UNRAVELLING CHINA'S LIVESTOCK TRANSITION:

NUTRIENT FLOWS AND GREENHOUSE GAS EMISSIONS

Zhaohai Bai

Propositions

1. The sustainability of China’s food system increasingly relies on the performance of its livestock production system.
   (This thesis)

2. Relocation of livestock and import of feed lead to enhanced resource use efficiency in China, but also to pollution swapping.
   (This thesis)

3. Promotion of organic agriculture globally enhances land expansion and related biodiversity losses.

4. Smallholder farms are more resilient to impacts of climate change than relatively large farms in Europe or Americans.

5. Meat eating habits are easier to change than beverage drinking habits.

6. The increasing number of pets threatens global biodiversity.

Propositions belonging to the thesis entitled

‘Unravelling China’s livestock transition: nutrient flows and greenhouse gas emissions’

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Unravelling China's livestock transition: nutrient flows and greenhouse gas emissions

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Unravelling China's livestock transition: nutrient flows and greenhouse gas emissions

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Contents

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CHAPTER 1

General introduction

1.1 Introduction

Livestock production contributed about 17% to the global food calorie consumption and about 33% to the global food protein consumption by humans in 2018 (FAO, 2021). Livestock provides also important additional roles in our society, e.g. draft power, nutrients for crop production and soil fertility build-up, landscape maintenance, and livestock is often used as a buffer against poverty by smallholders (Herrero et al., 2010). However, there are increasing concerns about the environmental impacts of global livestock production, especially in areas with high livestock density (Herrero et al., 2015; Liu et al., 2017; Wang et al., 2018).

Public awareness and recognition of the massive upstream and downstream impacts of livestock production strongly increased following the publication of Livestock's Long Shadow by the Food and Agricultural Organization of the United Nation (Steinfeld et al., 2006). The upstream impacts are related to the competition between food and feed production for natural resources (Table 1.1). Approximately 40% of the global arable land area and 55% of the global permanent pasture area were used to produce feed for livestock production in 2010 (Mottet et al., 2017). Global livestock consumed about 6 billion tons of feed in 2010, about 1/3 of this feed originated from cereals and soybean (Mottet et al., 2017). The increase in soybean production greatly drives land-use changes in Latin America (Mottet et al., 2017), and at the same time has greatly contributed to the rapid development of modern livestock production. The downstream impacts relate to environmental pollution; global livestock production was responsible for 14.5% of global greenhouse gas (GHG) emissions in 2005, or 2 times the total GHG emissions from global crop production in 2010 (Gerber et al., 2013; Xu et al., 2021). Moreover, the global livestock production-consumption chain accounted for over 60% of global ammonia (NH_3) emissions in 2010 (Uwizeye et al., 2020). Further, the livestock sector is a dominant contributor to water pollution in some regions, e.g. China (Strokal et al., 2016).

Table 1-1. Summary of main up-stream and down-stream effects of livestock production at the global scale.

Upstream effects			Down-stream effects		
Indicators	Values	Reference year	Indicators	Values	Reference year
Cereal feed consumption	950 Tg	2018 (1)	GHG emissions	7.1 Gt	2005 (5)
Arable land use	40 M km^2	2019 (2)	NH_3 emissions	11.6 Tg*	2010 (4)
Grassland use	30 M km^2	2019 (3)	Nitrogen losses	65 Tg*	2010 (4)
Synthetic N fertilizer use	55 Tg N	2010 (4)	Phosphorus losses	4.3 Tg	2010 (6)

*Note: * the whole production-supply chain;*

(1) FAO, 2021; (2) Poore and Nemecek, 2018; (3) Chang et al., 2021a; (4) Uwizeye et al., 2020; (5) Gerber et al., 2013; (6) Brownlie et al, 2021.

Livestock production has rapidly increased and livestock production systems have rapidly changed during the last decades, notably in affluent countries and countries with rapidly developing economies. The increase in livestock production is fueled by the increasing demand of livestock products, which is driven by increasing human populations, urbanization and wealth. The rapid changes in livestock production systems are mainly related to technology development and globalization of markets. The rapid increase in livestock production and associated changes in livestock production systems in several regions of the world are known as the ‘livestock revolution’. The term ‘livestock revolution’ was first coined by Delgado et al (1999) to describe the rapid changes in livestock production structure and efficiency, and to advise governments and industry to prepare for this continuing revolution. The livestock sector is largely demand-driven; forecasts indicate that livestock production will double between 2005 and 2050, mainly due to the growing world population and income, especially in developing nations (Alexandratos and Bruinsma, 2012). Evidently, the increasing livestock sector will compete for natural resources with other sectors and likely will create more environmental burdens locally and globally. Many solutions to the global challenges of feeding the 9 billion people in the future lie in the answers to the question of ‘how we understand and manage livestock production systems’ (Herrero and Thornton, 2013).

1.2 Trade-offs of different livestock production systems

The resources demand and environmental effects are different for different livestock categories and livestock production systems. Beef production required about 28 and 11 times more land and irrigation water, respectively, than poultry or pork production per kg of produce in the United States of America (USA) between 2000 and 2010 (Eshel et al., 2014). In addition, GHG emissions were 5 times higher from beef production than from poultry or pork production per kg of produce (Eshel et al., 2014). Variations between livestock production systems (within livestock categories) are mainly related to the large diversity in animal diets, animal breeds, herd management and environmental conditions (Herrero et al., 2013; Mottet et al., 2017; Uwizeye et al., 2020).

Robinson et al (2011) and Herrero et al (2015) categorized livestock production into 4 main production systems: pastoral/agropastoral, mixed extensive systems, mixed intensive systems, and specialized/industrialized systems, based on the integration of livestock production with crops, the relation to land, the agro-ecological zone, the intensity of production, and type of animal product. During the last few decades, global livestock production has largely shifted (i) from extensive mixed crop-livestock production systems towards more intensive and large-scale livestock production system, and (ii) from ruminant animals (beef, sheep and goat) to monogastric animals (pig and poultry), in response to the increasing demand for livestock products, and technological developments (Robinson et al., 2011). This intensification of livestock production, and the shift from ruminant-dominant to monogastric animal-dominant livestock production systems have been responsible for much of the growth in animal calorie and protein production in the world (Herrero et al., 2010). This transition of livestock production has tremendous effects on livelihoods and the environment in many areas of the world (Herrero et al., 2009), which are not well understood yet.

Several studies have suggested that the livestock transition to more efficient monogastric animals and to larger-scale and more intensive production systems provide significant resources and emission savings (Garnett et al, 2013; Herrero and Thornton, 2013; Havlík et al., 2014). However, the concentration of animals in some regions is contributing to a transfer of nutrients from areas where the feed is produced to areas where the feed is consumed by livestock; the latter become hotspots of environmental pollution. The difficulty of analyzing the impacts of the global livestock transition is also related to the multiple impacts livestock production has on the wider environment; the intensification of livestock production has been implicated with land use changes, air pollution, biodiversity losses, deforestation, groundwater depletion, changing nutrient cycles, eutrophication and climate change. For example, Leip et al (2015) estimated that 78% of the terrestrial biodiversity losses, 80% of ammonia and NOx emissions to air, 81% of GHG emissions from agriculture, and 73% of the pollution of water bodies by nitrogen (N) and phosphorus (P) from agriculture in the European Union (EU) were related to livestock production. But they did not assess the differences between different livestock production systems.

Recent global and regional studies have estimated the total biomass use by livestock, and the feed conversion ratio, livestock productivity, nutrient use efficiency and losses, and GHG emissions for different livestock production systems, using the GLOBIOM model (Herrero et al., 2013; Havlík et al., 2014; Chang et al., 2021b), MAgPIE model (Bodirsky et al., 2014; Weindl et al., 2015), and MITERRA model (Velthof et al., 2009; Lesschen et al., 2011). Some of these studies have suggested that the transition of production systems could act as an adaptation strategy to a changing climate in the future (Havlík et al., 2014; Weindl et al., 2015). However, it is questionable whether such suggestions are universally applicable, given the wide diversity in production systems and in socio-economic and environmental conditions, and the possible multiple effects of livestock production systems on the environment (Van Zantenet al., 2019; Van Selm et al., 2022). A better quantitative understanding is needed of the impacts of changes in livestock production on the wider environment, especially in countries undergoing a livestock revolution, and to explore more sustainable options (Herrero and Thornton, 2013).

1.3 Global effects of increasing trade of livestock products and feeds

The livestock revolution leads to the concentration of intensive livestock production systems in areas with good market outlets for livestock products and with adequate and cheap input supplies (feed, technology, advice), i.e., in the vicinity of large cities. This concentration of livestock production near markets provides economic benefits to producers, processing industry, and suppliers. The vertical integration along the land–livestock–food chain creates so-called economies of scope (Haan et al., 1998), which can react fast to growing and changing demands. Concentration of livestock production is associated with increased transport and trade of animal feed and also animal products.

International trade of agricultural products has several possible impacts. For example, it has been indicated that trade of food has contributed to local food security. Many African countries rely on food import to fill the gap between the increasing food demand by the growing human populations and the stagnant and relatively low domestic food production (Rakotoaroa, 2011). Some medium and high-income countries also rely on imports. For example, the United Kingdom imports almost 50% of the required food, as a form of outsourcing (de Ruiter et al., 2016). Oil-rich but water-scarce countries in the south-west Asia also import large amounts of food because domestic food production is not sufficient (FAO, 2021). China has a share of 60% of global soybean trade; this massive import is needed to supply enough protein for its livestock production (FAO, 2021).

However, there are concerns about the impacts of international trade of food and feed on resources use efficiency and environmental pollution. For example, the amounts of N embedded in the traded agricultural products have increased from 3.0 to 24 Tg N between 1961 and 2011, especially through the rapid increase in the trade of feed (Lassaletta et al., 2014). The international trade of food and feed products accounted for 16-20% of the global production from agricultural land (Fader et al., 2013; MacDonald et al., 2015). Groundwater depletion associated with the production of traded food was estimated at 25.6 km^3 in 2011, which was equal to 11% of the global groundwater depletion (Dalin et al., 2017). Further, around 15-25% of total NH_3 emission from the global food production-consumption chain was associated with internationally traded agricultural products in 2011 (Galloway and Allison, 2016; Oita et al., 2016). Emissions of NH_3 contribute to the formation of secondary fine dust particles ($PM_{2.5}$) in the air; about 22% of the total number of premature deaths caused by air pollution were related to $PM_{2.5}$ pollution associated with the international trade of goods and services in 2007 (Zhang et al., 2017). Also, about 17% of global biodiversity loss is due to the production of commodities destined for exportation (Lenzen et al., 2012; Chaudhary and Kastner, 2016). Further, exportation of beef, soybeans, and wood products was held responsible indirectly for deforestation in many countries (Henders et al., 2015).

Positive effects of international trade of agricultural products on resource use have been reported as well. For example, Chapagain et al (2006) and Fader et al (2011) reported that trade of food and feed have contributed to 'water savings' of approximately 263-352 km^3 yr^{-1}, equivalent to 3 to 6% of the annual precipitation in the production regions. It has also been reported that 41 to 90 million ha of land has been 'saved' through international trade of agricultural products, which is equal to 5 to 10% of the sowing area (Fader et al., 2011; Kastner et al., 2014a). However, the uncertainties in these estimates are large and contradictory results between different studies have been found as well (Kastner et al., 2014b). Contradictory estimations of the impacts of agricultural trade are mainly due to the complexity of the trade matrix between different countries, the mixed use of absolute and net trade in studies, and the use of different data, countries and years.

1.4 Understanding nutrient flows in the manure management chain

Roughly between 10 to 40% of the amount of N in feed consumed by livestock is retained in milk, meat and egg, depending on animal species and management. The remainder is excreted in urine and feces, i.e., 60 to 90% of the amount of N in the feed. Total global excretion of N in urine and feces by livestock is about 80-130 Tg N per year, from which only 20-40% is efficiently utilized by pastures and arable crops (Sheldrick et al., 2003; Oenema and Tamminga, 2005; Uwizeye et al., 2020). The remainder of the manure N is emitted to the atmosphere or leached to groundwater and surface waters. However, there are large differences between countries in manure management and utilization, which are related to the livestock production systems (e.g., grazing systems vs mixed crop-livestock systems vs specialized livestock production systems), environmental conditions, and the compliance to good agricultural practices and/or governmental regulations. For example, approximately 65% of excreted N in animal housing is recycled in intensively managed livestock production systems in the EU, due to strict regulations (Oenema et al., 2007). The N recycling efficiency varied from 6 to 99% for manure handling, and from 30-87% for manure storage in extensively managed livestock production systems in Africa (Rufino et al., 2006).

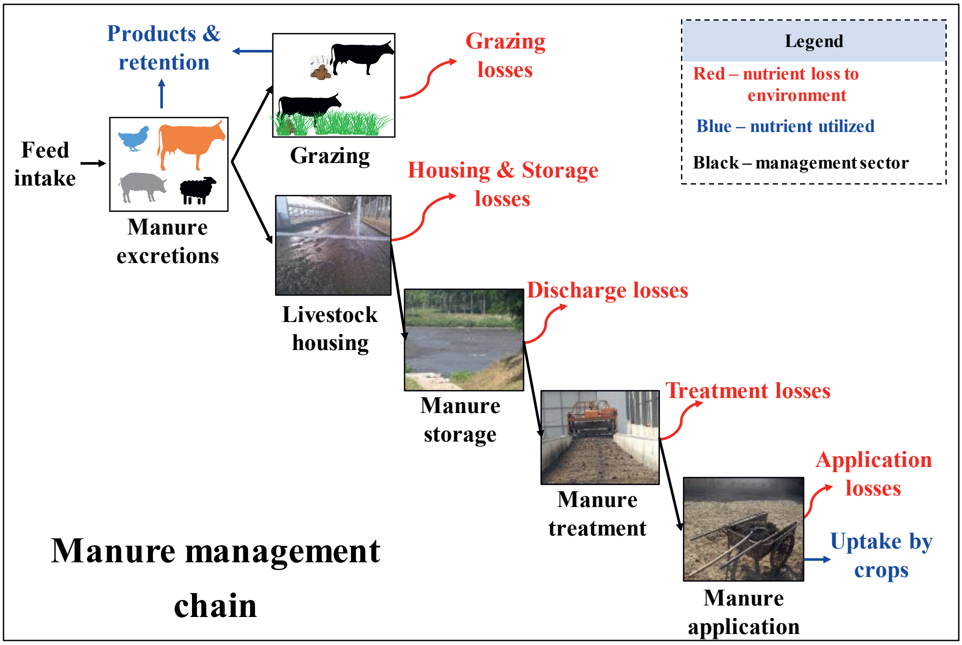

Figure 1.1. The manure management chain, showing schematically the flow, utilization and losses of nutrients from feed intake all the way to manure application to cropland (modified from Hou, 2016).

Losses of N and P from the manure management chain differ between different livestock production systems and manure management systems, and may occur through different pathways (Fig 1.1). Emissions of NH_3 are often a main loss pathway, depending also on production system (Oenema et al., 2007). Discharge of manure from pig production into surface waters bodies was found to be an important pathway in China (Bai et al., 2014). The main N loss pathways depend in part on livestock production systems. For example, N losses from traditional dairy production systems occurred mainly through NH_3 emissions, but through discharge of manure to water bodies in industrial dairy feedlots in China (Bai et al., 2013). Evidently, the main loss pathways and the magnitude of the N and P losses from manure may change rapidly through changes in livestock production systems, improvements in manure management, introduction of manure treatment technology and through implementation of strict governmental regulations. This suggests that there is an almost continuous need for monitoring changes in production systems, manure treatment and manure management practices, and their effects on N and P losses and recycling efficiency (Fig 1.1). However, there is often a scarcity of accurate measurements of N and P losses in practice, which indicates that approximations have to be made, using assumption and emission factors.

1.5 Changing livestock production in China and their uncovered impacts

Total livestock production has greatly increased and livestock production systems have greatly changed in China since 1980, in response to the increasing demands for animal-source food that resulted from the introduction of the 'open-door' policy, the economic development, and the subsequent increased prosperity of the increasing human population, especially in urban areas (Li et al., 2008). The average meat, milk and egg consumption per capita in China increased by 3.9, 10 and 6.9 times, respectively between 1980 and 2010, which were by far the largest increases during this period in the world (FAO, 2021).

In the 1990s, China exceeded USA as largest livestock producer in the world (FAO, 2021). Currently, 18% of the global number of livestock units (LSU) are housed in China, equivalent to 500 million LSU in 2010 (Liu et al., 2017). The increased consumption of animal-source food has contributed to an improved nutrition level of Chinese people, especially children, although there is also an increased incidence of diseases related to excessive consumption of animal-source food (Du et al., 2002). Meanwhile, the production of livestock manure has also increased greatly during the last decades. These manures contain huge amounts of carbon and nutrients, including N, P, potassium (K), and micronutrients which are essential for crop growth and for soil fertility maintenance and build-up. However, these huge amounts of manure are not managed properly, and thereby create severe environmental problems (Strokal et al., 2016; Zhao et al., 2017; 2019; Jin et al., 2019).

Manure is poorly managed due to the loose environmental regulations and the lack of knowledge and appropriated technology for conducting careful manure management. As a result, manure has become a main contributor to environmental pollution in China, through ammonia and greenhouse gas emissions to air, and leaching of nutrients to water bodies (Strokal et al., 2016; Zhang et al., 2017). Over 50% of the total ammonia emissions originated from livestock production in 2015 (Zhang et al., 2017). These emissions were responsible for the worsening of the air quality in recent decades. Estimates indicate that manure contributed over two-thirds of the nutrients in the northern rivers and 20%–95% of the nutrients in the central and southern rivers in China (Strokal et al., 2016). Changes in livestock production systems have contributed to an increase of the N use efficiency (NUE) of pig production at animal level, but to a decrease of NUE at farm level, from around 50% in 1960 to 10% in 2010 (Bai et al., 2013; 2014). Moreover, the spatial distribution of livestock across China has changed since 1980, with more livestock production in the coastal regions. The high livestock density creates severe environmental pollution in especially North China Plain and Yangtze River Delta (Ma et al., 2015; Lin et al., 2018).

On average only a small proportion of manure N is recycled back to cropland in China (Ma et al., 2010; 2012). Manure was once the main source of nutrients for crop production, and deemed as a precious product for food security (Cui et al., 2013). The livestock revolution has contributed to major changes in the way manure is perceived currently. The large specialized livestock systems are spatially decoupled from the specialized crop production systems, and thereby have great difficulty with the disposal of manure on crop land in agronomically and environmentally sound ways (Li et al., 2008; Jin et al., 2021). Meanwhile, Chinese government strongly promoted the use of synthetic fertilizers, for food security reasons. In 2010, the fertilizer subsidy program totaled US$18 billion (Li et al., 2013). Consequently, China became the biggest fertilizer producer and user in the world (FAO, 2021). Synthetic fertilizer largely substituted manure in crop production between 1970 and 2015, and has blocked the proper use of manure. To improve the situation, manure must be recycled in cropland and must replace synthetic fertilizer as nutrient source. However, there is little information about the degree to which manure can replace synthetic N, P and K fertilizers, and about the associated changes in N, P and K losses to the environment in China.

The changes in livestock production in China and its impacts have not been examined in detail yet (e.g., Li et al., 2008). As a result, there is limited understanding of the impacts of the livestock transition. China is an interesting case in reflecting the livestock revolution, which occurs also in many other rapidly developing countries in the world, because of the rate and size of the changes. It has been suggested that the livestock revolution is largely demand driven (Delgado et al., 1999), but it appears that governmental interventions have also played a significant role in China. There are also large regional differences in the development of the livestock sector, which are not well understood from a demand-driven perspective. Moreover, the demand for animal products is projected to increase further (Alexandratos and Bruinsma, 2012). As a result, livestock production is projected to double during the next few decades in China. Such changes seriously question the sustainability of future livestock production in China, and the feasibility of the projections. Thus, there is an increasing need to better understand the causes and effects of the livestock transition for different livestock categories and production systems. Also, the perspectives of future livestock production in China need to be explored in greater detail.

1.6 Objectives

The main objective of the research described in this thesis is to increase the understanding of the livestock transition in China and its impacts on livestock production, manure management, nutrient cycling and the environment. The specific objectives of this thesis are to:

- Develop and apply a method for a systematic analysis of the effects of international trade of food and feed during the last five decades on global land use, fertilizer N use and feed use efficiencies and livestock productivity (**Chapter 2**);

- Analyze the livestock transition in China and its driving forces, impacts and possible future consequences at national and international levels (**Chapters 3 and 4**);
- Analyze the N, P and K flows and losses in the ‘feed-animal-manure-cropland’ chain for the various livestock production systems in China, and to explore options for increasing nutrient use efficiency and recycling **(Chapter 5)**;
- Analyze the effects of the spatially uneven distributions of livestock production in China on nutrient recycling efficiency, and to explore the effects of spatial planning options for crop-livestock integration on nutrient recycling efficiency (**Chapters 6 and 7**).

1.7 Outline of the thesis

This thesis contains a general introduction (Chapter 1), six research chapters (Chapters 2 to 7) and a general discussion (Chapter 8). The coherence between different chapters are elucidated in Fig 1.2.

In Chapter 2, I developed and applied a new method for a systematic quantification of the complicated impacts of international trade of food and feed on the efficiencies of global crop and livestock production, in terms of energy and protein production, using statistical data over the period from 1961 to 2013.

In Chapter 3, I conducted a comprehensive analysis of the driving forces of the livestock transition in China between 1980 and 2010, and quantified the effects of this transition on domestic and global animal-source food provisioning, resource use, N and P losses, and greenhouse gas GHG emissions. The causes and effects of the changes in livestock production for different livestock categories, and the perspectives of future livestock production are discussed. Scenarios for 2050 explore the effects of a range of development pathways.

In Chapter 4, I examined the impacts of increasing domestic milk production versus increasing importation of milk from abroad on GHG emissions, N losses, land and water use, and economic performances across the main feed and milk producing countries in the world. The interrelationships and interdependencies of the whole 'production-consumption-trade' system were examined for contrasting socio-economic and technological pathways for the year 2050.

In Chapter 5, I analyzed the manure N, P, and K flows and losses in the 'feed-animal-manure-cropland' chain in China for different animal categories and production systems for the year 2010 using a modified version of the NUFER model. In addition, five scenarios were explored to assess the potentials for reducing manure N, P, and K losses and for replacing fertilizer inputs by manure nutrients in cropland by 2020.

In Chapter 6, I explored the potentials of the manure management systems in China to keep N and P use within derived 'planetary boundaries' for China. The impacts of the uneven spatial distribution of livestock production on N and P loss targets were quantified for different mitigation measures.

In Chapter 7, I explored pathways for improved spatial planning of livestock production aimed at minimizing the negative effects of concentrations of livestock production, using two contrasting criteria for optimization of the spatial planning, i.e., crop-livestock integration with increased manure recycling, and the exposure of humans to ammonia emission density, at county level (note, there are about 2800 counties in China).

Chapter 8 provides the general discussion and overall synthesis of my research.

