these regions, both crop yields and NUE could be improved together through an increased rate of application of N fertilizer (mineral or organic) and improved management (Mueller et al., 2014; Sutton et al., 2013). Other locations in East and South Asia, Europe and North America use excessive N fertilizers generating N surpluses, may require widespread technological improvements such as slow-release fertilizers, nitrification and urease inhibitors or fertigation to increase NUE without reducing crop yields (Zhang et al., 2015). Research into precision fertilization is likely to yield results by 2050 and the practical implementation of those practices could strongly assist organic systems to better use N (Zhang et al., 2015).

In organic systems, the application of precision farming principles regarding N management is a challenge on its own. Organic fertilizer application rates can be micromanaged, but the main contributor to the N balance is BNF, particularly for diets with less animal products that target GHG emissions reductions. Fine-tuning BNF is possible in two main ways: through increase of BNF rates, and through adjustment of the duration of crop-legume rotations. Other alternatives were not considered, namely modelling BNF in productive legume systems and/or intercropping legumes. Regarding the use of productive legumes for BNF, this would reduce the BNF per unit of area and at the same time reduce the cropland area demand due to the legumes produced in areas used for BNF. These are two opposite effects and therefore likely to have a null effect on the balance. Intercropping legumes would be an win-win solution, contributing to BNF in productive cropland area and avoiding further area for BNF. However, data to model intercropping legumes systems at global level is, to our knowledge, not available.

Doubling of the BNF rate in relation to the present average regional rates is described as plausible and within the range of variation found in the literature (Collino et al., 2015; Herridge et al., 2008; Muller et al., 2017). The maximum regional BNF rate assumed in this paper, about 360 kg N/ha, is similar to the maximum rate observed by Herridge et al. (2008). Such a doubling of the rate would require the selection of N-fixing species (and/or associations of species) and precision farming (Herridge et al., 2008; Keyser and Li, 1992). The potential of BNF to increase is highly dependent on the location (Herridge et al., 2008; Lassaletta et al., 2014), which makes improving BNF rate in all regions of the World an ambitious target, as the most promising experiences are restricted to specific locations (e.g. South America) (Herridge et al., 2008; Lassaletta et al., 2014). In general, fodder crops and pastures based on legumes with rhizobium typically have the maximum rates of BNF (Herridge et al., 2008). The use of those plant types with high potential for N fixation in rotations, selected from a pool of native species in each region, is the most promising alternative for achieving rates of BNF close to the one used in this study.

Adjusting durations of rotations is a necessary complement to increased BNF (Barbieri et al., 2019, 2017). Rotations with longer periods of unproductive legume farming, supply more N to the soil for future crop production. Getting the correct and necessary amount is the work of precision farming, as it requires striking the right balance. Very long rotations typically result, on a global scale, in farmland limitations due to low area devoted at any given time to food production. Very short rotations result in N limitation. The optimum duration of the rotation should be defined regionally in order to match supply and demand of N that maximizes food production up until the total food demand of the region (Barbieri et al., 2019).

There is a vast literature that discusses the implications of closing yield gaps for cropland production (Kanter et al., 2016; Mueller et al., 2012) but, surprisingly, strategies that aim at increasing of livestock efficiency are much less studied. We find that increasing the efficiency of livestock production is key for reducing land requirements, GHG emissions and N surplus (Herrero et al., 2016, 2013). The most direct way to close livestock yield gaps, used in this study, is to reduce FCRs for all livestock types and feed used. We operationalized this idea by assuming that we can have FCRs for each region of the world equal to the global minimum FCR. Achieving this target in all regions of the world would require the selection and territorial expansion of breeds that add more weight per unit of feed, adjusting age at slaughter to avoid feeding livestock once growth decelerates, careful design of livestock diets including new ingredients and better feed quality, among others factors (Steinfeld et al., 2010; Herrero et al., 2016; Thornton and Herrero, 2010). It is important to note that these new feeds may have higher environmental impacts than currently used feeds (Wilfart et al., 2016). However, owing to its large potentials, further research is necessary to understand the challenges and opportunities of improving FCRs at a regional level and mitigate adverse effects of raising FCRs on animal welfare.