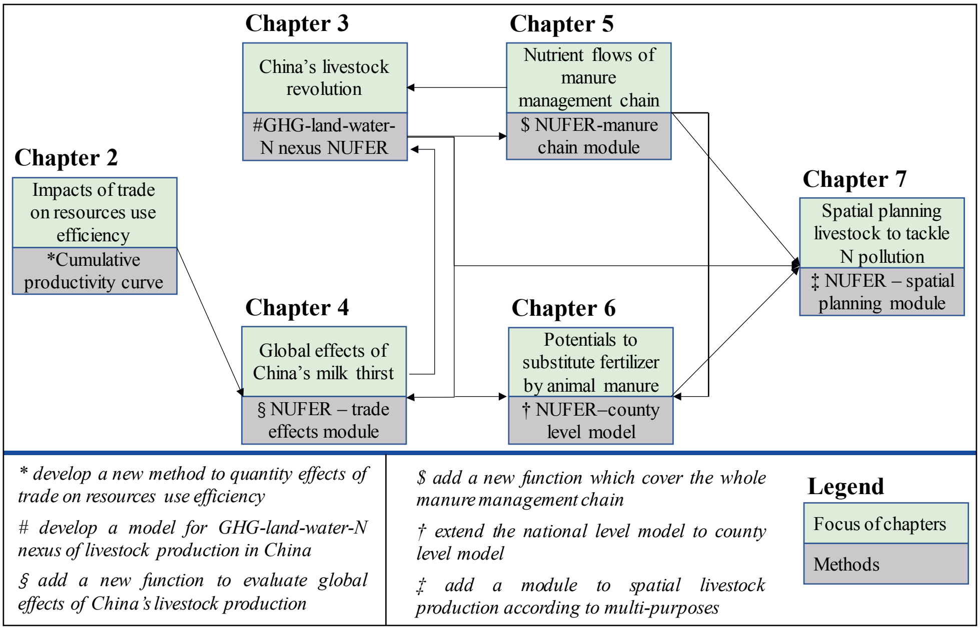

Figure 1.2. An overview of structure and coherence of the research chapters of my PhD thesis. NUFER is the abbreviation of the model 'Nutrient flows in Food chains, Environment and Resources use' (Ma et al., 2010).

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CHAPTER 2

2

Food and feed trade has greatly impacted global land and nitrogen use efficiencies over 1961–2017

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Abstract

International trade of agricultural products has complicated and far-reaching impacts on land and nitrogen use efficiencies. We analyzed the productivity of cropland and livestock and associated use of feed and fertilizer efficiency for over 240 countries, and estimated these countries' cumulative contributions to imports and exports of 190 agricultural products for the period 1961–2017. Crop trade has increased global land and partial fertilizer nitrogen productivities in terms of protein production, which equaled savings of 2,270 Mha cropland and 480 Tg synthetic fertilizer nitrogen over the analyzed period. However, crop trade decreased global cropland productivity when productivity is expressed on an energy (per calorie) basis. Agricultural trade has generally moved towards optimality, that is, has increased global land and nitrogen use efficiencies during 1961–2017, but remains at a relatively low level. Overall, mixed impacts of trade on resource use indicate the need to rethink trade patterns and improve their optimality.

2.1 Introduction

Concerns are increasing about the need to provide enough nutritious food for a growing global population within environmental limits (Erb et al., 2016). International trade in food and feed makes substantial contributions to local food security and has rapidly increased during recent decades (Falkendal et al., 2021). However, this trade also has complex impacts on water use (Dalin et al., 2017), biodiversity (Lenzen et al., 2012), air quality (Zhang et al., 2017), land use (Yu et al., 2013; Kastner et al., 2014) and climate change (de Ruiter et al., 2016; Scheelbeek et al., 2020). Currently, many African countries rely on food imports to fill the gap between increasing food demand and lagging domestic food production (Rakotoarisoa et al., 2011). Some medium- and high-income countries also require food imports; for example, the United Kingdom imports almost 50% of its food supply and increasingly relies on vegetable imports from climate-vulnerable countries (de Ruiter et al., 2016; Scheelbeek et al., 2020), while China is the largest importer of soybean to support its domestic livestock industry and vegetable oil demand (Bai et al., 2018; FAOSTAT, 2020). Hidden resource depletion and the environmental impacts associated with food and feed trade across country borders have been the subject of debate. Groundwater depletion by products used for export was reported to be equivalent to 11% of total global ground water depletion in 2011 (Dalin et al., 2017). Around 15–25% of global ammonia emissions associated with food production originate from internationally traded food products (Falloway and Leach, 2016; Oita et al., 2016) and the proportion of reactive nitrogen losses embedded in the trade of feed and livestock products is high (Uwizeye et al., 2020). However, these studies mainly focused on the impacts of trade on exporting countries, with little emphasis on the distributions of production efficiencies of exporting versus importing countries. Some studies have considered productivity differences between exporting and importing countries, but found contradictory results for the impact of trade on land use efficiency (Fader et al., 2011; Fader et al., 2013; Kastner et al., 2014). Two studies have used multiregional input–output data to investigate how global trade of all commodities contributes to the externalization of some environmental impacts (Wood et al., 2018; de Boer et al., 2019). Global land and nitrogen use efficiencies are important elements for achieving the United Nations' Sustainable Development Goals (Sachs et al., 2021), but information about the impacts of food and feed trade on global land and nitrogen use efficiencies is still limited. There is also little information available about the optimality of trade, specifically improving global land and nitrogen use efficiencies, that is, whether high-efficiency countries export to low-efficiency countries, and its variability in terms of land and nitrogen use efficiencies at the global level. In this article we aim to develop and use a systematic method to quantify the impacts of food and feed trade on global land and nitrogen use efficiencies, and to determine the non-monetary optimality of trade and changes at the global level over the period for which FAOSTAT data are available (1961–2017) (FAOSTAT, 2020). Global land and nitrogen use efficiencies were defined in terms of productivities. Four main productivity parameters were selected to assess the impacts of trade on global land and fertilizer nitrogen use efficiencies: (1) cropland productivity, (2) partial fertilizer nitrogen productivity in crop production, (3) livestock productivity and (4) partial feed nitrogen productivity in livestock production (see Methods and Table 2.1). These parameters have been used to develop productivity distribution curves, separately for importing and exporting countries, and two indicators that describe the essential features of these curves: the concentration of production in high-efficiency countries (CPHE), a dimensionless indicator describing the inequality in a given group of countries (high when productivity and production are both high in very few countries); and the concentration-weighted production efficiency (CWPE), representing the CPHE-adjusted productivity for a given group of countries.

Table 2.1. Indicators used to assess the impacts of trade on global resource use efficiencies.

Indicators	Unit	Interpretation	Equations
Cropland productivity-calorie Cropland productivity-protein	kcal ha^{-1} yr^{-1} kg protein ha^{-1} yr^{-1}	Cropland productivity, expressed as: (1) crop calories produced per hectare per year; and (2) crop protein produced per hectare per year.	Equation 2
Partial fertilizer nitrogen productivity-calorie Partial fertilizer nitrogen productivity-protein	kcal (kg fertilizer $N)^{-1}$ yr^{-1} kg protein (kg fertilizer $N)^{-1}$ yr^{-1}	Partial fertilizer nitrogen productivity in crop production, defined in terms of: (1) crop calories produced per kilogram fertilizer nitrogen applied per year; and (2) crop protein produced per kilogram fertilizer nitrogen applied per year. Note: nitrogen input to crop production via manure nitrogen, deposition and biological nitrogen fixation was not considered.	Equation 4
Livestock productivity-calorie Livestock productivity-protein	kcal LSU^{-1} yr^{-1} kg protein LSU^{-1} yr^{-1}	Livestock productivity, defined in terms of livestock production, and expressed as: (1) animal-source calories produced per LSU per year; and (2) animal-source protein produced per LSU per year.	Equation 3
Partial feed nitrogen productivity-calorie Partial feed nitrogen productivity-protein	kcal (kg feed $N)^{-1}$ yr^{-1} kg protein (kg feed $N)^{-1}$ yr^{-1}	Partial feed nitrogen productivity of livestock production, expressed in terms of: (1) animal-source calories produced per kilogram of feed protein nitrogen per year; and (2) animal source protein per kilogram of feed protein nitrogen consumed per year.	Equation 5

2

2.2 Material and Methods

2.2.1 Cumulative productivity distribution curve

The cumulative productivity distribution curve was developed to quantify the relative concentration of production in high-efficiency countries, and to evaluate trade optimality and functionality. The idea of this curve originates from the Lorenz curve, but is applied in a different way. We plotted each country in the world on the x axis in ascending order of productivity (for one product or for a combination of products). This is different from the Lorenz curve because our aim is to quantify the relative concentration of production of a certain product (or combination of products) in high-efficiency countries. The contribution of each country to the total global production of a commodity was plotted on the *y* axis (%). Then the cumulative productivity distribution curve was estimated.

Definition and estimation of CPHE. The relative concentration of production in high-efficiency countries (CPHE) was defined by area *A* over areas *A*+*B* in Fig 2.1a, that is, CPHE=*A/(A+B)*. A hypothetical value of CPHE=1.0 indicates that the most productive country in the world contributes 100% to the global production. A CPHE=0.50 indicates that productivity was equally distributed over low- and high-productivity countries.

The cumulative productivity distribution curves were approximated by piecewise-defined continuous and non-negative functions *f(x),* that is,

$$(x) = \begin{cases} f_1(x), a \le x \le a_1 \\ f_2(x), a_1 \le x \le a_2 \\ \vdots \\ f_n(x), a_{n-1} \le x \le b \end{cases}$$

where $[a,b]=[a,a_1]\cup[a_1,a_2]\cup\ldots[a_{n-1},b]$, and the functions $f_i(x)$ *,i=1,2, ...,n,* can be either a polynomial function or a logarithmic function.

Based on the simulation curve, we calculated the area following the definite integral method. The interval on the x axis between the minimum productivity and maximum productivity was denoted as [a, b]. The area below the graph of f over [a, b] was denoted as B. Then the area B is given exactly by the sum of the definite integrals of f_i over the corresponding subintervals, that is,

$$B = \int_{a}^{a_1} f_1(x)\,dx + \int_{a_1}^{a_2} f_2(x)\,dx + \cdots + \int_{a_{n-1}}^{b} f_n(x)\,dx$$

It is straightforward to check that the area of A + B is a rectangle with length $x_{max} - x_{min}$ and width $y_{max} - y_{min}$, where x is the productivity and y is the cumulative production. Therefore, the area A is the difference between the area $A + B$ and the area B. Areas A and B are sensitive for extreme low- and high-productivity values; hence very low- and very high-productivity countries with a low contribution（<1.0%）to the total production or trade were excluded. These extreme values may relate to statistical errors or to highly specific conditions. The impacts of the maximum productivity on CPHE are illustrated in Fig S1. We have also tested the sensitivity of potential resource savings to the selection of maximum productivity, when set at 98.5%, 99.0% and 99.5% contribution to the total production, and show that a 99.0% contribution presented the best value (Litchfield, 1999; Cobham and Sumner, 2013).

Definition and estimation of the CWPE. The CWPE represents a CPHE-corrected productivity of a given product. It was calculated as follows:

$$\text{CWPE} = \text{CPHE} \times \text{area}_{\text{rectangle}}$$

where the unit of the CWPE depends on the unit of productivity on the x axis, and

$\text{area}_{\text{rectangle}}$ represents the area of the rectangle (areas A+B), of which the length is from 0 to maximum productivity on the x axis and the height is from 0% to 100% contribution on the y axis (Fig 2.1a). Hence, $\text{area}_{\text{rectangle}}$ is equal to the maximum productivity multiplied by 100%, and basically equal to the maximum productivity. The CWPE is positively correlated to average productivity. In a few extreme situations the CWPE may equal the average productivity of given products across the world. For example, the CWPE may equal the maximum productivity when CPHE = 1.0 because only the highest-productivity country produces all the products.

Relationship between CPHE and CWPE. The CPHE and CWPE are interrelated because they both share the same cumulative distribution curve; a high CPHE usually means a high CWPE, and vice versa. Relationships between CPHE and CWPE vary when the maximum productivity (or partial fertilizer or feed productivity) varies, as follows from Fig S9.

Trade optimality and functionality. We applied the concepts of CPHE and CWPE to importing and exporting countries to estimate the functionality and optimality of the international trade of food and feed commodities at the global level (Fig 2.2). The indicator was estimated for both importing and exporting countries. International trade was considered 'functional' when the CPHE of exporting countries ($CPHE_{ex}$) was larger than that of importing countries ($CPHE_{im}$) and also >0.50, and trade was deemed as optimal when $CWPE_{ex}/CWPE_{im}$ was >1.0 (Fig 2.2; S1).

2.2.2 Agricultural production and trade data

We used data from the FAOSTAT statistical database to analyze worldwide crop and animal productivity distributions and trade efficiency distributions. In total, 164 crop products and 26 animal products from six main animal categories from >200 countries were selected for this study (Table S4). Cropland, livestock and partial fertilizer nitrogen productivities were expressed as described in Table 2.1.

2.2.3 Productivity indicators.

Global land and nitrogen use efficiencies were defined in terms of productivities. Four main productivity parameters were selected to assess the impacts of trade on global land and fertilizer nitrogen use efficiencies. (1) Land use efficiency was expressed in terms of 'cropland productivity', that is, the summed annual calorie (or protein) harvest of all crops in a country divided by the total harvested area of cropland in that country (Renard and Tilman, 2019). (2) Partial fertilizer nitrogen productivity in crop production was defined as annual total crop yield, in terms of energy (or protein) per kg of mineral fertilizer nitrogen applied in a country (Table 2.1). Hence, only the new nitrogen input via synthetic fertilizer was considered in the estimate of partial fertilizer nitrogen productivity, which gives an upper estimate because it neglects the nitrogen inputs via biological N_2 fixation and via recycling of manure, crop residues and net soil organic matter mineralization. (3) Livestock productivity was defined as annual total livestock production, in terms of energy (or protein) per LSU in a country. (4) Partial feed nitrogen productivity in livestock production was defined as total livestock production, in terms of calorie (or protein) per kg of feed nitrogen used in a country (Table 2.1). Hence, cropland and livestock productivities and partial fertilizer nitrogen and feed nitrogen productivities were evaluated both in terms of energy (calories) and protein because of their important but different roles in food security, trade and environmental impacts.

2.2.4 Crop and livestock productivity

Crop productivity. A weighted mean productivity of crop products per country was used in this study:

$$Crop\ productivity = \frac{\sum calorie\ or\ Protein_{crop\ product\ i}}{\sum harvested\ area_{product\ i}} \quad (1)$$

where 'cropland productivity' (Table 2.1) is the average calorie production per ha (or average protein production per ha) of all crops within a country, expressed in kcal ha^{-1}, or kg protein ha^{-1}; $\sum$calorie or protein $_{\text{product i}}$ is the sum of calorie or protein production of the harvested crop products per country per year, expressed in kcal or kg protein; and $\sum$harvested area $_{\text{product i}}$ is the sum of the harvested area of the crop species in a country in a year, expressed in ha. In addition, the productivity of single crops was also calculated based on their harvested areas, production quantities, and calorie and protein contents.

Livestock productivity. For livestock products, we calculated the average productivity per LSU, using the total production quantities, animal numbers and the calorie and protein contents of animal products. The livestock number was transferred to standard LSUs, following the coefficients used by Liu et al (2017)..

$$Livestock\ productvity = \frac{\sum calorie\ or\ Protein_{livestock\ product\ i}}{\sum Livestock\ unit_{product\ i}} \quad (2)$$

where 'livestock productivity' is the average calorie or protein production per LSU in a country, expressed in kcal LSU^{-1} or kg protein LSU^{-1} ; $\sum$calorie or protein $_{\text{livestock product i}}$ is the sum of the calories or proteins produced by all livestock categories in a country, expressed in kcal or kg protein per year; and $\sum$livestock unit $_{\text{product i}}$ is the sum of animal numbers, expressed in LSUs. Here, six livestock categories (pigs, layer hens, broilers, beef cattle, dairy cattle, sheep and goats) were considered; they accounted for 99% of total animal products trade in 2017 (Table S4). The calorie and protein contents and protein/nitrogen transfer index for each crop product and livestock product were derived from the literature (Mueller et al., 2012; Sutton et al., 2013; Eshel et al., 2014; Mottet et al., 2017).

Partial fertilizer nitrogen productivity. The average calorie or protein production per unit of fertilizer nitrogen input was used to quantify the partial fertilizer nitrogen productivity in crop production. The partial fertilizer nitrogen productivity only considered the inputs from mineral nitrogen fertilizer, and not the inputs from, for example, biological N_2 fixation, atmospheric nitrogen deposition or recycled nitrogen from animal manures, crop residues and composts, or the net mineralization of soil organic matter

$$PFP_{\text{crop}} = \frac{\sum \text{calorieor protein}_{\text{cropproduct i}}}{\text{fer}tilizer} nitrogen \quad (3)$$

where PFP_{crop} is the partial factor productivity of applied fertilizer nitrogen, or the average crop calorie or protein production per kg fertilizer nitrogen in a country, expressed in kcal kg N^{-1} or kg protein kg N^{-1}; and 'fertilizer nitrogen' is the total fertilizer nitrogen input in crop production, expressed in kg N. Fertilizer nitrogen inputs were derived from the Inputs Module of the FAOSTAT database (Table S5), and were corrected for the amount of fertilizer nitrogen used on grassland, following Lassaletta et al (2014a). We corrected for the estimated fertilizer nitrogen use in the Netherlands and New Zealand, because of the large share of fertilizer nitrogen use for managed grass production. However, estimated fertilizer nitrogen use in cropland is relatively uncertain for some countries. It should be noted that the partial fertilizer nitrogen productivity is an upper estimate of the actual fertilizer nitrogen use efficiency; partial fertilizer nitrogen productivity was used here mainly to show the applicability of our method and the relative differences between importing and exporting countries.

Partial feed nitrogen productivity. The partial feed nitrogen productivity in livestock production was estimated based on mass balance method as follows:

$$PFP_{\text{livestock}} = \frac{\sum \text{calorie or protein}_{\text{livestock product i}}}{\sum nitrogen_{product\,i} + \sum nitrogen_{\text{manure excretions i}}} \tag{4}$$

where the $PFP_{livestock}$ is the partial factor productivity of feed nitrogen, or the average animal-source calorie or protein produced per kg feed nitrogen in the livestock production sector in a country, expressed in kcal kg N^{-1} or kg protein kg N^{-1}; $\sum nitrogen_{product\ i}$ is the sum of nitrogen in the livestock products for the six livestock categories selected, expressed in kgN. Information about products of different livestock categories were derived from the Livestock Yield database of FAOSTAT (Table S5); and $\sum nitrogen_{manure\ excretions\ i}$ is the sum of manure nitrogen excreted by six livestock categories, expressed in kg N. Information about manure nitrogen excretions of different livestock categories was derived directly from the FAOSTAT database using the category of Agri-Environmental Indicators (Table S5).

Annual import and export of agricultural products. Since some countries import/ re-export certain products, such as soybeans and bananas, we used net food import and net export per food category from the FAOSTAT database (Table S5), combined with data on protein content and protein/nitrogen conversion factors, to calculate the annual nitrogen import and export for each food category in countries, and the share of each country/region to the global total import and export. Hence there is no need to quantify the import and re-export issue, or the different final use of a product, because we are using the net trade and convert all products to calorie or protein content. We used the recently updated (February 2020) trade data from Commodity Balance Module of FAOSTAT (Table S5).

2.2.5 Effects of trade on land and resources use

The effects of trade on global cropland productivity were estimated from the differences in the CWPE of exporting countries and importing countries. Potential saving of cropland through international trade was defined as:

$$\text{land}_{\text{saving}}\,or\ \text{wastage} = \sum \frac{\text{crop}_{\text{importi}}}{productivity_{\text{importi}}} - \sum \frac{\text{crop}_{\text{exporti}}}{productivity_{\text{exporti}}} \tag{5}$$

where $\text{land}_{\text{saving or wastage}}$ is the potential saving or wastage of cropland (in ha) through the trade in crop products; $\text{cropi}_{\text{mport i}}$ and $\text{crop}_{\text{export i}}$ are net import or export of crop products in a given net import or export country, respectively, expressed in kcal or in kg protein; and $\text{productivity}_{\text{import i}}$ and $\text{productivity}_{\text{export i}}$ are the national crop productivity of a given net import or export country, respectively, expressed in kcal or in kg protein. The evaluation of the impacts of trade of food and feed on the saving or wastage of livestock number, fertilizer nitrogen and feed nitrogen followed the same calculation method as presented above for cropland saving or wastage.

2.3 Results

2.3.1 A new analytical framework.

2.3.1.1 Cumulative productivity distribution curves to quantify the impacts of trade

A cumulative productivity distribution curve for all countries in the world was developed to quantify the concentration of agricultural production in high-productivity countries. This curve is derived from the Lorenz curve and the Gini coefficient (Atkinson, 1970; Piketty and Saez, 2003), which have been widely used to quantify the degree of inequality in the distributions of income and natural resources. We constructed the curve by plotting each country on the x axis in ascending order of commodity productivity (Fig 2.1a), while the contribution of each country to the total global production of a commodity was plotted on the y axis (%). The cumulative productivity distribution curve of a commodity divides the graph into two parts, namely: area A (dark green) lying between the y axis, the 100% contribution line and the cumulative productivity distribution curve; and area B (light blue) lying between the x axis, the maximum productivity line (max x) and the cumulative productivity distribution curve (Fig 2.1a)

2.3.1.2 Evaluation of trade functionality and optimality

We used two complementary indicators CPHE and CWPE, to assess the impacts of international trade on cropland and livestock productivities and partial fertilizer nitrogen and feed nitrogen productivities; these indicators stem from the cumulative productivity distribution curve developed in this study. CPHE is area A divided by areas A+B in Fig 2.1a. CPHE ranges from 0 to 1; a relatively high value indicates concentration of production in few high-efficiency countries (Fig 2.1a). CWPE is CPHE multiplied by areas A+B (max x in Fig 2.1a); CWPE ranges from minimum to maximum productivity in a few extreme situations but differs from average productivity (Fig 2.1a). Based on differences in the CPHE and CWPE of net importing and net exporting countries (Fig 2.1b), we developed a scheme for trade functionality and trade optimality. Trade was considered functional when the CPHE of exporting countries ($CPHE_{ex}$) was >0.50 and the CPHE of importing countries ($CPHE_{im}$) was 50% of that commodity is exported by relatively high-efficiency countries, and >50% of that commodity is imported by relatively low-productivity countries. Trade of a commodity is considered near-optimal when exporting countries have a higher CWPE than importing countries; this reflects that goods are transferred from areas of high to areas of low productivity. Conversely, trade was considered less optimal when $CWPE_{ex} < CWPE_{im}$; and trade was considered less functional when $CPHE_{ex} < 0.50$ and $CPHE_{im} > 0.50$ (Table S1). There are eight possible combinations of $CPHE_{ex}$, $CPHE_{im}$, $CWPE_{ex}$ and $CWPE_{im}$, as presented in Fig 2.2 and Fig S1. These eight combinations were categorized into two groups: an 'optimal' group (levels I–IV) (Fig 2.2a), and a 'non-optimal' group (levels V–VIII) (Fig 2.2b). Hence, trade optimality increases when the ratio of $CWPE_{ex}/CWPE_{im}$ increases, and trade functionality increases when the ratio of $CPHE_{ex}/CPHE_{im}$ increases (Fig 2.2a,b)

2.3.1.3 Potential saving or wastage of resources through trade

The framework allows the effects of trade on a potential saving or wastage of resources (that is, cropland, livestock unit, fertilizer nitrogen, feed nitrogen) to be estimated on a global scale, that is, based on the average productivity and total calorie or protein trade between exporting and importing countries, relative to the status in the absence of this trade. Such a comparison implicitly assumes that sufficient cropland (and other resources, such as labor, water and nutrients) would exist in importing countries (in the hypothetical situation without trade), and that the fraction of imported commodities would be produced additionally at the same productivity level as that of the existing domestic production. However, many importing countries face great shortages of cropland (and possibly other resources), which is a key driver for import of food and feed, such as in the case of China, Japan and the Netherlands (Hayami and Yamada, 1991; Liu et al., 2019). Hence, possible savings or wastage of resources may be lower than the potential values estimated here.

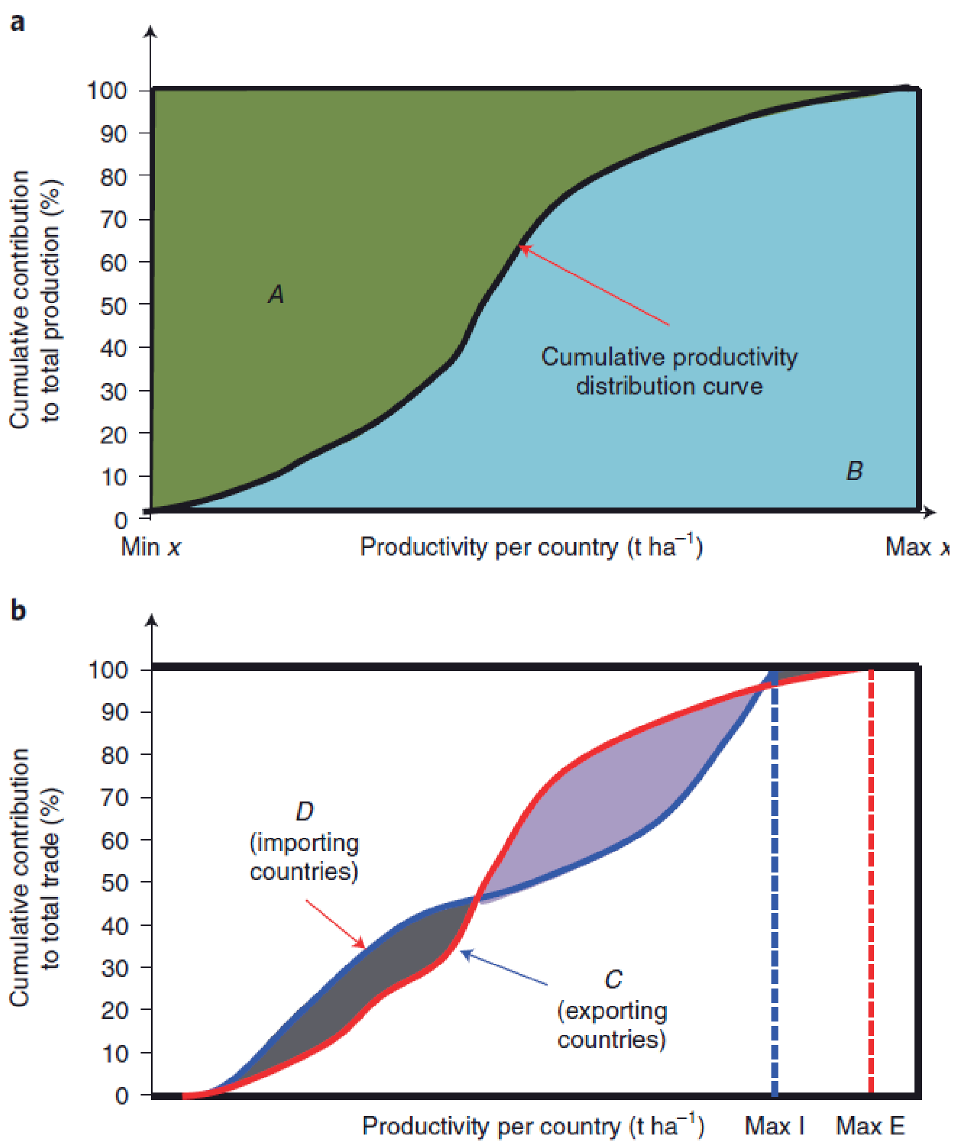

Figure 2.1. Productivity distribution curves. a, Illustration of the concept of a relative concentration of high-productivity countries in the world (CPHE=A/(A+B)). b, Illustration of the concept of CPHE applied to exporting and importing countries separately to evaluate global trade functionality and optimality (see Fig 2.2). *Countries were plotted on the x axis in ascending order of productivity. Max I is the maximum productivity for importing countries; Max E is the maximum productivity for exporting countries.*

2.3.2 Impacts of trade on resources during 1961–2017

Global cropland productivity. The impact of international trade of food and feed on cropland productivity was estimated from the total trade in crop products, and the difference between the $CWPE_{ex}$ and $CWPE_{im}$ for these products. The mean CWPE of crop production was 10.5×10^6 kcal ha^{-1} in net exporting countries and 11.2×10^6 kcal ha^{-1} in net importing countries during the past 57 years (Fig 2.3a). This suggests that crop products were exported from relatively low-productivity countries to relatively high-productivity countries in terms of crop energy production, which implies a potential decrease of global cropland use efficiency. The associated cumulative potential wastage of cropland due to international trade was 870 Mha when adding up areas each year over the period 1961–2017 (Fig 2.4).

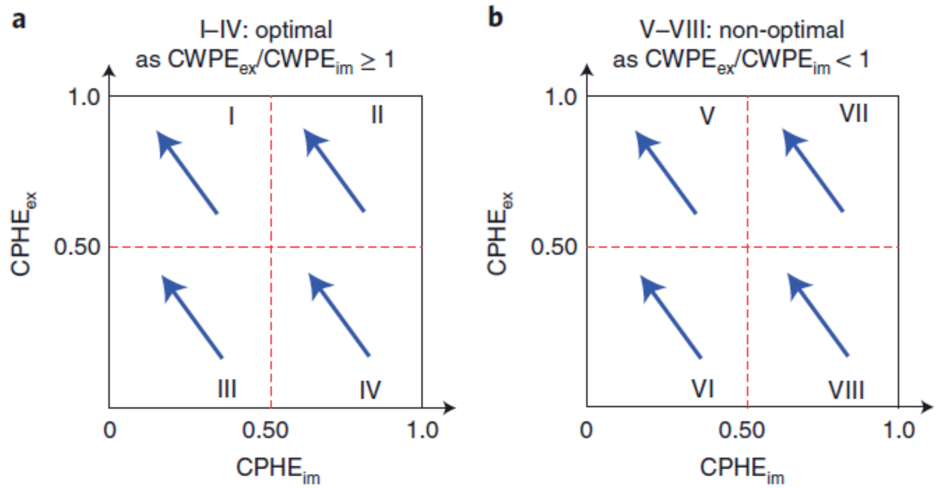

Figure 2.2. Illustrations of the concept of trade functionality and optimality, as determined by the CPHE and CWPE of exporting and importing countries. *Trade is defined as functional when $CPHE_{ex}>0.5$ and $CPHE_{im}<0.5$; it increases as the ratio of $CPHE_{ex}/CPHE_{im}$ increases. An optimal trade ($CWPE_{ex}/CWPE_{im}\geq1.0$) combined with a high trade functionality ($CPHE_{ex}/CPHE_{im}\geq1.0$) is associated with potential improved resource use efficiency at the global level (see Table S1 for further details). The optimality level of trade decreased in the order of I>II>III>IV>V>VI>VII>VII. The arrows represent the direction of increasing trade functionality in each quadrant. CPHE is*

the relative concentration of production in high-productivity countries applied to importing and exporting countries ($CPHE_{im}$ and $CPHE_{ex}$; dimensionless). CWPE is the weighted production efficiency, applied to importing and exporting countries ($CWPE_{im}$ and $CWPE_{ex}$; the unit of CWPE depends on the unit of the x axis; see Fig 2.1).

The potential wastage of cropland was on average 15 Mha of harvested area per year between 1961 and 2017. For comparison, the total area of cropland was 1,500 Mha in 2017 (FAOSTAT, 2020), hence the potential cropland wastage was of the order of 1% of the global cropland area. The gap between $CWPE_{ex}$ and $CWPE_{im}$ has been reduced from -3.80×10^6 kcal ha^{-1} in the 1960s to -0.16×10^6 kcal ha^{-1} in the 2010s, indicating that the potential negative effect of trading crop products on global cropland productivity has decreased over time (Table S2), an effect that was not fully compensated by the stark increase in trade volumes. Overall, potential wastage of cropland decreased, from 36 Mha harvested area each year in the 1960s to 4.9 Mha harvested area each year in the 2000s (Fig 2.5b).

In contrast, the $CWPE_{ex}$ was 36% larger than the $CWPE_{im}$ when cropland productivity was expressed in terms of crop protein production (Fig 2.3b). This indicates a potential increase in global cropland use efficiency through trade, as traded crop products were transferred from high-productivity to low-productivity countries. The cumulative potential saving of cropland through trade was about 2,270 Mha of harvested area between 1961 and 2017 (Fig 2.4). This equals a potential saving of on average 40 Mha of harvested area per year, which is equivalent to about 2.7% of the global cropland area in 2017. The average potential saving of cropland increased from nearly zero in the 1960s to 84 Mha yr^{-1} in the 2010s (note that only seven years were included in the data for the 2010s), which reflects an increasing gap between $CWPE_{ex}$ and $CWPE_{im}$ for crop protein productivity between the 1960s and the 2010s (Fig 2.5c,d). The average annual potential saving of cropland in the 2010s was 5.6% of the global cropland area (FAOSTAT, 2020).

2.3.3 Global livestock productivity

International trade of livestock products was from high-efficiency countries to low-productivity countries—because the $CWPE_{ex}$ was higher than the $CWPE_{im}$—in terms of both energy and protein production between 1961 and 2017 (Fig 2.6a,b). As a result, trade has led to a potential saving of 170 or 80 million livestock standard units (LSUs) during 1961–2017 when productivity was expressed in terms of energy or protein, respectively (Fig 2.4). Again, this potential saving implicitly assumes that there are no biophysical or policy limitations hindering importing countries from producing enough livestock products for domestic consumption. The potential saving of the total number of livestock units in 57 yr, through trade of livestock products, was equivalent to 20–50% of the average total number of livestock units in the world in a year (FAOSTAT, 2020; Liu et al., 2017). The leading high-efficiency livestock-exporting countries (responsible for around 80% of total livestock protein export) were the Netherlands, New Zealand and Germany. These countries had an average annual livestock productivity of >40 kg protein per LSU, and contributed most to the potential saving of livestock units in the past 57 yr (Fig 2.6b). The potential saving has increased in the 2010s to around 17 million LSUs (Fig 2.5j,l), which was equivalent to 4.2% of global LSUs in the 2010s (Liu et al., 2017).

2.3.4 Partial fertilizer nitrogen and feed nitrogen productivities

Crop products were sourced from countries with high partial fertilizer nitrogen productivity, and were imported by countries with relatively low partial fertilizer nitrogen productivity, because $CWPE_{ex}$ was 180–250% larger than $CWPE_{im}$ between 1961 and 2017 for partial fertilizer nitrogen productivity expressed in terms of calorie or protein production (Fig 2.3c,d). As a result, this trade has led to a cumulative potential saving of 360 Tg synthetic fertilizer nitrogen when expressed in terms of crop calorie production, and of 480 Tg synthetic fertilizer nitrogen when expressed in terms of crop protein production (Fig 2.4). Global synthetic fertilizer nitrogen consumption has rapidly increased during this period, from 11 Tg in 1961 to 109 Tg nitrogen in 2017 (FAOSTAT, 2020); international trade potentially saved 5.8–7.7% of the annual global synthetic fertilizer nitrogen consumption between 1961 and 2017. Around half of the potential saving of synthetic fertilizer nitrogen occurred in the last two decades (Fig 2.5f,h), although the difference between $CWPE_{ex}$ and $CWPE_{im}$ decreased between the 1960s and the 2010s (Fig 2.5e,g). The potential global synthetic fertilizer nitrogen saving ranged from 12 to 18 Tg yr^{-1} between 2011 and 2017, for calorie- or protein-based estimates, respectively, which was 11–16% of the global annual consumption in 2017 (FAOSTAT, 2020). However, our partial fertilizer nitrogen productivity indicator did not account for nitrogen inputs via manure or for biological nitrogen fixation, which have increased during the last few decades (Lassaletta et al., 2014a). Hence, the impact of trade on global nitrogen use efficiency is likely to have been overestimated in this study.

Trade has had contradictory impacts on global partial feed nitrogen productivity in livestock production (Fig 2.6c,d). A negative impact of trade on protein-based partial feed nitrogen productivity was noted, which was related in part to the finding that some large importing countries were efficient in converting feed nitrogen into animal protein. For example, leading importing countries, such as Japan, South Korea and Israel, had a relatively high partial feed nitrogen productivity of 1.0–2.0 kg protein (kg feed N)$^{-1}$ (Fig 2.6c,d), and these countries contributed as much as 70% to the total imports. The higher partial feed nitrogen productivity in Japan, South Korea and Israel may partly be due to a higher proportion of poultry animals to total livestock production, and to higher livestock productivity and better management (FAOSTAT, 2020; Uwizeye et al., 2020). Exporting countries with relatively low partial feed nitrogen productivity of >0.5 kg protein (kg feed N)$^{-1}$ were responsible for as much 50% of the total exports during the past 57 yr (Fig 2.6d). The negative gap between exporting and importing countries in livestock partial feed nitrogen productivity has decreased in recent decades both in terms of livestock calories and protein production (Fig 2.5n,p).

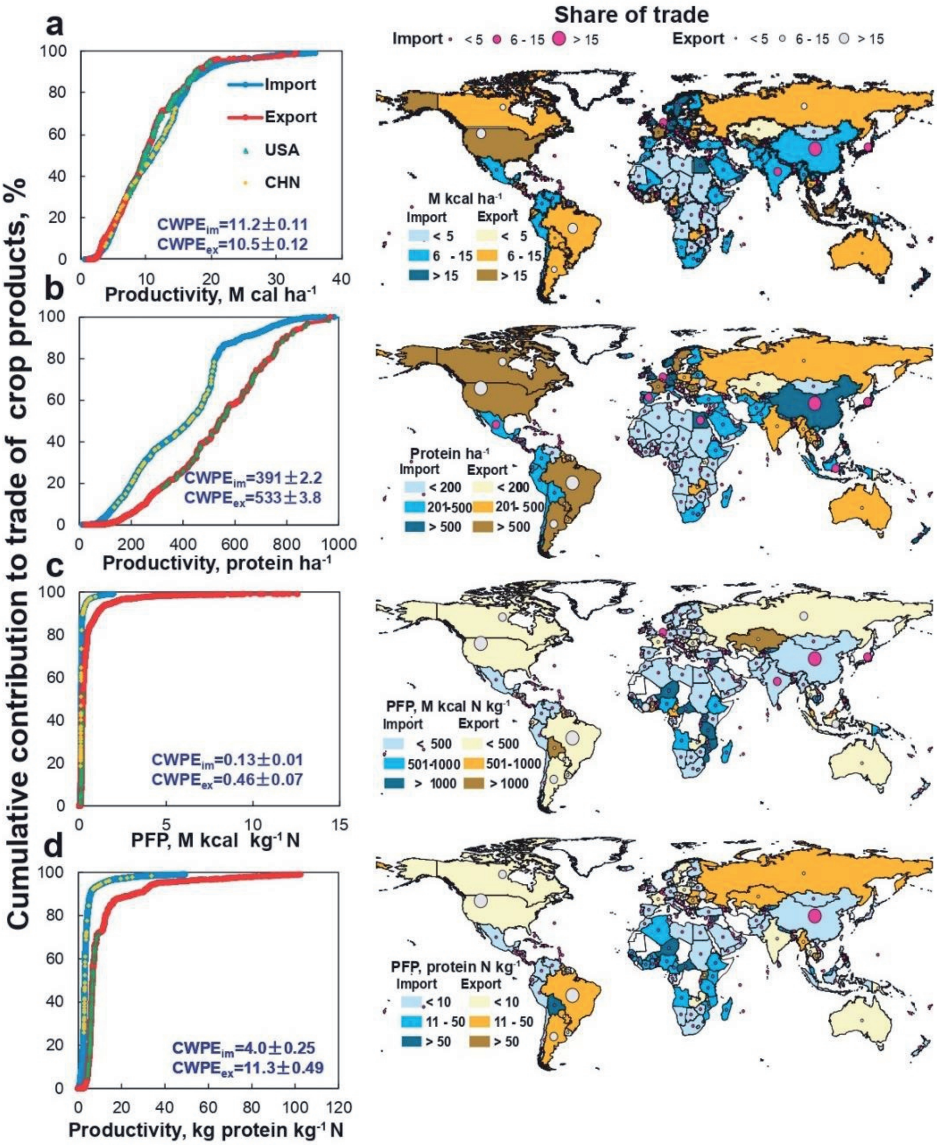

Figure 2.3. Cumulative productivity-trade distribution curves. a,b, Crop productivity of exporting and importing countries in terms of calorie (a) and protein (b) production. c,d, Partial fertilizer productivity (PFP) of nitrogen in terms of calorie (c) and protein (d) production. *Productivities from 1961 to 2017 are shown in the left-hand panels, and the contributions of each country to total trade in 2017 are shown in the right-hand panels. Colors in the maps represent the level of productivity of exporting and importing countries; the size of*

the circle of each country represents its contribution to total exports or imports. The error bars relate to the selection of the maximum productivity at 98.5%, 99.0% and 99.5% contributions to the total traded products.

2.3.5 Ultimate fate of traded nitrogen in importing countries

There is little information available about the ultimate fate of nitrogen embedded in traded crop and livestock products. Here we separate traded agricultural products into those used for human food and animal feed to estimate the distribution of traded nitrogen between utilization and losses to the environment (Fig S2). Our results indicate that much of the traded nitrogen ended up in the environment, and little was recycled in the crop production system. Globally, around 3.7 Tg nitrogen was embedded in the trade of human food in 2017 and this 3.7 Tg nitrogen was probably also excreted by humans because retention in human bodies is negligibly small. We estimated that about 40% (1.4 Tg nitrogen) of human excreted nitrogen was converted into N_2, in part following sewage treatment (Galloway and Leach, 2016; Oita et al., 2016). The latter occurred mainly in economically developed regions, for example, Japan, South Korea, the United States and the European Union, due to environmental regulations related to sewage collection and treatment (Fig S3). We estimated that of all feed nitrogen traded (10 Tg) in 2017, a total of about 2.5 Tg nitrogen was retained in milk, meat and eggs, about 3.1 Tg nitrogen was recovered as manure used to fertilize cropland and the remaining 4.4 Tg nitrogen was lost to the environment. China was a main leakage point of globally traded feed nitrogen, due to its large soybean import and poor manure management (Bai et al., 2016; Sun et al., 2018). Overall, more than 40% of total traded food and feed nitrogen (14 Tg nitrogen) was not recycled and ended up in the environment (Fig S3). This lost nitrogen probably contributed 5–10% to the excess above the 'safe operating space' for biogeochemical nitrogen flows (about 60 Tg nitrogen) (Steffen et al., 2015). These estimates indirectly indicate that trade of animal products rather than feed may improve the global nitrogen use efficiency at the food system level because some of the leading feed-importing countries currently have lower livestock nitrogen use efficiencies and manure recycling rates than the leading livestock-exporting countries (Bai et al., 2020).

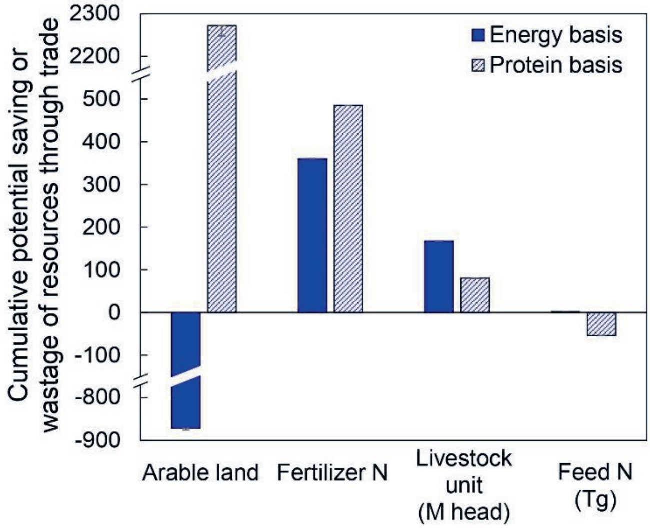

Figure 2.4. Cumulative potential saving. Positive values correspond to savings and negative values correspond to wastage of arable land (Mha), synthetic fertilizer nitrogen (Tg), livestock units (millions of head) and feed nitrogen (Tg), as a result of trade of cropland livestock products between exporting and importing countries with productivity differences during the period 1961–2017.

2.3.6 Optimality and functionality of traded products

We evaluated the international trade of crop products as being non-optimal and low-functional (level VI) in terms of cropland calorie productivity during the period 1961–2017 because the ratio of $CWPE_{ex}/CWPE_{im}$ was<1.0, the $CPHE_{ex}$ was <0.50 and the $CPHE_{im}$ was <0.50(Fig 2.7a) When expressed in terms of protein productivity, trade of crop products was evaluated as near-optimal (level I) (Fig 2.7b). Trade optimality was relatively high but trade functionality was relatively low from the point of view of partial fertilizer nitrogen productivity (Fig 2.7a,b). Trade of livestock products was evaluated as optimal and functional (level II) in terms of calorie- and protein-based livestock productivity (Fig 2.7a,b).

Trade of livestock products was optimal and functional (level I) when expressed in terms of calorie-based partial feed nitrogen productivity, but it was non-optimal and low-functional (level VI) in terms of protein-based partial feed nitrogen productivity (Fig 2.7b).

2.3.7 Changes over time

The $CPHE_{ex}$ of cropland calorie productivity has decreased from 0.50 in the 1960s to 0.36 in the 2010s (Fig S4, upper panel). This is a result of decreasing contributions of high-efficiency exporting countries to the total export of crop calories. However, the negative effect of trading crop products on global cropland productivity has decreased over time due to the faster increase of productivity in the net exporting country group compared to the net importing countries (Fig S4, upper panel); the negative gap between $CWPE_{ex}$ and $CWPE_{im}$ diminished (Fig 2.5a). Hence, trade optimality improved slowly from level VII in the 1960s to level VI in the 2010s in terms of crop calorie productivity (Fig 2.7c).