Circular economy strategies, such as waste reduction and N harnessing from municipal waste, require either the development of an appropriate financial/institutional infrastructure through restructuring of food production and distribution networks, and even new legislation or industry standards. Even in the medium/high income countries, reducing food waste during production to zero by 2050 may be unrealistic as it implies a long institutional conversation between producers, retailers and consumers, plus new distribution chains of fresh products as well as reclaim of wasted/spoiled products that can be reused in the food industry or refed into the system for new food production (Gustavsson et al., 2011; Parfitt et al., 2010). In low income countries, food waste is mainly connected to financial, managerial and technical limitations (Gustavsson et al., 2011). Previous studies concluded that reducing food waste has a limited effect of GHG emissions and N use (Springmann et al., 2018), similarly to what we found. Using municipal and human waste as fertilizer raises sanitary concerns and heavy metal contamination of the soil that may present barriers towards the application of these technologies (Hargreaves et al., 2008; Kominko et al., 2018). However, there is evidence that a safe utilization of these resources can be ensured with development and implementation of comprehensive industry standards (Hargreaves et al., 2008). The use of municipal and human waste as fertilizer can reduce both waste generation of food and GHG emissions with already existing technology (Bogdanov et al., 2019; Holm-Nielsen et al., 2009).

The decarbonization measures require involvement of additional actors. Biogas production from manure already exists in small/medium scale (Bataille et al., 2018; Holm-Nielsen et al., 2009; Lechtenböhmer et al., 2016) and the challenge is finding the correct financial incentives or restructuring of the energy sector for global spread of those technologies. Here, we considered only the oxidation of CH_4 into CO_2 and excluded extra CH_4 that can be obtained in manure biodigesters due to lack of data for global assessment. The CH_4 production rate in biodigesters is highly variable, depending among others on the mixing system and microorganisms involved (Koniuszewska et al., 2020).

From all options assessed in our study and after all improvements are implemented, the advantages of organic farming are maximized in diets with some level of animal products. Animals can be fed with co-products, biomass from marginal land and food waste, which has high nutritional value. Despite its visible effect, we made here several simplifications that curtailed the positive contributions of animals to the food system, namely: (1) improvements of BNF were applied to cropland rotations only and not to pasture land, (2) we assumed that animals did not graze legumes in rotations – grazing animals introduced during rotations return almost all N to the soil, which makes the rotation more efficient as the legumes are used for feed and still introduce N into the soil, and (3) in the vegetarian lacto and ovo-lacto diets only milk is consumed and not meat. Global conversion to vegetarian diets would decrease demand for meat to zero but dairy systems would continue to necessarily co-produce steers and old cows, and eggs could be produced through non-food feedstuff. Here, meat from retired laying hens and dairy systems was assumed to be lost. In reality this meat could supply a fraction of the demand for food using "demitarian" or "flexitarian" diets. If we considered those extra animal protein, they would fulfill some of the demand without needing additional inputs and therefore reduce used land, GHG emissions and N surplus of vegetarian diets. A scenario that considered those dairy and hen co-products could perform as well as the vegan diet. Further, in this work, fixed shares of animal products in human diets were considered with fixed ruminant/monogastric ratios. However, this may lead to regional inefficiencies of resource use.

The vegan diet when combined with improved conventional production performs best in terms of reducing GHG emissions, sparing land and minimizing N surplus. However, this option and the best performing vegetarian diets combined with organic farming show a similar performance. With this regard, it is important to note that our assessment is strictly limited in scope. For instance, health risks, such as those of inappropriate implementation of diets have not been assessed. For instance, vegetarian and vegan diets have a higher risk for developing vitamin B12 deficiency (Pawlak et al., 2014), or a low intake of calcium and vitamin D (Schüpbach et al., 2017).