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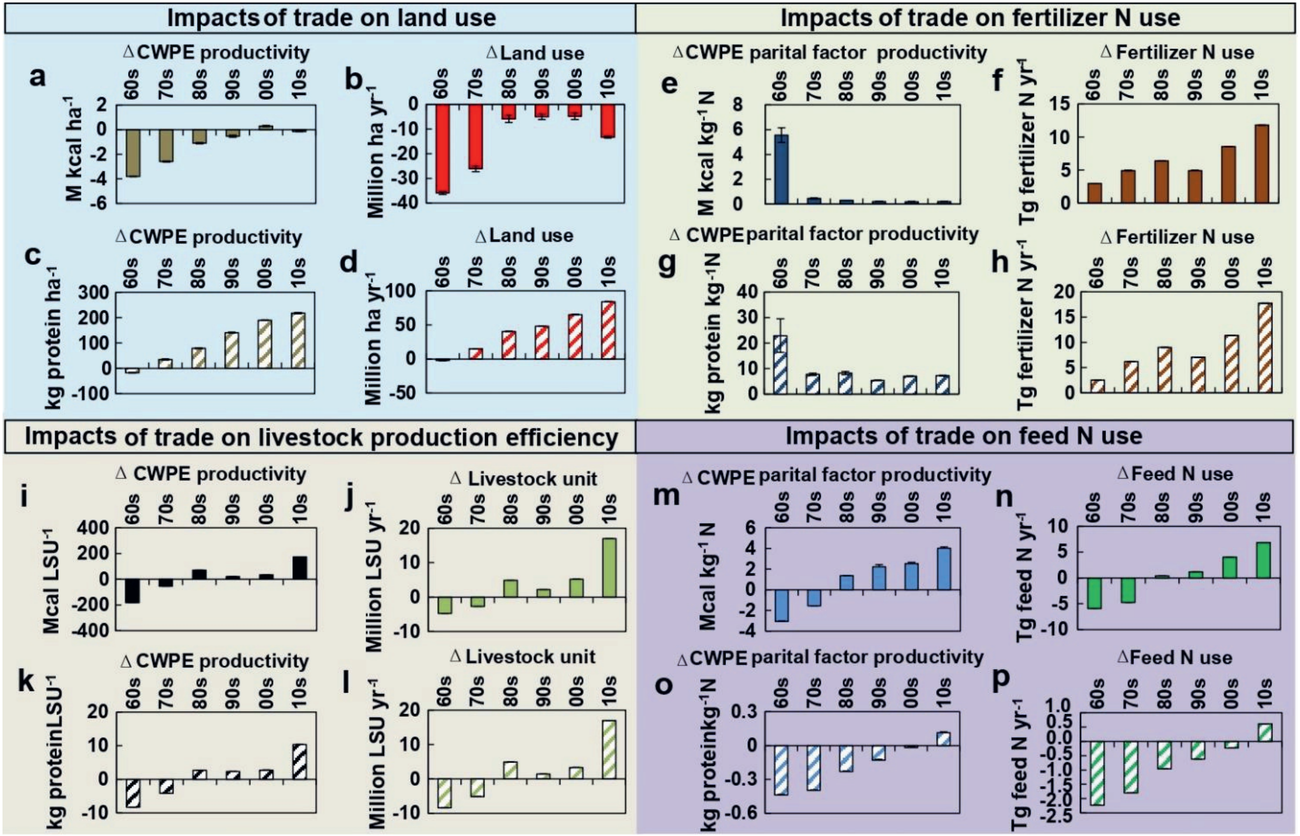

Figure 2.5. Changes per decade in the impacts of trade. a–p, Trade impacts on crop productivity (a,c) potential land saving (b,d), partial fertilizer nitrogen productivity of crop production (e,g), potential synthetic fertilizer nitrogen saving (f,h), livestock productivity (i,k), potential livestock unit saving (j,l), partial feed nitrogen productivity of livestock production (m,o) and potential feed nitrogen saving (n,p). *Δ represents the differences between exporting and importing countries. The 2010s include data for 2011–2017. The error bars relate to the selection of the maximum productivity at 98.5%, 99.0% and 99.5% of total traded products. Solid columns are energy-based results; hatched columns are protein-based results.*

The international trade in crop products has had a positive effect on global cropland productivity over the last six decades (except in the 1960s) when cropland productivity is expressed in terms of protein production per hectare (Fig 2.5c). There were also steady increases in trade functionality of crop products (Fig 2.7e), which is partly related to the massive expansion of soybean production in Brazil, the United States and Argentina for export to China and the European Union over the last two or three decades, but which was partially at the cost of precious tropical forests and related biodiversity (Barona et al., 2010; Bowman et al., 2012).

The mean CWPE values of exporting and importing countries for partial fertilizer nitrogen productivity have decreased over time (Fig 2.7d), which was related to the rapidly increasing use of synthetic nitrogen fertilizer in the past six decades, especially in emerging economies, such as China (Vitousek et al., 2009; FAOSTAT, 2020). Differences between $CWPE_{ex}$ and $CWPE_{im}$ for partial fertilizer nitrogen productivity were positive, and were relatively high in the 1960s but greatly decreased thereafter (Fig 2.5e, g). However, there were no changes in trade functionality level in terms of partial fertilizer nitrogen productivity; trade functionality was at the bottom-left of quadrant III, when expressed in terms of either calorie or protein production (Fig 2.7d,f).

International trade in livestock products has contributed to an increase in global livestock productivity, both in terms of livestock calorie and protein production, during the last four decades (from the 1980s to the 2010s) (Fig 2.5i,k). Some countries with high livestock productivity are main importers of crop products and main exporters of livestock products; these countries import calorie- and protein-rich feed to produce and export milk, meat and egg (for example, Denmark, Germany, the Netherlands and Spain). There were no large changes in trade functionality during the last four decades, both in terms of calorie- and protein-based livestock productivity (Fig S5). However, trade optimality and functionality varied in the past six decades, and the trend was different when the partial feed nitrogen productivity was expressed in terms of calorie and protein productivity (Fig 2.5m,o; Fig S5-6).

2.3.8 Trade optimality of different products

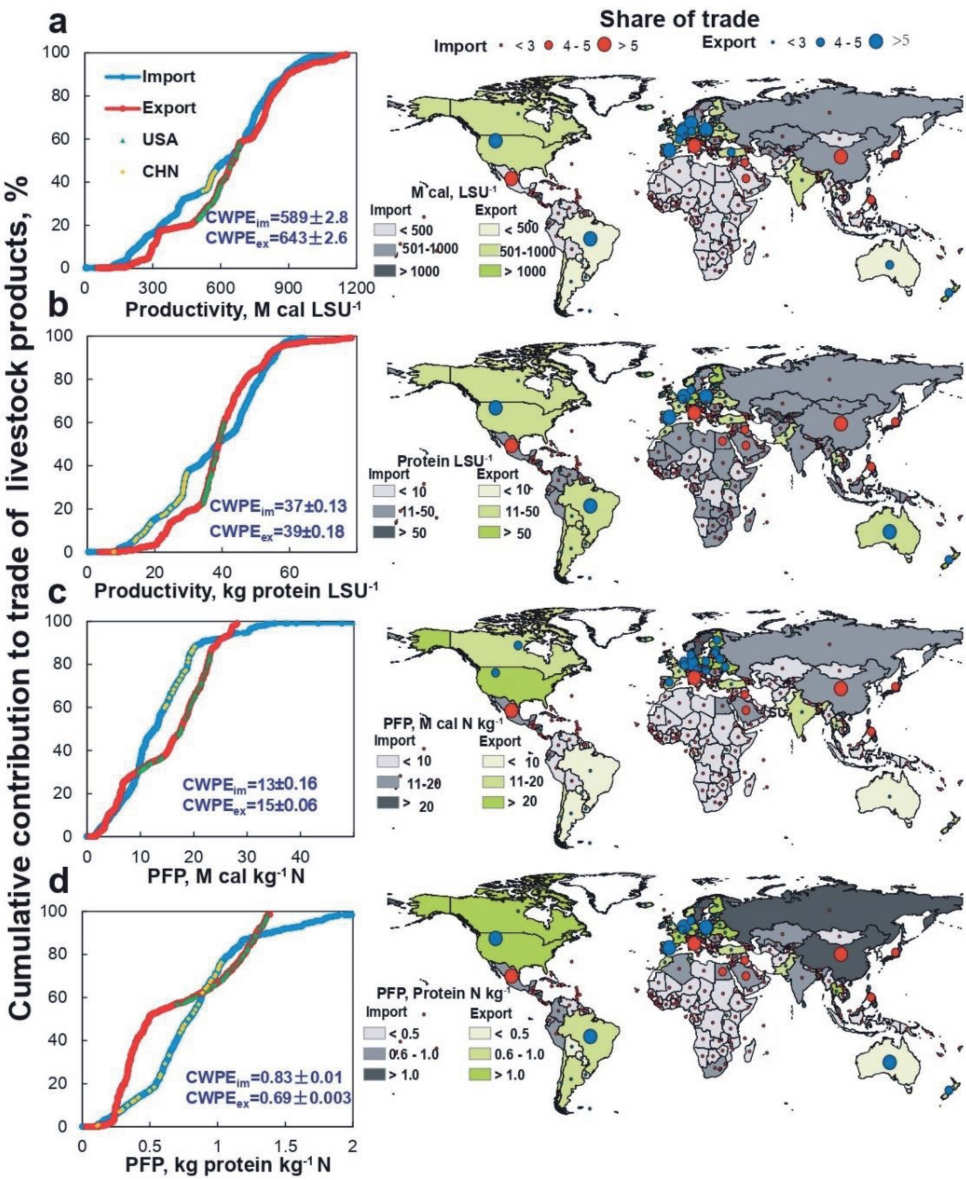

Figure 2.6. Cumulative productivity–trade distribution curves of exporting and importing countries. a,b, Livestock energy (a) and protein production (b) per LSU. c,d, Livestock energy (c) and protein production (d)

per feed nitrogen input. Productivities from 1961 to 2017 are shown in the left-hand panels and the contributions of each country to total trade in 2017 are shown in the right-hand panels. *Color in the maps represents the level of productivity or efficiency of exporting and importing countries; the size of the circle of each country represents the contribution to total export or import. The error bars relate to the selection of the maximum productivity at 98.5%, 99.0% and 99.5% contributions to the total traded products.*

The international trade of six selected main traded crop products (maize, wheat, rice, barley, soybean and potato) was optimal in terms of crop calorie and protein productivity between 1961 and 2017 (Fig S7). Trade of maize and soybean had a relatively high optimality level, which is reflected by the larger diameter of the red circles in Fig S7. However, trade functionality varied among these six crop products, with maize, soybean and barley in quadrant I (Fig S7). Additional information about different crop and livestock products can be found in Table S3.

2.4 Discussion

In the absence of trade restrictions or cultural barriers, trade allows an exchange of reciprocal productivity advantages between different regions, communities or cultures. Hence, food and feed trade was expected to contribute to improved global land and nitrogen use efficiencies. Our study identified contradictory results, however, when comparing cropland and livestock productivities and partial fertilizer nitrogen and feed nitrogen productivities on the basis of calorie versus protein production. This may indicate that protein productivity more strongly influences the establishment of trade flows than the calorie content of the products. This may require a rethinking of the main functions of agricultural trade, especially as the current UN Sustainable Development Goal on 'Zero Hunger' mainly addresses daily dietary energy supply (Sachs et al., 2021).

2.4.1 Implications of trade for cropland productivity

The estimated average annual potential saving of cropland through international trade of food and feed in the 2010s was comparable to estimates of previous studies when expressed in terms of crop protein production (Fader et al., 2013; Kastner et al., 2014). However, trade of crop products contributed to a potential wastage of global cropland when productivity was expressed in terms of calorie production (Fig 2.5a). This was related to the import of crop products by some leading high-efficiency importing countries, such as the Netherlands and Japan, with an average crop calorie productivity >16×10^6 kcal ha^{-1} (Fig 2.3a); it reflects a relative scarcity of cropland. The cropland area also declined in these countries because of competition from infrastructure and nature conservation (Fig S8a–c). Conversely, export-oriented production in Brazil, Malaysia and Indonesia was associated with cropland expansion and deforestation (Koh and Wilcove, 2007; Barona et al., 2010) (Fig S8d–f). Expanding high-efficiency cropland at the expense of natural land in some areas may contribute to saving cropland at the global level when a large expansion of low-productive cropland in other areas can be minimized. However, this may conflict with the concept of land sharing to protect biodiversity and reduce greenhouse gas (GHG) emissions, that is, expanding soybean production in Brazil may increase global protein productivity, but at the cost of biodiversity losses (Kremen, 2015).

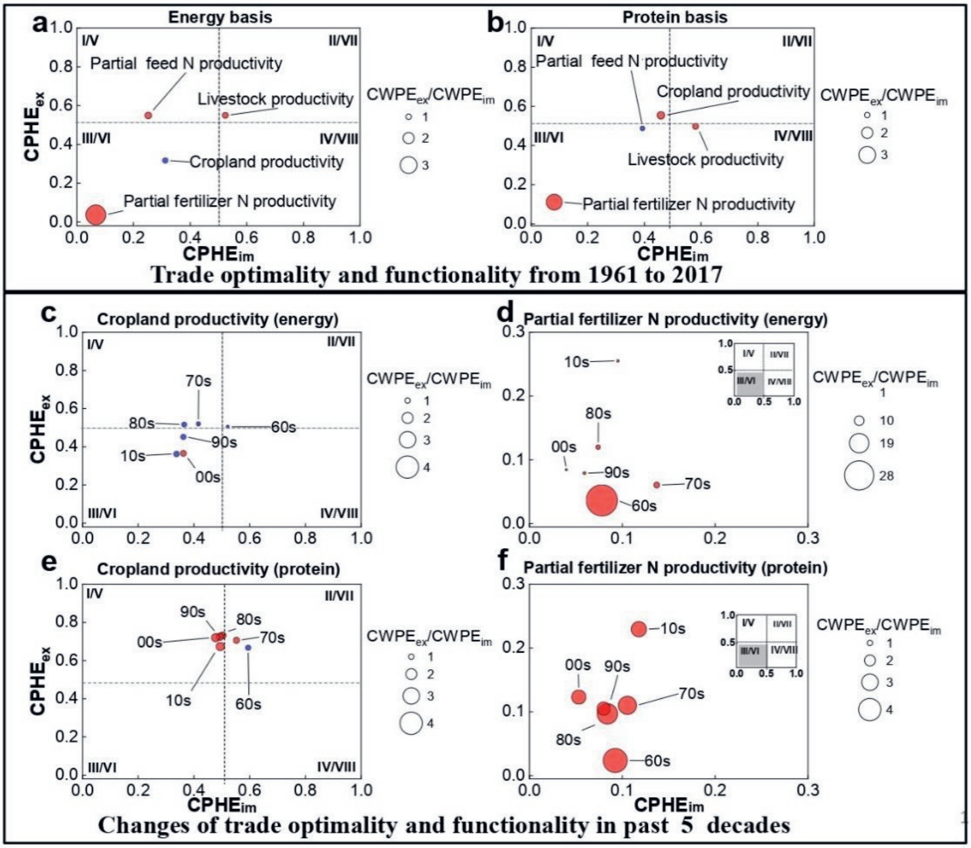

Figure 2.7. Trade optimality and functionality levels. a,b, Trade optimality and functionality of crop and livestock production from 1961 to 2017 shown in terms productivity on a calorie basis (a) and a protein basis (b). c–f, Changes in trade optimality and functionality of crop and livestock production over the past five decades on a calorie basis (c,d) and a protein basis (e,f). *The size of the circles represents* $CWPE_{ex} - CWPE_{im}$. *The red solid dots represent positive trade optimality (levels I–IV; that is,* $CWPE_{ex}/CWPE_{im} \geq 1.0$*); the blue solid dots represent negative trade optimality (levels V–VIII; that is,* $CWPE_{ex}/CWPE_{im} < 1.0$*).*

The idea of trade optimality is that production occurs in areas with the best possible output–resource input ratio, and that products are transferred (traded) from these high-efficiency areas to areas with lower output–resource input ratios. High-productivity importing countries with little land could expand their domestic crop production in high-tech and high-productivity greenhouses (CBS, 2020), which would decrease $CWPE_{im}$ and hence increase the $CWPE_{ex}$/ $CWPE_{im}$ ratio. An increase in the trade optimality level could also be achieved by increasing $CWPE_{ex}$ via transfer of knowledge and technology. This is important for exporting countries with low productivity, such as Kazakhstan, Russia, Zambia and Uruguay (Fig 2.3), because it may increase crop calorie productivity and subsequent export without expanding cropland (Sanchez, 2015). Increasing productivity in high-productivity countries faces the challenge of reaching potential yield limits; for example, wheat yields in some European countries have reached biophysical limits (Muller et al., 2012).

The potential saving of livestock units as a result of international trade of livestock products will probably have contributed to a reduction of several million tonnes of nitrogen losses and GHG emissions into the atmosphere, as the livestock sector has probably contributed to the emission of 7.1 billion t CO_{2eq} and 119 Mt of ammonia annually during the last decade (Oita et al., 2016; Sutton et al., 2013). The subsequent effects of trade on the potential saving of livestock units in terms of potential saving of feed use and cropland area have not been assessed in this study, but may be large (Mottet et al., 2017). However, these effects are difficult to quantify because part of the feed consumed in a country may have been imported from other countries, and there are large differences in feed composition and feed conversion ratio between animal categories and between countries (Bai et al., 2018; Eshel et al., 2014; Gerber et a., 2013).

2.4.2 Implications of trade on partial fertilizer nitrogen productivity

The positive impact of the trade of food and feed on partial fertilizer nitrogen productivity at the global level through time is in part related to the inefficient fertilizer use at the beginning of the study period for some of the world's major crop exporters. It is also related to the increasing proportion of export coming from countries with high partial fertilizer nitrogen productivity (for example, Africa and South America) (Zhang et al., 2015). The high partial fertilizer nitrogen productivity in African countries results from soil nitrogen mining, which is not sustainable for any country in the longer term (Zhang et al., 2015; Lassaletta et al., 2014a). The high partial fertilizer nitrogen productivity in South American countries was probably related to the relatively large nitrogen input via biological nitrogen fixation in soybean production, which we did not account for.

Partial fertilizer nitrogen productivity may also increase through better utilization of nitrogen from animal manure and household wastes, and an equivalent decrease in synthetic fertilizer use (Lassaletta et al., 2014a). We estimated that 1.4 Tg nitrogen contained in traded food was converted into N_2 following treatment in sewage treatment plants, the residue of which can potentially be recycled into agricultural production systems. Around 4.4 Tg nitrogen in traded animal feed nitrogen was lost from animal houses and manure storage facilities. For example, only around one-third of China's livestock manure nitrogen was effectively applied to cropland; the remainder was either emitted to the air or discharged to watercourses and landfills (Bai et al., 2014). Technological development and investments in low-emission animal housing and manure storage facilities, and in low-emission manure transport and application facilities, would help to reuse a greater proportion of the nitrogen embedded in traded feed products (Lassaletta et al., 2014b). Total synthetic fertilizer nitrogen use in China could be reduced from around 30 Tg in 2012 to 5 Tg were these technologies and advances in crop and livestock production fully implemented. This would contribute greatly to the global attempt to keep nitrogen use within the planetary boundaries (Jin et al., 2020).

2.4.3 Trade optimality level and implications

Trade optimality and functionality as defined in this study do not consider wider ecosystem impacts. However, it is well known that some leading exporting countries have increased the export of crop and livestock products in part through land expansion and deforestation (Barona et al., 2010; Bowman et al., 2012). For example, soybean export from Latin America is associated with deforestation and biodiversity loss (Bowman et al., 2012; Soterroni et al., 2019). Palm oil export from some Southeast Asian countries is associated with deforestation, peatland degradation and biodiversity loss (Koh and Wilcove, 2007). Similarly, some leading livestock exporting countries, such as the Netherlands, Denmark and Germany, suffer from nitrogen pollution and biodiversity loss caused by NH_3 emissions from livestock production, especially in livestock-dense regions (Jongbloed et al., 1999). Hence, although trade of crop and livestock products may be evaluated as optimal and functional in terms of land and nitrogen use efficiencies, it may be non-optimal and low-functional when evaluated in terms of GHG emissions, biodiversity conservation and environmental pollutions. The new analytical framework with the cumulative productivity distribution curve developed in this study allows such indirect impacts to be included, but it will require additional indicators for quantitative assessments, such as land use change, GHG emissions, nitrogen losses and biodiversity losses. Further, the trade of crop products (for example, sugar cane, corn, soybean) used for biofuel, and products used for pharmaceuticals and industry may also be evaluated using this framework.

Overall, our framework allows uniform assessments for importing and exporting countries to be made, using multiple indicators, and may help to set priorities for specific countries and specific products. In addition, the framework developed here is simple, transparent and easily extendable. It provides a functional tool and various useful indicators for researchers and policymakers. More applications of the cumulative production curve approach can be envisaged, including in industry and ecology.

2.5 Acknowledgements

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CHAPTER 3

China's livestock transition: Driving forces, impacts, and consequences

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Abstract

China's livestock industry has experienced a vast transition during the last three decades, with profound effects on domestic and global food provision, resource use, nitrogen and phosphorus losses, and greenhouse gas (GHG) emissions. We provide a comprehensive analysis of the driving forces around this transition and its national and global consequences. The number of livestock standard units (LSUs) tripled in China in less than 30 years, mainly through the growth of landless industrial livestock production systems and the increase in monogastric livestock (from 62 to 74% of total LSUs). Changes were fueled through increases in demand as well as, supply of new breeds, new technology, and government support. Production of animal source protein increased 4.9 times, nitrogen use efficiency at herd level tripled, and average feed use and GHG emissions per gram protein produced decreased by a factor of 2 between 1980 and 2010. In the same period, animal feed imports have increased 49 times, total ammonia and GHG emissions to the atmosphere doubled, and nitrogen losses to watercourses tripled. As a consequence, China's livestock transition has significant global impact. Forecasts for 2050, using the Shared Socio-economic Pathways scenarios, indicate major further changes in livestock production and impacts. On the basis of these possible trajectories, we suggest an alternative transition, which should be implemented by government, processing industries, consumers, and retailers. This new transition is targeted to increase production efficiency and environmental performance at system level, with coupling of crop-livestock production, whole chain manure management, and spatial planning as major components.

3.1 Introduction

Global food security and the sustainability of food production and consumption greatly depend on how to manage livestock production and animal source food consumption (Godfray et al., 2010; Herrero and Thornton, 2013). Livestock production systems use a great proportion of the world's crucial resources, such as land and water (Steinfeld et al., 2006; Deutsch, 2010; Steinfeld et al., 2013), and are a main source of non-CO_2 greenhouse gas (GHG) emissions and ammonia (NH_3) in air, as well as of nitrogen (N) and phosphorus in surface waters (Deutsch, 2010； Bouwmanet al., 2013). Recent global assessments have provided a systematic quantification of the biomass use, feed conversion ratio, and productivity of different livestock production systems (Herrero et al., 2013). The changes in livestock production, to more efficient monogastric animals and landless production systems (where animals are housed, feed is imported from other farms and countries, and manure is only partly returned to crop land), have contributed to significant resources and emission savings at the animal level. However, the overall impact of livestock production has greatly increased, through increased production level (Garnett et al., 2013; Herrero and Thornton, 2013; Havlík et al., 2014) and decoupling of feed and animal production on farm, with a greater reliance on purchased cereal and pulse-based feeds, the human edible feeds (Wilkinson and Lee, 2017). Other studies have emphasized the large differences between farms in production efficiency and environmental performance and the scope for improvement (Eshel et al., 2014).

The term "livestock revolution" was first coined by Delgado et al (1999) to describe the rapid changes in production structure and efficiency and to advise governments and industries to prepare for this continuing revolution. The market value of the global increases in meat and milk consumption between 1970 and 1990 was two times higher than the market value of the increase in cereal consumption through the better known "Green Revolution," more specifically wheat, rice, and corn. The livestock

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revolution or transition has been characterized as “demand driven” and the green revolution as supply- or technology-driven, but there are regional differences (Steinfeld et al., 2013), especially for countries with strong central governments.

China is an interesting case of the livestock revolution or transition. The average meat, milk, and egg consumption per capita increased by 3.9, 10, and 6.9 times, respectively, between 1980 and 2010, which was by far the largest increase during this period in the world (FAO, 2017). In the early nineties, China exceeded the United States and Europe as the world’s biggest livestock producer (FAO, 2017). Mean livestock productivity was low, and nutrient losses and GHG emissions per unit of animal protein produced were relatively high compared with those of the United States and European Union (EU) (Bai et al., 2013; Eshel et al., 2014; Guo et al., 2017). Moreover, the demand for animal products is projected to increase further in China (Alexandratos and Bruinsma, 2012). As a result, livestock production will nearly double during the next few decades, which may have huge environmental and socioeconomic impacts, as recently discussed for China’s increasing demand for milk on the global dairy sector (Bai et al., 2018) and likewise on the pig production sector (Bai et al., 2014). These changes question the sustainability of future global livestock production.

The causes and effects of past changes in livestock production for different livestock categories and the perspectives of future livestock production in China are not clear. Livestock production changes are in response to phase shift changes in food demand, but, in addition, there are profound changes in livestock functions and categories, systems, actors, and impacts, that are poorly understood. Moreover, there is a need for insight into more sustainable livestock production pathways.

The overall aims of this study are to increase the understanding of the drivers and impacts of the livestock transition in China and to explore possible solutions to achieve sustainable livestock production in the near future. We selected a range of indicators, following the Driver-Pressure-State-Impact-Response framework (Kristensen et al., 2004), and used these indicators to comprehensively analyze changes and impacts in the three main livestock production systems: mixed, grazing, and landless/industrial systems. We focused on the period 1980–2010 because the most rapid changes took place in this period and because of data availability. Scenarios for 2050 explore the effects of a range of development pathways.

3.2 Materials and Methods

This materials and methods section consist of a description of the research system and boundary, a definition of livestock production systems, an introduction to the NUFER animal model, and a description of scenarios for 2050, which explores a more sustainable crop-livestock production future.

3.2.1 Research boundary

Fig S7 shows the system concept of this study; livestock production and feed production in China. Inputs (synthetic fertilizers, biological N fixation (BNF), atmospheric deposition, and feed import) are listed on the left-hand side. These inputs are considered "new" inputs. Outputs (livestock products, manure export to other systems, nutrient losses) are listed on the right-hand of the figure. Internal flows between the feed production compartment and the livestock production compartment are shown by dotted arrows. Nutrient accumulated or depletion may occur in the crop or grassland systems.

3.2.2 Definitions of livestock production systems

In this study three main livestock production systems were distinguished, according to the feed regime and manure recycling, and based on statistical data: (i) so called mixed crop-livestock production systems, (ii) grazing production systems and (iii) landless industrial production systems. Additionally, 20 sub-systems were distinguished, related to the main 6 livestock categories. The definitions of different livestock production systems are briefly described in Bai et al (2016). Traditional and backyard systems are classified as mixed production systems, medium and large size industrial production systems are classified as landless production systems.

The livestock production structure for the years 2000 and 2010 was derived from MOA and FAO databases (FAO, 2017; NBSC, 2017), and for the years 1980 and 1990 from expert consultations.

3.2.3 Overview of the calculation method and NUFER-animal model

3.2.3.1 Total feed DM intake

The method used in this study allows the partitioning of the feed consumption, land use, greenhouse gas (GHG) emissions, nutrient use and losses to the 6 main livestock categories (pigs, layers, broilers, dairy, beef and draught cattle, and sheep and goat), and to the 3 main livestock production systems. We began with the estimation of total biomass required by livestock production. The feed intake prediction module was based on the energy requirements for maintenance, body weight gain, lactation and reproduction. The calculation method has been described in detail by Bai et al (2013, 2014, 2016) for the year 2010. We assumed that there were no big changes in coefficients between 1980 and 2010. Yet, the mean body weight, live weight gain, milk yield and egg yield changed over time, and these changes were taken into account; for pig production see Bai et al (2014) for dairy production see Zhang et al (2017), for other animal categories see Bai et al (2016).

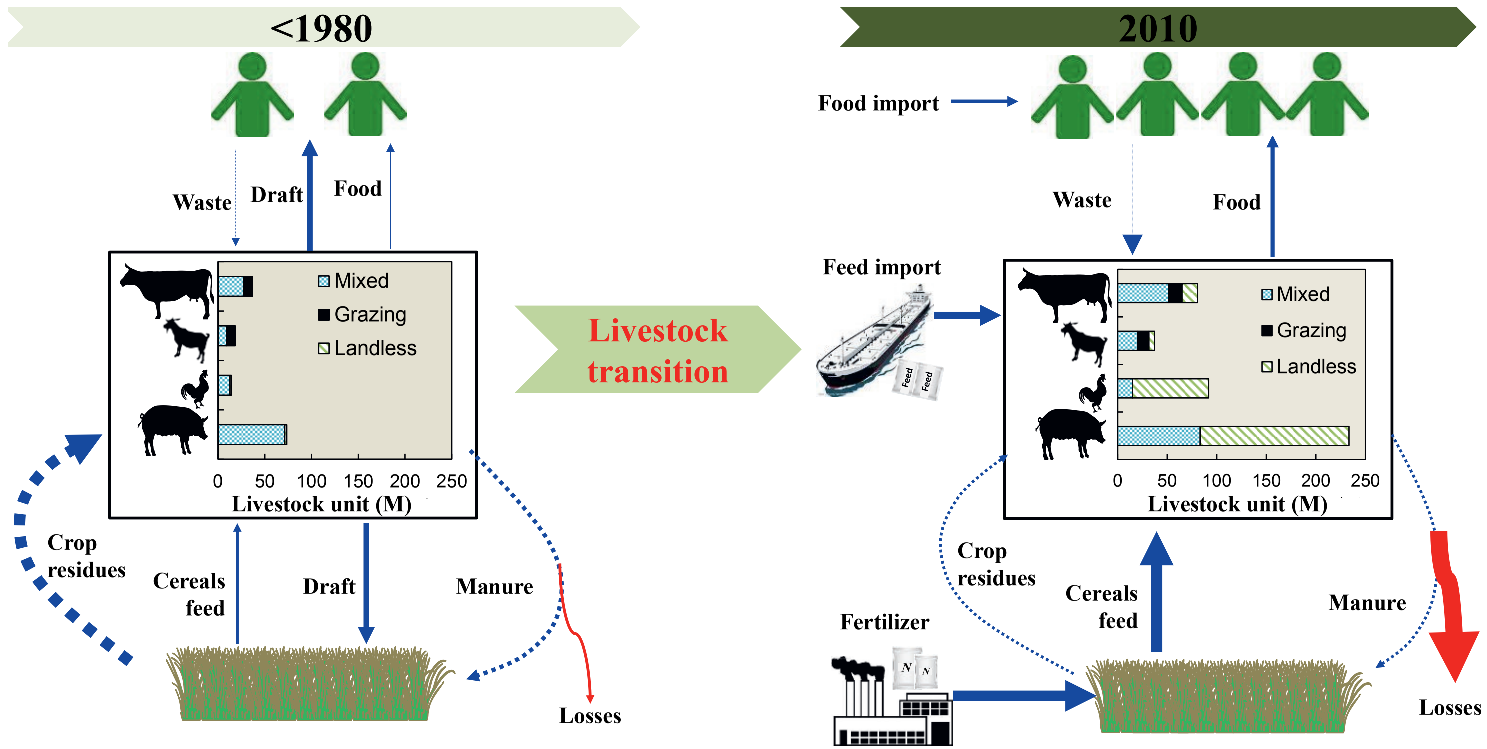

Figure 3.1. Concept of the livestock transition in China between 1980 and 2010. The left and right-hand graphs show the crop production (bottom part), livestock production (middle part) and the consumption of food (top part).

Note: Solid arrows represent nutrient inputs and outputs; the dotted arrows represent nutrient recycling flows. The thickness of the arrows reflects the size of the flows. Draft is the draft power provided by the draft animals.

3.2.3.2 Partitioning of the feed ingredients to different livestock categories

The feed ingredients consumed by each livestock category were calculated by feed compositions and the total feed intake per round.

$$Ia_{\text{feed consumption}} = Ia_{\text{DM intake}} \times Ia_{\text{feed compositions}} \qquad (1)$$

$Ia_{\text{feed consumption}}$ (kg) is the consumption of a specific feed ingredient (DM) by a specific livestock category in a specific production system per year. In total there were 15 feed ingredients, 6 livestock categories and 3 production systems. $Ia_{\text{DM intake}}$ is the total feed intake (sum of all 15 feed ingredients) consumed by a specific livestock category in a specific system; $Ia_{\text{feed composition}}$ is the percentage of the total feed intake of a specific feed ingredient, including corn, rice, wheat, soybean, rice, wheat, vegetables, by-products of food processing (chaff, bran and some main products), straw, grass, tubers, kitchen waste, leafs and green straws, animal by-products, residues of vegetables, and other unknown feed.

The feed compositions of landless production systems of each livestock category were collected from published literature. For traditional production systems, the feed compositions were based on literature, expert judgments, and model calculations. We applied the mass balance (total feed intake = total feed supply) at national level, to be able to check for inconsistencies in the feed balance. Available feed ingredients were allocated over livestock categories assuming that high-quality feeds (corn, soybean, rice and wheat) were used in the order (i) large-scale industrial systems > medium-scale industrial systems > traditional systems > backyard systems, and in the order (ii) poultry > pigs > dairy cattle > beef and draught cattle > sheep and goat. Further, landless systems used more concentrate feeds than traditional and grazing systems. Finally, we assumed that the feed composition per system did not change much over time. Detailed information about feed compositions for pigs are presented by Bai et al (2014) for dairy cattle by Zhang et al (2017), and for the other animal categories are presented in the SM.

3.2.3.3 Land requirement for feed production

Land requirement for feed production was calculated by the required amount of feed and the mean crop yield of the various feed ingredients. Imported feed and by-products were assumed to require no domestic land resources. The import of corn, soybean, rice and wheat were derived from FAO database. Grains from corn, wheat and rice, soybean and cakes, and grass (forage, including alfalfa) were seen as main products, which demanded land and new nutrients. The remainder of the feed ingredients were seen as by-products feed.

$$Oc_{land} = Ia_{DM\ intake} * (100\% - \text{Feed import}) / CY \quad (2)$$

Oc_{land} is the area of land used for feed production, based on land areas of corn, wheat, rice, soybean and cakes, and forage used for feed; Feed import (in % of the total feed use of corn, wheat, rice, soybean and cakes, and forage) was derived from FAO database; CY is the crop (forage) yield (kg ha^{-1}) for the various crops, derived from FAO database.

The virtual land requirement was derived from the feed import (corn, soybean, rice, wheat and forage) and the global average productivity of these feeds.

3.2.3.4 BNF and atmospheric N deposition

The inputs of N via BNF and atmospheric deposition to domestic feed production were calculated as follows:

$$Ic_{BNF} = Oc_{land} * BNF \quad (3)$$

Ic_{BNF} is the N input through biological N_2 fixation, in kg, Oc_{land} is the area of land with N_2 fixing crops, in ha, and BNF is the mean biological N_2 fixation of specific crops, in kg N per ha (Bai et al., 2013).

$$Ic_{deposition} = Oc\ land * Deposition \quad (4)$$

$Ic_{deposition}$ is the wet N deposition, in kg; Deposition is the atmospheric N deposition, in kg N per ha (Bai et al., 2013).

3.2.3.5 Fertilizer application

The fertilizer application was calculated as follows:

$$Ic_{\text{fertilizer}} = Oc\ feed / New\ \text{“N/P”}\ efficiency \qquad (5)$$

$Ic_{\text{fertilizer}}$ is the amount of fertilizer applied to certain crops, such as corn, soybean, wheat and rice, in kg; Oc feed is the nutrient content of the main feed species, in kg. New “N/P” efficiency is the use efficiency of the applied fertilizers, in kg N uptake per kg fertilizer N applied and in kg P uptake per kg fertilizer P applied. The use efficiencies are presented in the SM.

3.2.3.6 Nutrient intake by livestock

The total nutrient consumption by livestock was calculated from the nutrient content of the feed and the feed consumption (Bai et al., 2016).

$$Ia_{\text{nutrient intake}} = Ia_{\text{feed intake}} \times Feed_{\text{nutrient content}} / 1000 \qquad (6)$$

$Ia_{\text{nutrient intake}}$ (kg N or P) is the nutrient intake by different animal categories; $Ia_{\text{feed intake}}$ is the feed ingredient intake by an animal category, and $Feed_{\text{nutrient content}}$ (g kg^{-1}) is the nutrient content of specific feed ingredients (Bai et al., 2016).

3.2.3.7 Livestock products output

The main output of animal products was as meat, milk and egg. The mean carcass fraction (%) was used to convert live-weight to carcass weight. The average carcass fraction for pigs, chicken, beef cattle, and sheep and goat were set at 75%, 80%, 60% and 50%, respectively.

$$Oa_{\text{products}} = Animal_{\text{yield}} \times Animal_{\text{number}} \qquad (7)$$

Oa_{products} is the total amount of animal products, in kg; $Animal_{\text{yield}}$ is the yield of meat, milk or egg, in kg $head^{-1}$ yr^{-1}; $Animal_{\text{number}}$ is the number of animals per animal category. The protein output of animal products was calculated as follows:

$$Oa_{\text{protein of products}} = Oa_{\text{products}} * Protein_{\text{content}} / 1000 \qquad (8)$$

$Oa_{\text{protein of products}}$ is the total protein output of animal products, in kg protein; $Protein_{\text{content}}$ is the protein content of animal products, in g kg^{-1}. The protein content was derived from the N contents of different animal products (Bai et al., 2016).

3.2.3.8 Nutrient retention by livestock

The nutrient output via animal products were calculated from the animal products output and nutrient content of the animal products.

$$Oa_{\text{nutrient in products}} = Oa_{\text{products}} \times Products_{\text{nutrient content}} / 1000 \quad (9)$$

$Oa_{\text{nutrient in products}}$ is the amount of nutrient retained in the animal products, in kg; $Products_{\text{nutrient content}}$ is the nutrient content of animal products, in g kg^{-1} (Bai et al., 2016).

3.2.3.9 Manure nutrient production

The mass balance method was adapted to calculate the production of manure nutrients:

$$Oa_{\text{nutrient excretion}} = Ia_{\text{nutrient intake}} - Oa_{\text{nutrient in products}} \quad (10)$$

$Oa_{\text{nutrient excretion}}$ is the amount of nutrient excreted per animal category, in kg (Bai et al., 2016).

3.2.3.10 Nutrient losses from the manure management chain

The partitioning of manure nutrients and losses followed the description in Bai et al (2016). Reactive N losses include N losses via NH_3 and N_2O emissions to air, direct discharge of manure to the water bodies and/or landfill, and leaching, runoff and erosion of N from the manure management chain. The following four sources of GHG emissions were considered: (i) N fertilizer production, (ii) feed production, (iii) livestock production, and (iv) manure management (excluding the GHG emissions of imported feed). Both direct and indirect (from NH_3 emissions, nitrate leaching, and discharge of manure to water bodies) N_2O emissions were calculated. The non-CO_2 emission from the manufacture and use of chemical N fertilizers was set at 13.5 kg CO_2 eq per kg of N (Zhang et al., 2013). The indirect N_2O-N emissions from NH_3 volatilization and NO_3 leaching were set at 1% and 0.75%, respectively. Emissions of N_2O-N from discharged manure N were also set at 1% (Strokal and Kroeze, 2014). The average CH_4 emissions from enteric sources and manure management were derived from FAO (2017), which was based on the IPCC Tier 1 default factors. The global warming potential was 25 and 298 CO_2 eq for CH_4 and N_2O, respectively. Total GHG emissions were calculated as follows:

$$\text{GHG emissions} = \text{Fertilizer}_{\text{GHG emissions}} + \text{Feed production}_{\text{GHG emissions}} + \text{Livestock production}_{\text{GHG emissions}} + \text{Manure management}_{\text{GHG emissions}} \quad (11)$$

3.2.4 Scenarios for 2050

We designed and analyzed 6 scenarios to explore the possible impacts of livestock production in 2050, including a business as usual scenario (BAU).

3.2.4.1 Scenario business as usual (SSP2)

This scenario followed the SSP2 storyline, and the prediction of livestock food demand was based on Alexandratos and Bruinsma (2012), SSPs studies in China (Wang et al., 2017) and other information sources (Table S1). Further, we assumed that the increase in animal production between 2010 and 2050 will take place in landless systems (as was mainly the case between 2000 and 2010), and that the production in mixed crop-livestock systems (including traditional and backyard systems) and grazing systems will remain as in 2010. Also, we assume that the feeding practices and manure management remain as in 2010 (conservative estimate).

3.2.4.2 Scenario SSP1e

This scenario followed the SSP1 storyline, but the forecast of livestock food demand was as in SSP2. Four major technological changes were considered, in separate SSP1 variants, and a combined option was considered. SSP1a: improved feed and herd management, SSP1b: improved manure management and connected crop and livestock production, SSP1c: accelerated transformation of mixed systems to landless systems, SSP1d: structural adjustment; all additional required animal source food provided by monogastric animals, and SSP1e-combination of SSP1a and SSP1b. Detailed characteristics of these scenarios are shown in the Supplementary material.

3.3 Results

3.3.1 Characterization of the livestock transition

China's livestock population almost tripled between 1980 and 2010, from 142 to 441 million livestock standard units (LSUs). Functions of livestock also changed. Before the transition, livestock had multiple functions; it provided draft power, used household wastes, and provided manure to fertilize cropland, next to supplying animal protein. Between 1980 and 2010, supplying animal protein became much more important, facilitated by animal breeding programs and the increased availability of high-quality animal feed, increasingly through import (Fig 3.1). The increased availability of subsidized synthetic fertilizers made animal manure redundant for fertilizing cropland, while small machines replaced animal draft power.

Systems also changed. Traditional backyard and mixed crop-livestock systems were in part replaced by landless systems. In 1980, 2.5% of the total number of livestock (expressed in LSUs) was in landless systems (3.6 million LSUs in landless systems compared to the whole 142 million LSUs) and, in 2010, it was 56% (247 million LSUs in landless systems compared to the whole 441 million LSUs) (Fig 3.2A). At the same time, there was a shift from ruminant livestock (dairy cattle, other cattle, and sheep and goat) to monogastric livestock (pigs, layers, and broilers); the proportion of monogastric livestock to total LSUs increased from 62% (88 million LSUs in landless systems to the whole 142 million LSUs) in 1980 to 74% (325 million LSUs in landless systems compared to the whole 441 million LSUs) in 2010 (Fig 3.2B). Total animal protein production increased from 3.0 Tg in 1980 to 18 Tg in 2010. Landless systems produced 0.2 Tg of protein in 1980 and 12 Tg in 2010 (Fig 3.2C). Meanwhile, the annual gross economic value of livestock production increased from 35 to 2100 billion Yuan, a 58-fold increase (Fig 3.2E).

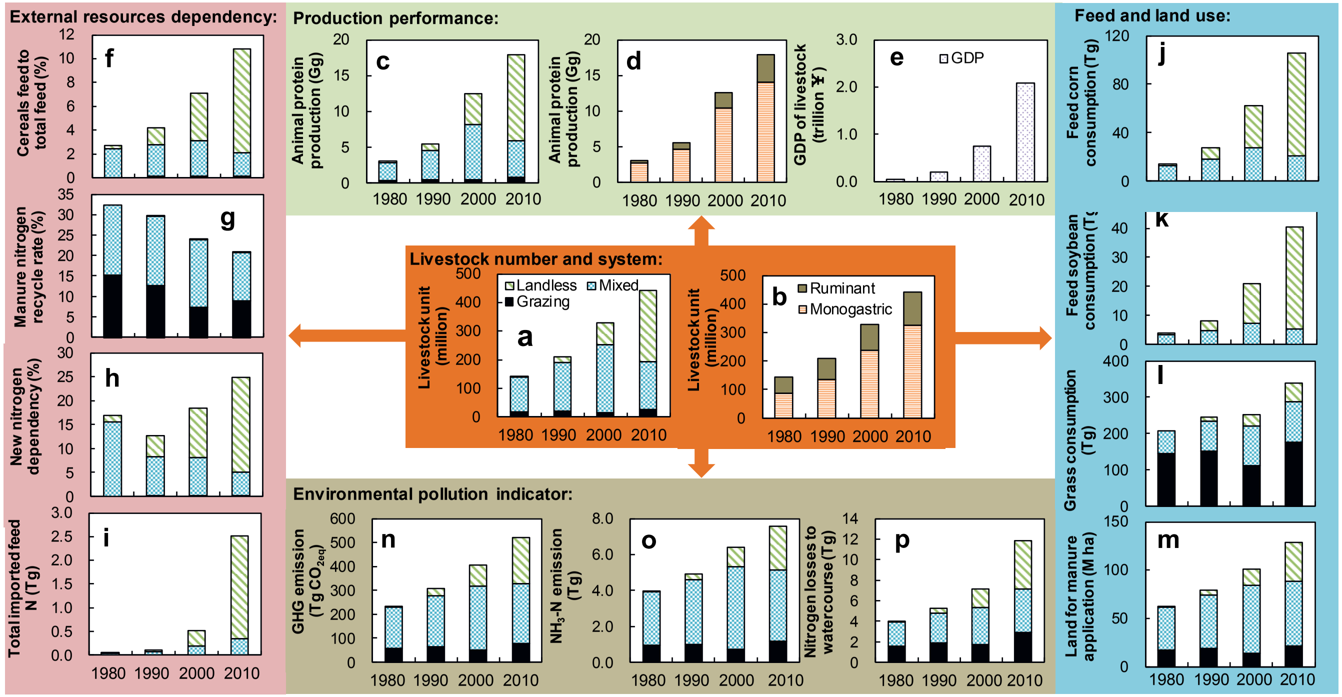

Figure 3.2. Changes in livestock production and performances between 1980 and 2010; livestock number and systems, in livestock units (central part a, b, respectively); Production performances: animal protein production and economic value (c, d, e); Nutrient use and recycling: external resources dependency indicator expressed in cereals feed dependency (f), manure nitrogen recycle rate (g), new nitrogen dependency (h) and imported feed nitrogen (i); Feed and land use: corn (j), soybean (k), grass (l) and land requirement for manure application (m); Environmental pollution: greenhouse gas (GHG) emissions (n), NH_3-N emissions (o) and N losses to watercourses (p). ***Note: soybean includes soybean and soybean cakes. Feed is expressed as dry matter. Land for manure application is defined as the area of land needed to apply the manure at an application rate of 170 kg N ha^{-1}yr^{-1}.***

3.3.2 Driving forces of the livestock transition

Increases in human population, economic growth [changes in gross domestic production (GDP)], and urbanization are commonly seen as main driving forces of the livestock transition, although research and technology development and food chain actors (suppliers, processing industry, and retail) also played an important role. Changes in livestock number were positively related with changes in human population and degree of urbanization (Fig 3.3). Livestock numbers were, however, not clearly related to GDP per capita; when the average GDP increased above 750 to 1500 US$ per capita, livestock number and percentage of monogastric livestock did not increase much further (Fig 3.3B). The ratio of monogastric to total livestock number will likely not increase, as recent findings indicate that the consumption of beef, mutton, and dairy products is increasing much faster than the consumption of pork and poultry meat (Table S1).