The effect of synthetic fertilizers and pesticides on human health were outside the scope of this work. Nevertheless, there is evidence that organically produced products, with lower/no use of pesticides and synthetic fertilizers, have higher concentrations of antioxidants and other compounds, and lower incidence of severe diseases (e.g. cancer, neurological disorders and diabetes) (Barański et al., 2014; Rani et al., 2021; Silva Pinto et al., 2020; Smith-Spangler et al., 2012). Manufacturing pesticides also generates GHG emissions that were not considered in this study. However, their emissions are significantly lower than all other GHG emissions included

(Audsley et al., 2009). Pesticides additionally contribute to degradation of soil quality, among other environmental issues (Rani et al., 2021; Silva Pinto et al., 2020).

In this work, we did not conduct a formal uncertainty analysis. Theoretically it would be possible to consider for example an uncertainty range for the most important variables/parameters, e.g. nitrogen content and nitrogen response, combined with a Monte Carlo approach. Plausible ranges of variation would have to be found for critical values and a probability distribution would have to be used – and those may vary among food products. This analysis would likely not affect the general results for comparison between scenarios given the "extreme worlds" approach used in this work. However, follow-up work should strive for incorporating uncertainty as that would enable, for example, understating when differences in performance between scenarios is statistically significant.

The policy aim of no deforestation for the future leads to a significantly higher number of scenarios that are unfeasible (e.g. scenarios with diets rich in meat). The use of the year 2000 for the baseline was more demanding of the future, as we compared 2050 to a year with more forest areas. Compared to 2020, this means that we expect some afforestation in 2050. Therefore, if we had used 2020 for the baseline year we could have ended up with some additional feasible scenarios, but those scenarios are on the verge of being infeasible and therefore are never the focus of our paper, which aimed to find transformative changes of food systems.

This study also demonstrates the effect of some of the measures proposed in the "Farm to Fork" strategy of the European Green Deal (European Commission, 2020). The target to reduce excess of nutrients (reducing nutrient losses and use of fertilizers) were addressed in this paper through both agroecological and decarbonization strategies, mainly the improvement of NUE and decarbonization of synthetic fertilizer production. These strategies showed that a reduction of more than 50% of GHG emissions due to lower emissions from N fertilizer production. In terms of N surplus, a reduction of more than 70% can also be obtained at global level. Farm to fork strategies also envision 25% of total farmland in the European Union being used for that organic production by the year 2030, which makes the development and application of the strategies studied here urgent.

Finally, we conclude that options for a sustainable global food system exist. Even while facing rising populations and food demand, it is possible to devise a food system where land can be spared for nature, emissions of GHG are substantially reduced and N losses into ecosystems are minimized. However, our results show that no single measure could achieve this goal. Organic farming without additional measures results in N deficits. Dietary changes are not a panacea – due to their trade-offs regarding the N balance. Thus, dietary change is an important, but not a sufficient precondition for a sustainable food system transition. On top of this, other formidable challenges prevail: improvements of BNF or NUE are likely to require technological innovations and their fast deployment. An implementation of N collecting schemes is tedious and itself associated with many difficulties and trade-offs, in particular in regions that do not have easy access to resources such as technology and knowledge, including monitoring. A decarbonization of the energy systems similarly requires massive social and political efforts. On top of this, only the combination of measures regarding N management and the decarbonization of the energy systems can alleviate the pressures from the food system, as no individual solution will be sufficient. We thus conclude that there is a pressing need to move forward from simple comparative assessments of diets or between the advantages and disadvantages of conventional and organic farming. Improving the global food system requires taking into account system level effects and feedbacks and move towards the implementation of improvements such as the ones studied here. Such an effort will require massive restructuring of the food system, as well as technological improvement and deployment of decarbonizing technologies outside the agri-food sector. This is the result of a biophysical analysis of preconditions and constraints. Understanding the impacts of social dimension that facilitate or hinder such a transition is a considerable next challenge for science and research.

Competing interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Author Contribution

T.M. performed all data collection and analysis, performed most calculations and co-led the writing of the manuscript with R.T., who also provided general guidance for the direction of the work and its interpretation. T.M, C.L, M.T., A.M and L.K. implemented the carbon and nitrogen model. W.W., H.H, T.D. and K.E. supervised and organized the work carried out and the interpretation of results. All authors assisted with results interpretation and revising draft versions of the manuscript.

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