Governmental policies and subsidies also stimulated livestock production and transition, and three types of policies played a role (Fig 3.3 and Table S2). First is the liberation of markets and removal of barriers, such as the autonomy right to produce in 1980 and the free-market price policy in 1985 (Table S2). The supply of animal source food was controlled by the central government until the early 1990s. Meat coupons were essential for people to buy animal source food (Table S2). Most people could only eat meat during the Spring Festival, when there was additional meat supply. In the early 1990s, meat coupons were abolished, and farmers were allowed and encouraged to set up new livestock production farms. Between 1980 and 2010, the consumption of animal products per capita increased 2-fold for pork and 13-fold for beef products (Table S1). Second, economic incentives were provided for livestock producers, processing industry, and retail. The Vegetable Basket program of the government promoted industrial livestock farms around cities, and the Green Channeling policy-facilitated the processing and transport of animal products (Table S2). More than 10 billion Yuan of subsidies were directed annually to the livestock sector since 2007. Third, there were no or only loose environmental protection regulations

(Table S2). The few environmental regulations issued between 1980 and 2010 required only modest investments in manure storage, treatment, and application facilities. This lax environmental policy indirectly boosted large-scale and landless livestock production with poor manure management. These landless systems were economically much more profitable than the small traditional systems, which had relatively good manure management through a more coupled crop-livestock production system.

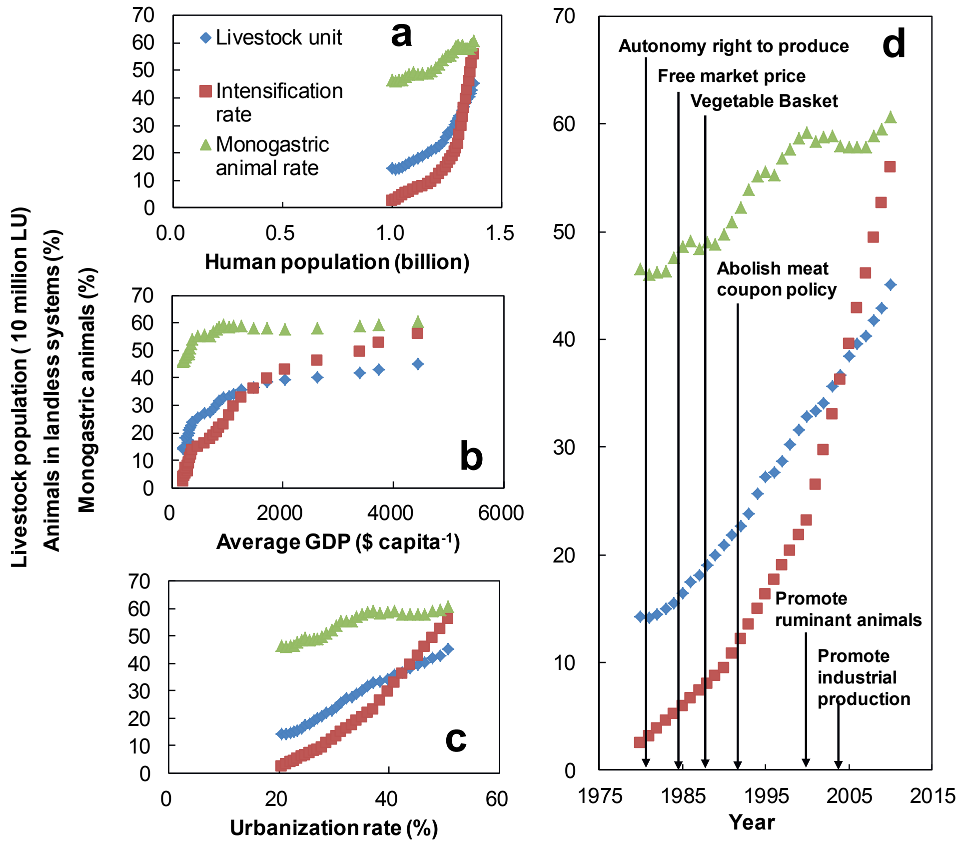

Figure 3.3. Relationships between livestock number (in LSU), the percentage of monogastric animals (in LSU) to total number of animals (in LSU), and the percentage of livestock in landless systems (% landless) versus human population (a), average gross domestic production value (GDP) per capita (b), urbanization (c), years of the introduction of governmental support policies (d).

Note: For details about the livestock production support polices see Table S2.

3.3.3 Pressures of livestock transition on resource use and the environment

The rapid increase in the number of livestock greatly enhanced the demand for feed and hence cropland. Consumption of corn (*Zea mays*) as livestock feed increased from 15 to 107 Tg dry matter (DM) between 1980 and 2010 (Fig 3.2J), and that of soybean (*Glycine max*) increased from 4.0 to 41 Tg (including soybean cakes) (Fig 3.2K). Most of these increases were driven by landless industrial systems; they were responsible for more than 80% of the consumption of feed corn and soybean in 2010, respectively (Fig 3.2J, K).

The increasing competition between plant source food production and livestock feed production affected cultivated areas and the prices of commodities. The demand for livestock feed increased beyond domestic feed production capacity, and increasing amounts of soybean, corn, and alfalfa (*Medicago sativa*; forage for cattle) had to be imported (Fig S2). Livestock production in China became dependent on feed import, while some traditional sources of animal feed (kitchen wastes, food losses, and crop residues) were left unused and have become a burden for the environment (see the changes in the thickness of the arrows between 1980 and 2010 in Fig 3.1). Consumption of forages (grass) increased from 209 to 341 Tg (Fig 3.2L). An increased land area was also needed for livestock manure disposal (Fig 3.2M).

Changes in livestock production also increased emissions of GHG and NH_3 to air and of N to water. Total GHG emissions from the livestock production chain increased from 233 to 520 Tg CO_{2e} between 1980 and 2010. The contribution of landless systems to total GHG emissions increased from 1.5% in 1980 to 37% in 2010 (Fig 3.2N). Total NH_3-N emissions increased from 3.9 Tg in 1980 to 7.6 Tg in 2010. Most of the NH_3 emission was from traditional, mixed production systems, but the contribution from landless systems increased rapidly (Fig 3.2O). Losses of N and phosphorus to surface water increased more than proportionally with livestock numbers, because of the decoupling of crop and livestock production. An increasing shortage of land near livestock farms led in turn to increasing difficulty in recycling manure back to cropland. As a result, a significant fraction of the manure was discharged to evaporation ponds and other water bodies, instead of being recycled to cropland/grassland, and substituting for synthetic fertilizer. Discharge of manure N to water courses increased from 4.0 to 12 Tg between 1980 and 2010 and posed an increasing threat to water pollution. Landless systems became the biggest contributor (Fig 3.2P). For the resources requirement and environmental performance of poultry, pig, cattle, sheep, and goat production, see Fig S3.

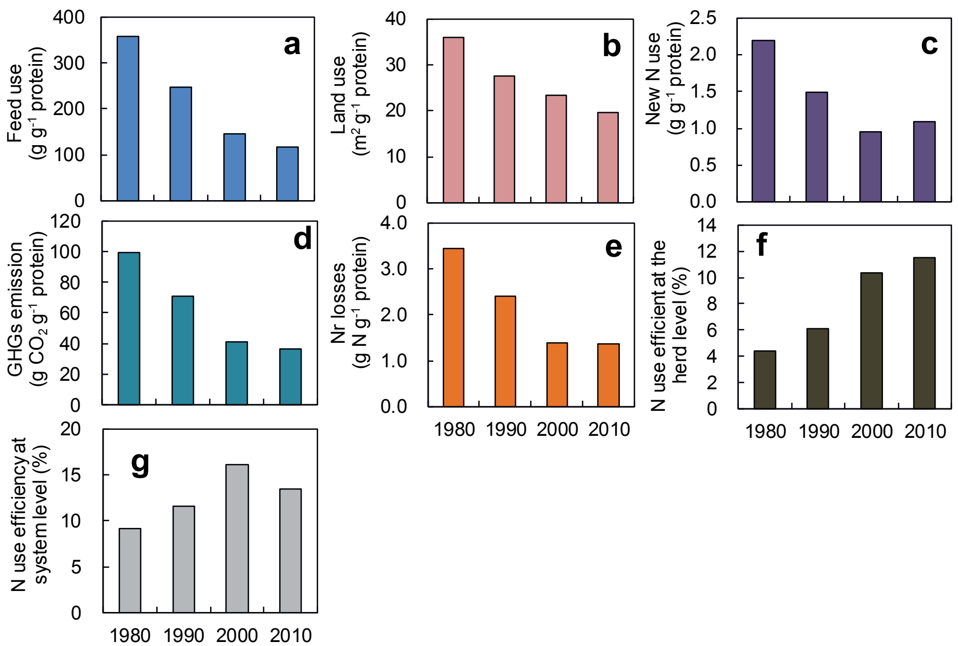

Figure 3.4. Changes in livestock production efficiency between 1980 and 2010: feed use per unit protein produced (a), land use per unit protein produced (b), new nitrogen (N) use per unit protein produced (c), greenhouse gas (GHG) emission per unit protein produced (d) reactive N (Nr) losses per unit protein produced (f), nitrogen use efficiency (NUE) at herd level (f) and NUE at the whole system level (g). *Note: NUE was calculated at herd level, including all main livestock categories, breeding animals*

and replacement animals (cattle, pig, poultry, sheep and goat), and at system level, including the whole soil-feed-livestock production chain.

3.3.4 Impacts of the livestock transition on productivity, efficiency, and manure recycling

The productivity and resource use efficiency of the livestock production sector greatly improved per LSU (Fig 3.4A) and per unit of animal protein produced. Animal protein production increased from 3.0 Tg in 1980 to 18 Tg in 2010, much faster than the increases in the number of livestock (Fig 3.2 A, B). The amount of feed needed to produce 1 g of edible animal protein decreased from 357 g in 1980 to 116 g in 2010. Similarly, the requirement for arable land decreased by about 46% per unit of edible animal protein produced (Fig 3.4A, B). Average nitrogen use efficiency (NUE) at herd level increased from 4.4% in 1980 to 11.5% in 2010. At the crop/feed-livestock system level, NUE increased from 9.1% in 1980 to 13.5% in 2010. A decrease of system-level NUE was observed between 2000 and 2010, likely because of the increased decoupling of crop and livestock production and the associated decrease in the utilization of manure nutrients (Fig 3.4F, G). Mean reactive nitrogen (Nr) losses from livestock production were decreased from 3.4 g in 1980 to 1.4 g (g animal protein^{-1}). Mean GHG emission decreased from 99 g CO_{2e} in 1980 to 36 g CO_{2e} (g animal protein^{-1}) in 2010 (Fig 3.4C, D).

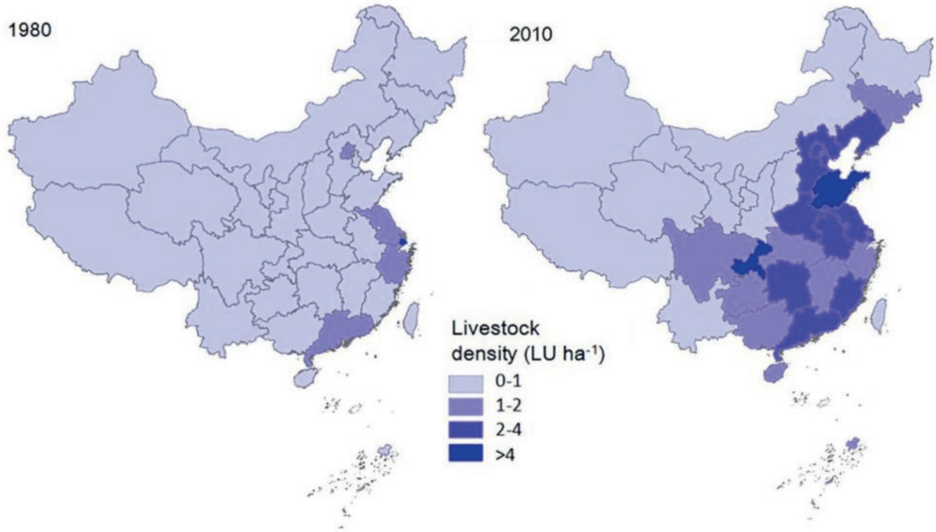

Figure 3.5. Livestock density at province level in China in 1980 and 2010.

Through the decoupling of livestock production and crop production, and the limited areas of cropland around large livestock production farms, the manure N recycling rate (percentage of manure N returned to crop land) decreased from 33% in 1980 to 21% in 2010, a decrease of 36% (Fig 3.2G). The remainder (79% in 2010) was emitted to air as NH_3 and N_2, dropped by livestock in grassland, dumped in landfill, and/or discharged to watercourses without much pre-treatment. As a consequence, many rivers, lakes and coastal waters, and air have become polluted by manure nutrients, while the use of synthetic fertilizer N, in percentage of total N use, increased from 17 to 25% (Fig 3.2H). Further, animal production became unevenly distributed between regions (Fig 3.5). In 1980, the livestock density was below 1 LSU ha^{-1} in most regions; only four provinces and cities (Beijing, Jiangsu, Zhejiang, and Guangdong) had a livestock density >1 LSU ha^{-1}. In 2010, many provinces in the southeast had a high livestock density (>2 LSU ha^{-1}), and most provinces in the north and west still had a low density (<1 LSU ha^{-1}) (Fig 3.5). This highly uneven distribution of livestock production contributed to a decreased effective manure recycling.

3.3.5 Exploring solutions: Responses to different pathways for 2050

We explored livestock production in China in 2050 using two contrasting Shared Socio-economic Pathways (SSPs) scenarios as basis, that is, a SSP2 route, which is a business-as-usual (BAU) scenario, and a SSP1 route (a set of scenarios). The SSP1 route emphasizes technology development, environmental concerns, and system redesign but without a reduction of animal source food consumption. The scenarios followed the main structure of the SSPs, however, with details regarding animal food consumption and livestock production structure predicted in this study. In SSP2, the consumption of animal protein per capita and the total demand for livestock products are projected to increase from 47 to 165%, depending on animal product (Table S1 and Fig S4). The gap between domestic demand and domestic production of livestock products in 2050 will have widened (Table S3), and the import of animal source food and/or the import of livestock feed will have increased. In case of import of animal source food only, China would import in 2050 0.5 to 8.4 times the total global trade of livestock products of 2010, depending on animal product (Table S3). In case of import of livestock feed only, China would import in 2050 0.7 to 1.4 times the total global trade of livestock feed in 2010 (Table S3). The choice for either import of animal source food or livestock feed has impact on the development of the livestock production sector and the associated global environmental burdens. The production of all animal source food domestically will increase GHG emissions in SSP2 from 520 Tg CO_{2e} in 2010 to 805 Tg CO_{2e} in 2050 (Fig 3.6C). Similarly, NH_3 emission will increase from 7.6 to 11 Tg in SSP2, and N losses to watercourse will increase to 18 Tg (Fig 3.6D).

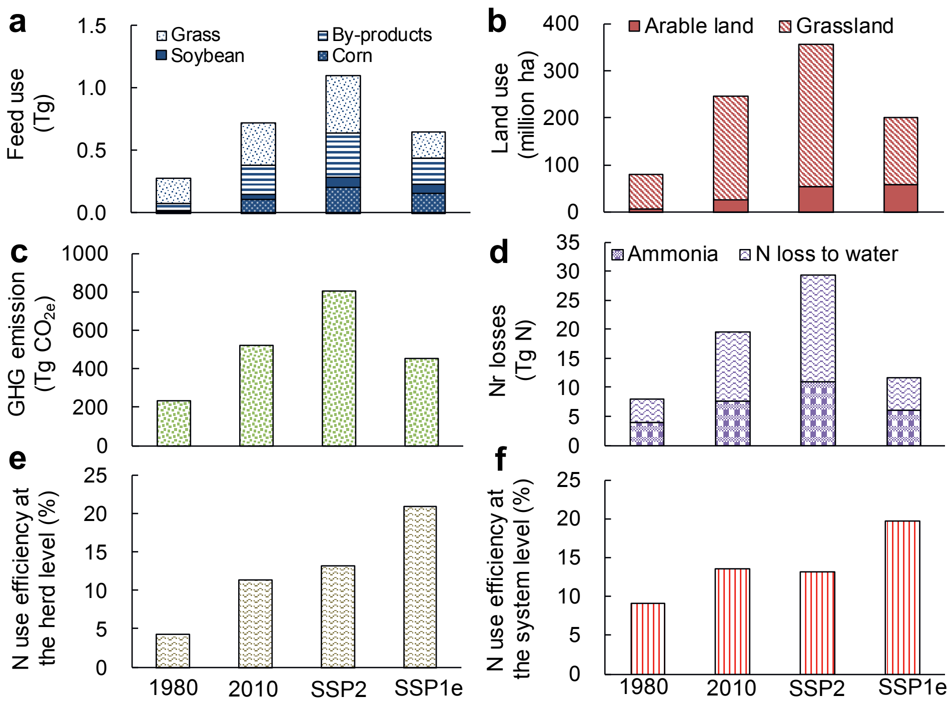

Figure 3.6. Changes in livestock production performance between 1980 and 2010, and forecasts for 2050 following the Shared Socio-economic Pathways SSP2 and SSP1e: total feed use (a), total land use (b), total greenhouse gas (GHG) emissions (c), total reactive N (Nr) losses (d), nitrogen use efficiency (NUE) at herd level (e), and NUE at whole system level (f).

Note: SSP2, shared socio-economic pathway 2; SSP1e, shared socio-economic pathway 1e (combined technologies). Soybean includes both soybean and processed e.g. soybean cake.

SSP1 offers the potential of an environmentally more sustainable path toward 2050 through improved feed and herd management (SSP1a), whole chain manure management to couple crop production with livestock production (SSP1b), production structure change toward more intensification (SSP1c), production structure change toward more monogastric animals (SSP1d), and combined technologies (SSP1e). Combined technologies showed overwhelming superiority compared to the single options (Fig S5). The combined technology SSP1e pathway leads to improvements in both crop (feed) and livestock production efficiencies, while a reconnect of crop and livestock production will allow improved utilization of manure nutrients and a reduction in competition for human edible food as livestock feed. As a result, the corn consumption in 2050 will decrease from 207 Tg in SSP2 to 160 Tg in SSP1e. Similarly, the need for soybean decreases by 11 Tg and that of forage (grass) by 248 Tg in SSP1e relative to SSP2. Further, GHG emissions will decrease from 805 Tg CO_{2e} in SSP2 to 451 Tg CO_{2e} in SSP1e, and Nr losses from 30 Tg N in SSP2 to 12 Tg N in SSP1e (Fig 3.6C, D). The herd- and system-level NUE will increase in SSP1e by 58 and 49% relative to SSP2, respectively (Fig 3.6E, F). Hence, the prospect of combined technologies and systems redesign in SSP1e is large, relative to the BAU of SSP2.

3.4 Discussion

China's livestock transition between 1980 and 2010 has been unprecedented in the world in terms of scale, speed, and global impact. It has changed the consumption of animal source food from a luxury for the few and a delicacy during the Spring Festival to a common food for almost a billion people. The system change increased livestock productivity, in terms of feed and land use per unit of produced animal protein, and at the same time its environmental burden. The transition was both demand- and supply-driven and as yet without much regulation of its environmental impact. The livestock transition in China is part of the global livestock revolution but unique in terms of scale, speed, impact, and driving forces. Within 30 years, total LSUs number tripled, and the number of LSUs in landless industrial-scale systems increased 70-fold. These relative increases have occurred, for example, for dairy cattle in some Arabic countries (through import of high-genetic merit cattle) (FAO, 2013), but these increases have not been reported across all main livestock categories (cattle, pigs, and poultry) and combined with profound system changes, from mixed and backyard systems to landless industrial systems.

3.4.1 Driving forces of China's livestock transition

Diet change has contributed more to the rapid increase in livestock number than the increase of the human population (Fig S3). Developments in livestock sciences and technology, including breeding, molecular genetics, and precision feeding have further contributed to the livestock transition (Li et al., 2008). Direct subsidy policies from the central government have also strongly supported modernization and industrialization of the livestock sector and have stimulated the start-up of livestock production farms (Table S2). The livestock transition has furthermore benefited from loose and ineffective environmental regulations. Construction of large livestock farms near cities was, for example, facilitated even in the case where these farms did not have sufficient arable land nearby for the recycling of manure. Instead, the manure was landfilled or partially treated and then discharged into water bodies (Bai et al., 2016). In summary, the livestock transition in China is propelled by demand and wealth, as described by Delgado et al (1999), but is also strongly facilitated through subsidies, deregulation policies, and weak environmental regulations.

3.4.2 Impacts on resource use and the environment

The impacts of the livestock transition were profound and large, in terms of livestock productivity, international trade of feed, and environmental pollution. The livestock transition also enhanced competition for agricultural land. Increasing demand of agricultural land may further unbalance land use choices between human needs and ecosystem function (de Fries et al., 2004). The area cropped with corn increased at the expense of the area devoted to wheat and rice in China, due to the increasing demand for livestock feed (NBSC, 2017). This change was facilitated also by (indirect) subsidies to farmers growing corn during the livestock transition. However, the increase in domestic feed production was not sufficient. As a consequence, feed import increased greatly. In 2010, feed import was equivalent to 16 million ha of arable land, which is equal to 45% of China's arable land used for feed production (Fig S6). The massive import of livestock feeds affects the world market, and through changes in commodity prices, it also affects countries in Africa, which also depend on the import of soybean and/or cereals (FAO, 2107). Massive imports also induce large-scale changes in land use in exporting countries, including the deforestation of Amazon in Brazil (de Fries et al., 2010).

Emissions of GHG and Nr per unit of animal produce in 2010 were still much higher in China than in the United States and EU (Eshel et al., 2014; Leip et al., 2015). NH_3 emission from livestock production amounted to 7.6 Tg in 2010, which was 62% of the total NH_3 emission in China (Gu et al., 2015). These emissions substantially contributed to the formation of $PM_{2.5}$ (atmospheric particular matter less than 2.5 mm in diameter) and are in part responsible for the air quality problems in China; an additional factor is the concentration of livestock production near urban areas. Further, livestock farms negatively affect water quality; a significant fraction of the manure N and phosphorus (P) ends up in watercourses. Manure N and P are implicated for their role in the severe eutrophication of major rivers, lakes, and coastal waters (Strokal et al., 2016). Forecasts following the SSP2 route suggest that NH_3 emissions from livestock production will have increased to 11 Tg NH_3-N and N losses to watercourses to 18 Tg N by 2050. These losses significantly contribute to the suggested global maximum losses to be able to stay within planetary boundaries (Steffen et al., 2015). Hence, China's livestock transition has local, regional, and global dimensions.

Manure is a main resource of soil organic matter and nutrients that may improve soil quality and replace mineral fertilizer. However, manure is a source of pollution when not managed properly. During the transition, manure N recycling rate decreased greatly because of the disconnection between crop and livestock production at the national and global level. As a result, use of new N (synthetic fertilizer and feed import) increased in livestock production (Fig 3.2H). Similar results can be found at the global level because of the trade of agricultural products (Lassaletta et al., 2014).

3.4.3 A new livestock transition for sustainable livestock production

Forecasts for 2050 following the SSP2 scenario suggest that the pressures on the environment, resulting from China's increased animal source food demand, will greatly increase and may even threaten the sustainability of global livestock production. If China chose to feed the livestock domestically (no importation of animal source food), then livestock production will exert huge pressure on its scarce resources (for example, land and fresh water) and the environment. China will need to import about 97 to 100% of the current global traded corn and soybean, respectively, if there is no improvement of domestic feed production. This may also affect animal feed use in EU, since it imported around 20% of global traded soybean in 2010 (FAO, 2017). These increases in corn and soybean demand may tempt feed exporting countries to increase the acreage of corn and soybean at the expense of wheat production or at the expense of grassland and forest, as has happened in the past 30 years in Brazil and Argentina (FAO, 2017). Conversely, if all additional needed animal source food will be imported from abroad in the SSP2 scenario, then exporting countries may face similar challenges, as recently discussed for the dairy sector (Bai et al., 2018). Although the efficiency of production is much higher in most exporting countries than in China, exporting countries will also face (biophysical and/or regulatory) limits (Eshel et al., 2014; Leip et al., 2015; Guo et al., 2107; Bai et al., 2018).

Managing animal source food demand can be also an effective strategy to reduce N and P losses from the whole food production and consumption chain (Ma et al., 2013). We did not consider this option in the SSP1 scenarios because managing demand seems less realistic during the first few decades; half of the population still has a relatively low consumption of animal source food, and current governmental incentives and dietary guidelines promote balanced diets with modest portions of animal source food, as adopted in the SSP1 scenarios currently.

Our results show compelling differences between SSP2 and SSP1e (Fig 3.6), with SSP1e as the environmentally preferred and most challenging pathway. This pathway would require another livestock transition built on the current transition with further intensification, improving feed quality, improving herd management and breeding, but with emphasis on improving the agronomic and environmental performance of the whole livestock production sector, and including recoupling of feed-livestock production. Such a livestock transition is difficult to manage without the joint efforts of the livestock sector, suppliers, and the government. Locations of livestock farms have to be planned strategically away from watercourses and other sensitive areas and with sufficient cropland in the vicinity. For proper recycling of manure nutrients, approximately 0.3 to 0.6 ha of agricultural land per LSU is needed, depending on soil fertility level and environmental conditions (Liu et al., 2017). The importance of location of livestock farms was reiterated recently by the Chinese government; pig farmers were expelled from areas near eutrophication-sensitive lakes, following the implementation of the water protection law. Through coupling of crop and livestock production with manure nutrient recycling, large amounts of synthetic fertilizer can be saved (Bai et al., 2016), which is a main policy objective (MOA, 2014). At the same time, NUEs at the system level can be increased, and eutrophication of lakes, coastal seas, and other sensitive areas can be diminished. The additional requirement of land for feed production will be 31 million ha (Fig 3.6B), which is nearly as large as the area of crop land in 2010 (Fig S6). However, the total area of grassland covers more than three times the area of arable land in China, and part of this area has the potential to increase production through better nutrient and water management. Further, closing the crop yield gap through integrated soil-crop management has been shown to be effective for most regions in China; it can increase cereals yield without increasing N input (Chen et al., 2014) and can also contribute to covering the increased feed demand.

The promise of the SSP1e scenario can only be realized through the implementation of advanced designs and technologies in breeding, animal feeding, and manure management, without sacrifice of animal welfare. Key to realizing the promise of SSP1e are as follows: (i) targeted spatial planning of livestock production, (ii) coupling of crop and livestock production, and (iii) improved grassland management and concentrate feed production, with reduced competition with human-edible food. Achieving the SSP1e requires targeted socioeconomic policies, environmental regulations, and large investments in improving livestock feed production and quality, livestock housing, and manure management.

Overall, the livestock transition in China between 1980 and 2010 has had a significant impact on the livestock production sector itself, food consumption patterns of consumers, and the environment and on the international trade of feed. The number of livestock has tripled, the number of livestock in landless industrial systems increased 70-fold, and the proportion of monogastric animal units increased further from 62 to 74% of total LUs. The changes have been driven by demand and supply factors, including subsidy policies and loose environmental regulations. Though GHG and reactive N losses per unit of animal protein decreased significantly, total GHG and Nr losses increased greatly. The loss of N and P contributed seriously to eutrophication of rivers and lakes. Domestic feed production was insufficient, and increasing amounts of soybean, corn, and, more recently, alfalfa had to be imported, which affected global markets. Forecasts for 2050 reveal that yet another livestock transition will be needed, with again a clear role for the central government, retailers, consumers, and the industry itself. The management of the new transition should focus equally on the spatial planning of livestock farms, the improvement of livestock production efficiency, animal feed production (including forage and grasslands), and manure management.

3.5 Acknowledgements

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CHAPTER 4

Global environmental costs of China’s thirst for milk

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Abstract

China has an ever-increasing thirst for milk, with a predicted 3.2-fold increase in demand by 2050 compared to the production level in 2010. What are the environmental implications of meeting this demand, and what is the preferred pathway? We addressed these questions by using a nexus approach, to examine the inter-dependencies of increasing milk consumption in China by 2050 and its global impacts, under different scenarios of domestic milk production and importation. Meeting China's milk demand in a business as usual scenario will increase global dairy-related (China and the leading milk exporting regions) greenhouse gas (GHG) emissions by 35% (from 565 to 764 Tg CO_{2eq}) and land use for dairy feed production by 32% (from 84 to 111 million ha) compared to 2010, while reactive nitrogen losses from the dairy sector will increase by 48% (from 3.6 to 5.4 Tg nitrogen). Producing all additional milk in China with current technology will greatly increase animal feed import; from 1.9 to 8.5 Tg for concentrates and from 1.0 to 6.2 Tg for forage (alfalfa). In addition, it will increase domestic dairy related GHG emissions by 2.2 times compared to 2010 levels. Importing the extra milk will transfer the environmental burden from China to milk exporting countries; current dairy exporting countries may be unable to produce all additional milk due to physical limitations or environmental preferences/legislation. For example, the farmland area for cattle-feed production in New Zealand would have to increase by more than 57% (1.3 million ha) and that in Europe by more than 39% (15 million ha), while GHG emissions and nitrogen losses would increase roughly proportionally with the increase of farmland in both regions. We propose that a more sustainable dairy future will rely on high milk demanding regions (such as China) improving their domestic milk and feed production efficiencies up to the level of leading milk producing countries. This will decrease the global dairy related GHG emissions and land use by 12% (90 Tg CO_{2eq} reduction) and 30% (34 million ha land reduction) compared to the business as usual scenario, respectively. However, this still represents an increase in total

GHG emissions of 19% whereas land use will decrease by 8% when compared with 2010 levels, respectively.

4.1 Introduction

The increased international trade of agricultural products has received much attention recently due to the impacts of production on land use, deforestation and associated biodiversity loss, impaired nutrient cycling, and greenhouse gas (GHG) emissions. Currently, around 23% of the food produced for human consumption is traded internationally (D'Odorico and Carr, 2014). It has been estimated that the global trade of nitrogen (N), embedded in the products, has increased from 3 to 24 Tg N between 1961 and 2010, with the largest contributor relating to animal feed (Lassaletta et al., 2014). Oita et al (2016) analyzed the reactive N emitted during the global production, consumption and transportation of commodities, and estimated that 15% of the global N footprint is from commodities internationally traded. Exportation of beef, soybeans (Glycine max), and wood products was responsible for 12% of the deforestation in seven countries with high deforestation rates (Henders et al., 2015). Additionally, up to 30% of global species threats are due to international trade, via production of commodities in export countries (Lenzen et al., 2012) and 17% of global biodiversity loss occurs due to commodities destined for exportation (Chaudhary and Kastner, 2016).

The trade of milk will likely increase strongly during the next decades due to the increasing demands from China and some other rapidly developing countries, for example, India (Alexandratos and Bruinsma, 2012). In 2013, around 125 Tg milk was traded between countries, which was an 8-times increase since 1961, and equal to 20% of the global milk production (Food and Agriculture Organization (FAO), 2016). European Union (EU), New Zealand (NZ) and United States of America (USA) were the top three milk exporting region and countries, accounted for more than 80% of total export in 2013 (FAO, 2016). Currently, China is the leading milk importer, importing 12 Tg fresh milk equivalent in 2013, which was 123-times larger than that in 1961, and equal to 25% of the domestic consumption in 2013 (FAO, 2016).

Globally, consumption of animal products is driven by culture, population growth and prosperity (gross domestic production, GDP), with high GDP countries consuming on average higher amounts per capita (Tilman et al., 2011; Tilman and Clark, 2014). This holds also for milk, but with significant variation between countries (Fig S1). It is projected that global milk consumption will increase by 60% between 2010 and 2050, especially in traditionally lower consumption regions, such as China (Alexandratos and Bruinsma, 2012). Historically, China had low milk consumption per capita (<2 kg capita^{-1} year^{-1} in 1961, partially due to the severe food crisis), but given the growth of its economy and urbanization rate, milk consumption has increased over 25-times during the past 5 decades, leading to China becoming the world's fourth-largest milk producer (FAO, 2016). Milk consumption and importation is likely to increase further in China, because of population and GDP growth and urbanization (Wang et al., 2017; Fig S2), and a halt of milk production due to the reduction in small traditional dairy production units (<5 head farm; Fig S3), which facilitates the milk quality control.

China became the world's largest milk importer in 2010, following the melamine scandal in 2008 which eroded public confidence in domestically produced milk (Pei et al., 2011; FAO, 2016). China also imports massive amounts of soybean and increasing amounts of maize (*Zea mays*) and alfalfa (*Medicago sativa*) to feed its increasing domestic pig, poultry, and dairy cattle populations (FAO, 2016). The increasing imports of animal feed are related to the increasing domestic consumption of animal derived food and to the relative scarcity of agricultural land and fresh water. Meanwhile, EU abolished its milk quota system in 2015, and New Zealand and Chile are preparing for the projected increase in milk demand from China and other rapidly developing countries, for example, India (European Commission, 2014; Oenema et al., 2014). The impact of China's thirst for milk related to resource demands, climate change, eutrophication, and biodiversity loss need to be predicted so pathways for a more sustainable solution can be mapped. China is facing both food security and water security challenges as well as vast environmental challenges, which underpin the importance of researching alternative future projections (Piao et al., 2010; Liu and Yang, 2012).

Here, we present the results of a novel nexus approach to examine the interdependencies of increasing milk consumption in China and its impact on GHG emissions, N losses, land and water use, and economic performances across the main feed and milk producing countries. Dairy cattle account disproportionally to GHG emissions, predominately because of enteric fermentation and the release of methane (CH_4; Steinfeld et al, 2006; Gerber et al., 2013). We analyzed the interrelationships and interdependencies of the whole ‘production-consumption-trade’ system for 2050 under contrasting Shared Socio-economic Pathway scenarios (SSP): (i) Business as usual (BAU) – increase in milk consumption in 2050 aligned to current proportional contributions of domestic production and import (SSP2), (ii) Produce all additional milk domestically (PA) – increase in milk consumption in 2050 delivered through increased domestic output (SSP3), and (iii) Import all additional milk (IM) – increase in milk consumption in 2050 delivered through increased imports from three leading producing regions (EU, USA, NZ; SSP5). Furthermore, we evaluated two extra scenarios following the Shared Socio-economic Pathway 1 (SSP1) storyline, which focuses on technological improvements: (i) Dairy Production Improvement (DPI) - assuming that productivity and manure management in China can reach the current level of the leading milk exporting countries by 2050; and (ii) Farming Systems Improvement (FSI) – toward crop-dairy integration and forage-based systems with increased productivity of forages, building on scenario DPI.

4.2 Material and methods

The approach we took was to split the study into four carefully defined areas to perform the assessment: i) determine the factors which will drive the prediction of milk consumption in China; ii) set the system boundary of the study; iii) assign and calculate multiple sustainability indicators (one economic, three physical and four environmental indicators); iv) describe the scenarios to be tested to meet the demand and to analyze the consequent impact on the sustainability indicators. For more information please see the supplementary information (SI), and SI is available online.

4.2.1 Prediction of milk consumption in China

We estimated average per capita milk consumption in 2050 using different sources and the following assumptions. First, we calculated the relations between average milk consumption per capita and average GDP per capita, and milk consumption per capita and urbanization rate (Fig S2). Milk consumption in 2050 was then estimated assuming a mean GDP of 10,904 \$ capita^{-1} yr^{-1} and an urbanization of 78% in 2050 (FAO, 2016; World Bank, 2016). Second, a predicted increase in average milk consumption of 1.80% yr^{-1} in developing countries between 2005 and 2050 (Alexandratos and Bruinsma, 2012). Third, following the national guidelines for a healthy diet, the average milk consumption is 300 g capita^{-1} d^{-1} in 2050 (CSN, 2014).

Total milk consumption was calculated as:

$$\text{Milk}_{\text{total}} = \text{Population} * \text{Milk}_{\text{average}} \tag{1}$$

Where, Milk$_{\text{total}}$ is the total milk consumption in kg, Population is the total human population, and Milk$_{\text{average}}$ is the average milk consumption in kg capita^{-1}, calculated using the three assumptions outlined above. Forecasts suggest that the human population will be 1.4 billion in China in 2050 (FAO, 2016).

Table 4.1. Greenhouse gas (GHG) emissions, reactive nitrogen (Nr) losses (including losses during feed production), land and irrigation water requirement for feed production, feed requirement, production cost, and N and P excretion by dairy cattle in China, New Zealand, European, and United States. The references are indicated with the number. The figures without superscript are derived from calculations with the NUFER model.

	China						New Zealand	European	United States
	2010	BAU	PA	IM	DPI	FSI			
GHG (kg CO_2eq kg^{-1} milk)	2.9	2.9	2.9	2.9	1.9	1.9	2.1[1]	1.6[1]	1.9[1]
Nr losses (g N kg^{-1} milk)	34	31	31	32	11	10	12[2]	9.0[3]	12[4]
Land requirement (m^2 kg^{-1} milk)	5.2	2.4	2.1	3.8	1.9	1.9	1.3[2]	2.5[5]	1.9[6]
Blue water requirement (m^3 kg^{-1} milk)	145	206	213	173	57	51	48[7]	46[7]	60[7]
Feed requirement (kg DM kg^{-1} milk)	2.6[8]	1.7	1.6	1.9	0.9	1.1	1.1[9]	1.2[9]	0.9[9]
Costs ($ t^{-1} milk)	445[10]	445	445	445	372	383	376[10]	418[10]	360[10]
N excretion (g N kg^{-1} milk)	32[8]	28	28	30	20	24	30[11]	20[12]	18[13]
P excretion (g P kg^{-1} milk)	5.6[8]	4.5	4.4	4.7	2.6	2.8	2.2[14]	3.0[12]	2.5[15]

1. Opio et al., 2013; 2. Flysjö et al., 2011; 3. Leip et al., 2014; 4. Powell et al., 2010; 5. Lesschen et al., 2011; 6. Eshel et al., 2015; 7. Mekonnen and Hoekstra, 2011; .8. Bai et al., 2013; 9. Appuhamy et al., 2016; 10. FAO, 2016; 11. de Klein et al., 2005; 12. Velthof et al., 2015; 13. Powell et al., 2006; 14. Monaghan et al., 2007; 15. Powell et al., 2006.

Scenarios: BAU (SSP2): Business as usual, with a milk self-sufficiency of 75%; PA (SSP3): Produce all additional milk in 2050 domestically; IM (SSP5): Import all additional milk in 2050; DPI (SSP1a): Dairy production Improvement, on top of BAU; FSI (SSP1b): (Farming system improvement, on top of DPI.

4.2.2 System boundary

Milk import was assumed to be from the current top three milk exporting regions, namely: EU, NZ and the USA in 2010 (FAO, 2016). The resource requirements (feed, land and water) and environmental performance (GHG emissions, reactive N (Nr) losses, N and phosphorus (P) excretions) parameters related to dairy production in these countries were collected from peer-reviewed published literature, and then used to calculate the domestic and global impacts of supplying the calculated 2050 milk demand in China (Tables 4.1, S2-3).

4.2.3 Determining the sustainability indicators to be used in the assessment

A total of eight indicators at the herd level (accounting for lactating cow, heifers and calves, dairy related beef production was not considered), with three physical indicators (feed, land and water requirement), one economic indicator (GDP value of milk production) and four environmental impact indicators (GHG emissions, reactive N losses, and N and P excretions), were selected to evaluate the impacts of the projected increase in milk consumption and production. The economic value of milk production was derived from the milk production price in 2010 recorded in the FAO database and used as an indicator of the economic importance, assuming that the milk price will remain more or less constant (FAO, 2016). In practice, milk price will depend on the balance of milk demand and supply, which will depend on many factors and opportunities, however a basal value is required to assess economic performance. Feed requirement and the related land and water requirements to produce the feed were used as indicators for resource use. Emissions of GHG and Nr and the production of manure N and P were chosen as agri-environmental impact indicators, as China is facing severe challenges associated with current emissions and associated climate change, nutrient losses and manure management problems (Bai et al., 2016).

4.2.4 NUFER-dairy model

The resource use and environmental effects of different dairy production systems in China were calculated by the NUFER-dairy model (Bai et al., 2013; Zhang et al., 2017). The model has been developed to quantitatively evaluate GHG emissions, nutrient flows, and land, water and feed resource requirements for various systems of operation at animal, herd, and system levels. The model consists of an input database, a calculator, and an output module. The input database includes herd demographics, milk yield and feed composition. The calculation module includes a feed intake prediction sub-module and a nutrient balance sub-module. Calculation of feed intakes by calves, heifers, and milking cows are based on the energy requirements. The nutrient balance is calculated from the nutrients flows through the whole soil-feed-milk production chain. The output module provides results for land, water and feed use, N losses and GHG emissions (Bai et al., 2013; Zhang et al., 2017).

Table 4.2. Key parameters of different dairy production systems for different scenarios.

		BAU	PA	IM	DPI	FSI
Domestic milk self-sufficiency rate (%)		75[1]	100[1]	33[1]	75[1]	75[1]
Share of grazing, medium size and industrial system to domestic milk production (%)		6, 13, 81[1]	4, 10, 86[1]	14, 30, 56[1]	6, 13, 81[1]	33, 33, 33[1]
Crop and dairy integration rate		Low[1]	Low[1]	Low[1]	High[1]	High[1]
Yield of selected feed (t ha^{-1})	Corn	5.5[2]	5.5[2]	5.5[2]	5.5[2]	9.2[3]
	Soybean	1.8[2]	1.8[2]	1.8[2]	1.8[2]	2.0[3]
	Grass	1.0[4]	1.0[4]	1.0[4]	1.0[4]	3.0[4]
Importation rate of selected feed (%)	Corn	3.9[2]	3.9[2]	3.9[2]	3.9[2]	0[1]
	Soybean	85[2]	85[2]	85[2]	85[2]	85[1]
	Alfalfa	10[1]	11[1]	6.2[1]	19[1]	0[1]

1. This study; 2. FAO, 2016; 3. Chen et al., 2014; 4. Eisler et al., 2014.

Scenarios: BAU (SSP2): Business as usual, with a milk self-sufficiency of 75%; PA (SSP3): Produce all additional milk in 2050 domestically; IM (SSP5): Import all additional milk in 2050; DPI (SSP1a): Dairy production Improvement, on top of BAU; FSI (SSP1b): (Farming system improvement, on top of DPI.

4.2.5 Three physical indicators (feed, land and water)

4.2.5.1 Feed requirement

The feed requirement of dairy cattle was calculated as follows:

$$Feed_{total} = Milk_{produced} * Feed_{milk} \tag{2}$$

Where Feed $_{total}$ is the total feed requirement (dry matter) in kg, milk $_{produced}$ is the total milk produced in each region in kg, and Feed $_{milk}$ is the feed to milk conversion ratio in kg kg^{-1} (Tables 4.1, S1). The feed conversion ratio of China's dairy production was calculated per production system and their contribution to the total milk production (Table S2). The feed conversion values for NZ, EU and USA were derived from a literature review (Appuhamy et al., 2016), and are shown in Table 4.1.

4.2.5.2 Land requirement

The agriculture land required for dairy production was calculated from total milk production and the average land demand kg^{-1} milk.

$$Land\ requirement = Milk_{produced} * Land\ requirement_{milk} /10000 \tag{3}$$

Where, Land requirement is the area of arable land and grassland required for feed production, in ha. Land requirement $_{milk}$ is the average area of land needed to produce 1 kg of milk, in m^2 kg^{-1} milk. The area of arable land and grassland for producing feed for China's dairy production was calculated using total feed requirement (excluding the imported feed), and average crop and grassland yields. Information about the land requirement in the three milk exporting countries is listed in Table 4.1.

4.2.5.3 Water use

The water use was obtained by calculating the blue water (from surface and ground waters, for irrigation) use for milk production:

$$Water = Milk_{produced} * Water_{milk} \quad (4)$$

Where Water is the total water requirement in m^3; Water $_{milk}$ is the mean blue water use for milk production in m^3 kg^{-1} milk. The blue water use of China's dairy production covered the blue water demand of related feed production, i.e. 74 m^3 t^{-1} maize, 129 m^3 t^{-1} soybean, 387 m^3 t^{-1} rice, and 455 m^3 t^{-1} wheat (Mekonnen and Hoekstra, 2011). These figures do not include the demand for drinking and service water, due to lack of information and their small contribution (<5%) to the total water footprint (Mekonnen and Hoekstra, 2012). The blue water use for milk production by the three main milk exporters was derived from literature (Table 4.1). Here, differences in crop water use efficiency associated with different scenario assumptions have not been considered.

4.2.6 One economic indicator (GDP value of milk production)

4.2.6.1 Economic value

The economic value of dairy production was calculated according to the average milk production value in 2010.

$$Economic\ value = Milk_{produced} * Costs_{milk} \quad (5)$$

Where, Economic value is the total economic value of produced milk in US$ in 2010; Costs $_{milk}$ is the average production cost of milk, derived from FAO database in US$ t^{-1} milk. The average milk production cost was 445, 376, and 360 US$ t^{-1} milk for China, NZ and USA, respectively in 2010. For EU, we used a weighted average value, which was 418 US$ t^{-1} milk in 2010 (Table 1). The job opportunities provided by dairy production was calculated from the total GDP of dairy production, and assuming an income of 18,000 Yuan $person^{-1}$ in 2010 (China Statistic Yearbook, 2011).

4.2.7 Four impact indicators (GHG emissions, N losses, N and P excretion)

4.2.7.1 GHG emissions

The GHG emissions (CO_2, CH_4, and N_2O) from the soil-feed-dairy production and feed-milk transportation chains were calculated as:

$$GHG = Milk_{produced} * GHG_{milk} + Milk_{export\ to\ China} * GHG_{milk\ export} \quad (6)$$

Where GHG is the total GHG emissions of dairy production in kg CO_2 equivalents (CO_2eq), Milk $_{produced}$ is the amount of milk produced in each region (China, EU, USA, and NZ) in kg. GHG $_{milk}$ is the carbon footprint in kg CO_2eq kg^{-1} milk. Milk $_{export\ to\ China}$ is the amount of milk exported to China by the top three milk exporting regions (weighted values) in 2010. GHG $_{milk\ export}$ is the GHG emissions associated with the transportation of milk to China. Milk $_{total}$ is listed in Table S1, and GHG emissions parameters are presented in Table 1. The GHG emissions related to the transportation of milk to China were based on the average transport distance of milk to China from NZ, EU (the Netherlands) and USA, 11,144, 7,821 and 11,100 km, respectively (Food Miles, 2016). The average GHG emissions rate was 0.0345 kg CO_2eq ton^{-1} km during shipping (Van Passel, 2013). We assumed that all the milk export to China was as milk powder, as only 2% of the milk transported to China was as fresh milk in 2010 (FAO, 2016). The average fresh milk to dry milk conversion ratio was set at 7:1.

4

4.2.7.2 Nr losses

Nr losses were based on the average Nr losses and milk production of different dairy production systems calculated by NUFER-dairy (Table S2). In scenarios, Nr losses were weighted per their share of total dairy production (Tables S3). Nr losses of leading milk export regions were collected from the literature (Table 4.1). In our calculations, the following Nr losses have been considered: nitrate leaching to groundwater and surface waters and emissions of N_2O and ammonia (NH_3) to the atmosphere, from animal housing, manure management and soils.

$$Nr\ losses = Milk_{produced} * Nr\ losses_{milk} \quad (7)$$

Where Nr losses are the total Nr losses of dairy production in kg. Nr losses $_{milk}$ are the Nr losses per kilo of milk in kg kg^{-1} milk, data for China see

Table S2 and for other regions see Table 4.1. The Nr losses were assessed at the system level (soil-crop-dairy), and included the losses during feed production.

4.2.7.3 N and P excretions

The N and P excretions by dairy cattle were calculated as:

$$N(P)\ excretion = Milk_{produced} * N(P)\ excretion_{milk} \quad (8)$$

Where N(P) excretion is the total amount of manure N(P) produced by dairy cattle in kg yr^{-1}, N(P) excretion $_{milk}$ is the average N(P) excretion per kilo of milk produced, in kg (Table 4.1).

4.2.8 Feed use and import, and related virtual land import

Consumption of different feed items was calculated as follows:

$$Feed_{items} = Feed_{total} * Feed_{composition} \quad (9)$$

Where, Feed $_{items}$ is the consumption of different feed items, i.e. maize, soybeans, and alfalfa, in kg. Feed $_{total}$ is calculated by Equation 5. Feed $_{composition}$ is the feed composition used in different countries in % of Feed $_{total}$. Feed composition was collected from published studies; Bai et al (2013) for China, Hou et al (2016) for EU, and Herrero et al (2013) for NZ. The feed import in 2010 was derived from FAO database (Table S4). No dairy feed was imported into USA. Feed import related land virtual import was calculated based on the feed import and feed productivity in the feed export regions, which were derived from the FAO database.

4.2.9 Development of scenarios

4.2.9.1 Business as usual scenario (BAU - Milk self-sufficiency maintained at 75%)

This followed the SSP2 storyline that social, economic and technological trends do not shift markedly from historical patterns (O'Neill et al., 2016). Therefore, we assumed that milk self-sufficiency in 2050 will be maintained at the current level (75%) (FAO, 2016). The milk imported will come from the current top three global milk exporters: EU (77%), NZ (13%), and the

USA (10%) (FAO, 2016). Domestic milk will be provided by grazing systems, medium size systems and industrial systems; following current trends in dairy production, their relative contributions will be 6, 13, and 81%, respectively (Table 4.2). We assumed that the 'traditional' dairy system (≤9 head cattle per farm) will have disappeared by 2050 (MOA, 2015).

4.2.9.2 Scenario: Produce All (PA) – Milk self-sufficiency will increase to 100%

Scenario PA considered that all required milk will be produced domestically, following the SSP3 storyline with governmental policies focusing on national food security. Relative milk production contributions from grazing, collective and industrial systems were assumed to be 4, 10, and 86%, respectively, based on current trends (Table 4.2). We assumed again that the 'traditional' dairy system (≤9 head cattle per farm) will have disappeared by 2050.

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4.2.9.3 Scenario: Import Milk (IM) – Milk self-sufficiency will drop to 33%

The IM scenario assumes that domestic milk production will remain at the level in 2010 and that all additional milk will be imported. As a result, milk self-sufficiency will drop to 33%. Relative milk production from grazing, collective and industrial systems is assumed to be 14, 30, and 56%, respectively (Table 4.2). Imported milk was assumed to be supplied by the same three countries with the same proportion as in BAU (Table S1).

Table 4.3. List of strategies for sustainable pathways of dairy production in China.

	Feed production	Dairy production and manure management
Research, scientists' strategy	**Level 1:** Integrated Soil-crop System Management technology (ISSM) to improve crop productivity[1]; **Level 2:** Improve nutrient	**Level 1:** Genetic improvements to increase milk productivity, i.e. build up the national dairy herd improvement data source[6];

	management in grasslands and production of grass in southern China to boost the high quality grass production[2,3]; **Level 3:** Design new human-edible feeds; and design forage and crop production systems in China, i.e. rice-grass rotation in southern China, maize-rye grass rotation in northern China to increase grass production[4]; **Level 4:** Water saving irrigation systems to boost feed production in northern and western China[5].	build up the nucleus group; adapt the sex-sorted sperm and embryo transfer technologies[7]; import high performances breeds from abroad. **Level 2:** Feed improvement, i.e. using the high quality roughages, whole corn silage and alfalfa silage; total mixed ration feed; improve the quality of corn silage[8]. **Level 3:** Herd management, i.e. improved reproduction; select the high performances calves and heifers; decrease the mortality rate; increase disease control and animal welfare control.
Implementation policies	**Level 1:** Economic incentives to adopt new technology; **Level 2:** Incentives to design sustainable farming system, for example incentives for grass production and processing; **Level 3:** Training and extension services to improve dairy farmer's knowledge of feed production; **Level 4:** Incentives for integrated dairy cow and feed production.	**Level 1:** Strict restrictions of milk quality for milk production and recycle of manure; **Level 2**: Incentives for importing high performance dairy cows and forage breeds; **Level 3:** Incentives for high technique manure management equipment and machinery, to couple crop-dairy production; **Level 4**: Build up more effective extension services or farm organizations, i.e. pioneer dairy farm to test the advanced technologies and training the farmers

1. Chen et al., 2011; 2. Li et al., 2007; 3. Li and Lin., 2014; 4. Pan et al., 2007; 5. Deng et al., 2006; 6. Zhou et al., 2012; 7. Xu et al., 2006; 8. Wang et al., 2009.

4.2.9.4 Scenario: Dairy Production Improvement (DPI) – Improved feed, herd and manure management - Milk self-sufficiency maintained at 75%

The DPI scenario follows the SSP1 storyline that the world shifts toward a more sustainable path, emphasizing more inclusive development, with improvements in agricultural productivity and rapid diffusion of best practices (O'Neill et al., 2016). We assumed that China's grazing systems will reach NZ's current level by the end of 2050 (both in terms of milk production efficiency and environmental performance, but not the feed production efficiency, see Table 4.2). Similarly, we assumed that China's collective dairy farms will get close to the EU's current production efficiency and that China's industrial dairy farms will have caught up with the current performance of USA's large dairy operations. Thus, under this scenario, the grazing, collective and industrial dairy production systems were assumed to have a similar production, economic and environmental performance as the corresponding dairy production systems in NZ, EU and the USA. Especially for the integration of dairy and feed production, since the disconnection of crops and livestock could reduce efficiency at the system or global level even with significant improvements in efficiency at the herd level (Bai et al., 2014; Lassaletta et al., 2016). Strategies for improved dairy production efficiency and environmental performance are listed in Table 4.3.

4.2.9.5 Scenario: DPI with Farming Systems Improvement (FSI) - Milk self-sufficiency maintained at 75%

Scenario FSI builds on scenario DPI, while assuming that all milk will be produced in equal portions by grazing, collective and industrial systems, due to the concern of arable land competition, increased natural grassland utilization and manure local recycling issues. Domestic forage and feed production will have increased to a level that no forage and feed has to be imported (except for soybean). Mean grass yields will have increased from 1.0 to 3.0 t ha^{-1} (Eisler et al., 2014). Yields of cereals can be improved through Integrated Soil-crop System Management technology (ISSM) with nutrient inputs similar to current levels; we assumed that mean crop yields will increase from 5.5 to 9.2 t ha^{-1} for maize, from 6.5 to 7.7 t ha^{-1} for rice and from 4.7 to 6.9 t ha^{-1} for wheat between 2010 and 2050 (Chen et al., 2014; FAO, 2016). Strategies for improved feed production are listed in

Table 4.3.

Note that BAU, PA and IM scenarios shared similar technological level, where the differences in indicators were due to differences in the share of the dairy production systems in China, except for production price which was due to lack of information (Table 4.1).

4.3 Results

4.3.1 Prediction of average milk consumption in China in 2050

Current milk consumption in China is 31 kg capita^{-1} year^{-1}. We estimated the average milk consumption per capita in 2050 based on various sources of information and assumptions. The predicted value was the smallest based on the FAO prediction (56 kg/capita) and the highest when based on the national guidelines (110 kg/capita). Evidently, there is a wide range between these estimates, with an average of 82 kg/capita based on all projections (Fig 4.1).

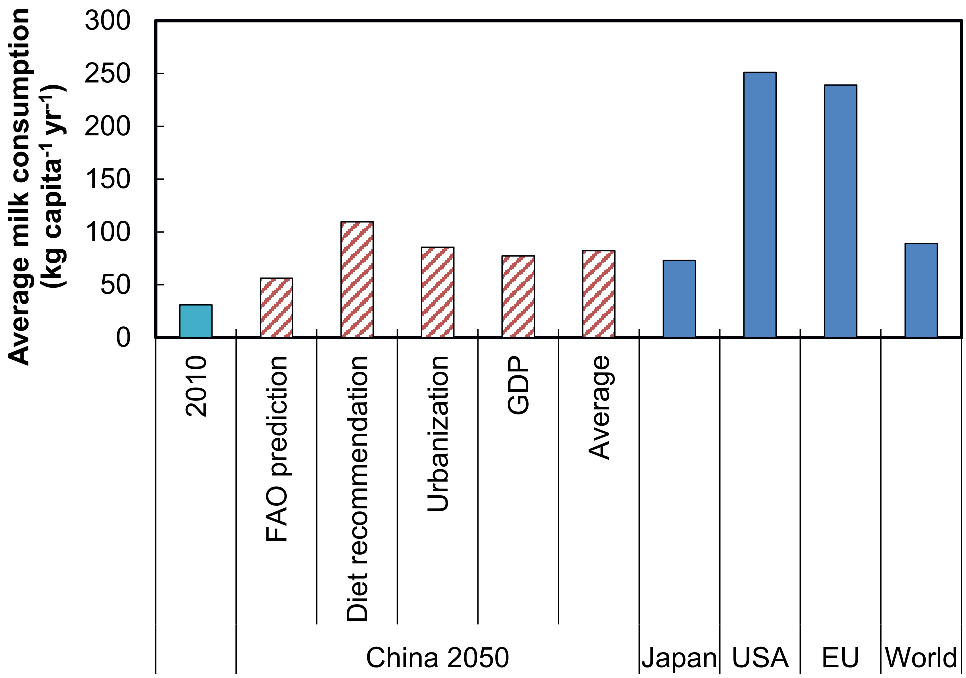

Figure 4.1. The estimated average milk consumption in China in 2050 based on four different estimation methods, in comparison to the current (2010) milk consumption levels in China, Japan, United States of America (USA), Europe (EU), and the world.

4.3.2 Expected impacts of increased milk consumption - Scenario BAU

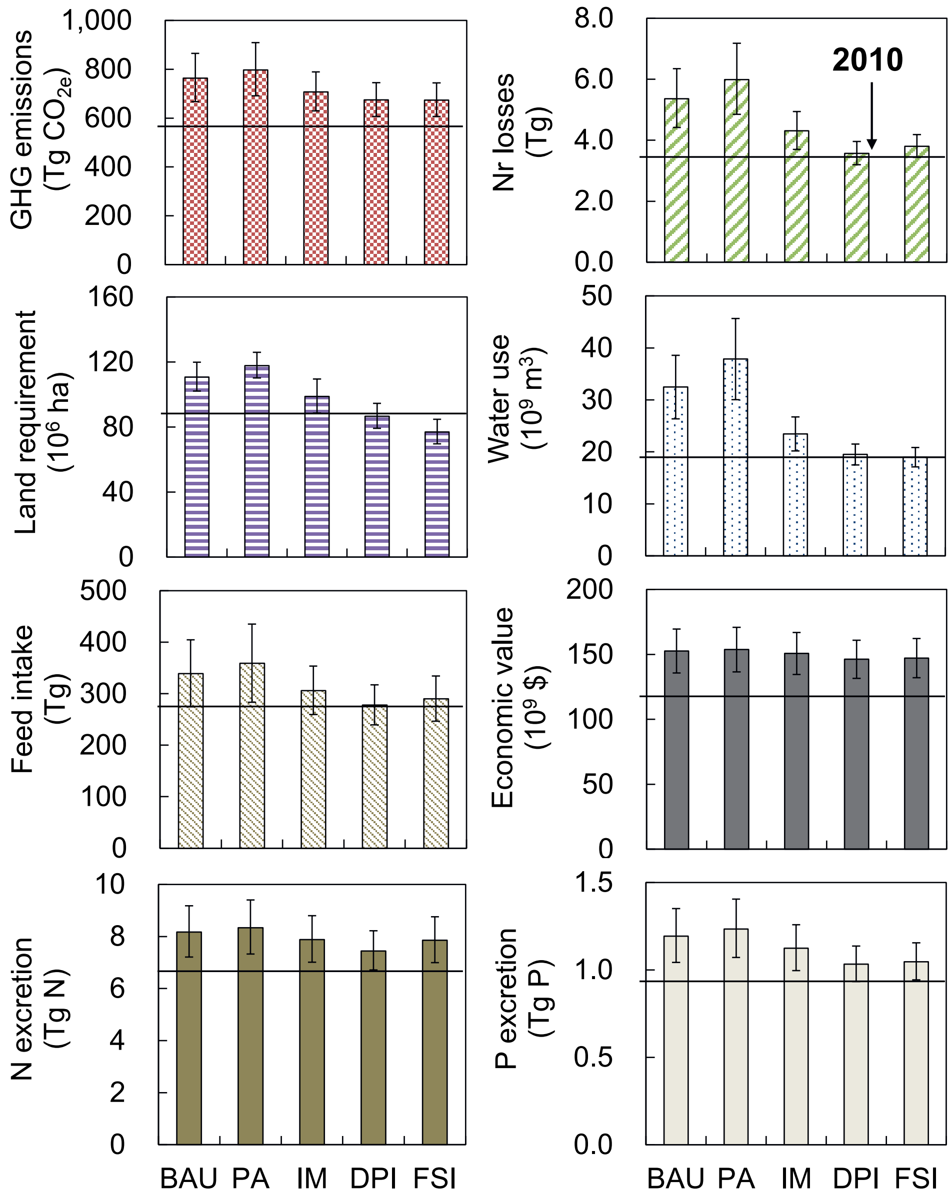

Figure 4.2. Impacts of increased milk consumption for the global dairy production (China and with three leading milk export regions) by 2050; results of 5 scenarios (BAU, PA, IM, DPI and FSI), i.e., greenhouse gas (GHG, Tg CO_2 equivalent) emissions, reactive nitrogen (Nr, Tg N) losses, land requirement (million ha), irrigated water requirement (billion m^3), animal feed intake requirements (Tg dry matter), economic value (billion $), nitrogen excretion (Tg N) and phosphorus excretion (Tg P) in the four countries considered in this study (China, European Union, New Zealand, United States of America). The solid lines represent the situation in 2010. The error bars reflect the expected lowest and highest milk consumption in 2050.

All the indicators were calculated based on the total milk production in all the selected regions. Scenarios: BAU (SSP2): Business as usual, with a milk self-sufficiency of 75%; PA (SSP3): Produce all additional milk in 2050 domestically; IM (SSP5): Import all additional milk in 2050; DPI (SSP1a): Dairy production Improvement, on top of BAU; FSI (SSP1b): (Farming system improvement, on top of DPI.

Total milk production of the global dairy production and supply group (China and the leading milk exporting regions) will increase by 28% compared to 2010, and reach up to 375 Tg in BAU scenario. Total milk consumption in China will be 116 Tg in 2050 (range 80–155 Tg), which is around 3.2-fold the milk production level of 2010 (Table S1). The additional milk demand was assumed to be supplied by industrial production systems. Results of the BAU scenario show that the global dairy-related GHG emissions will increase by 18%–53%, with an average value of 35% (increase from 565 Tg CO_{2eq} in 2010 to 764 Tg CO_{2eq} in BAU) compared with 2010 (Fig 4.2a). The land needed for feed production will increase by 32% (from 84 to 111 million ha; Fig 4.2c). Water use and Nr losses related to dairy production will increase by 77% (from 18 to 33 billion m^3) and 32% (from 3.6 to 5.4 Tg N) when compared to 2010, respectively (Fig 4.2b,d). China's domestic dairy-related GHG emissions and total Nr losses will be tripled (Fig 4.3a, b).

4.3.3 Expected impacts of increased milk consumption – Scenario PA

Producing all additional milk domestically (PA) with current technology and management, will increase total dairy related GHG emissions (China, EU, NZ and USA) by 34 Tg CO_{2eq}, compared to BAU (Fig 4.2a). PA will boost the Chinese dairy sector to nearly 52 billion US $, and substantially increase domestic employment opportunities compared to BAU (Fig 4.3e, Fig S6). However, without major improvements in domestic feed production (yield and quality), it will need to import 8.5 Tg of cereals and protein-rich crops (mainly from USA and Brazil), and 6.2 Tg forages (mainly from USA and Canada; Table 4.4). The demand of land for feed production will increase by 6% (equal to 7.1 million ha), irrigation water by 17% (equal to 5.4 billion m^3 blue water), Nr losses by 12% (equal to 0.6 Tg N) and nutrient excretions by 2-3% (equal to 0.17 Tg N and 0.04 Tg P) for the four regions considered here, compared to BAU (Fig 4.2b, c, g, h).

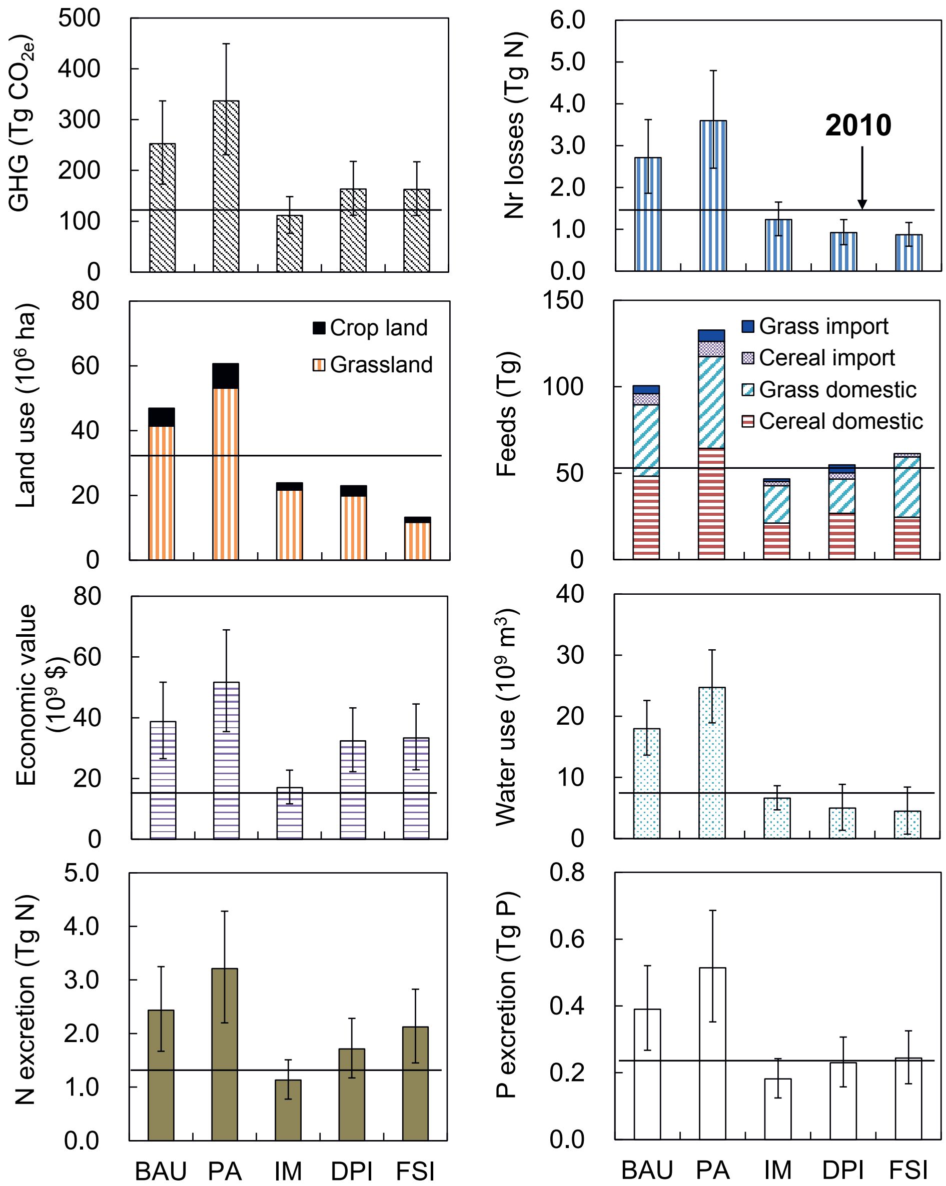

Figure 4.3. Impacts of increased milk consumption in China by 2050; results of five scenarios (BAU, PA, IM, DPI and FSI), i.e., GHG emissions, Nr losses, requirement of crop land and grassland, concentrate feed and forage imported and domestically produced, economic value, water use, N excretion and P excretion in China. The solid line represents the situation in 2010. The error bars reflect the expected lowest and highest milk

consumption in 2050.

Scenarios: BAU (SSP2): Business as usual, with a milk self-sufficiency of 75%; PA (SSP3): Produce all additional milk in 2050 domestically; IM (SSP5): Import all additional milk in 2050; DPI (SSP1a): Dairy production Improvement, on top of BAU; FSI (SSP1b): (Farming system improvement, on top of DPI.

4.3.4 Expected impacts of increased milk consumption – Scenario IM

If China would import all additional milk (IM) from EU, NZ and USA, then the global trade of milk will increase by 78 Tg/year. Milk will become a bulk trade commodity, almost comparable in size to soybean now (Fig S5). Compared to PA, the land and water use for dairy feed production would reduce by 16%–38% at the global scale, GHG emissions will decrease by 7%, and total Nr losses will reduce by 28% compared to PA (Fig 4.2a–d).

The milk imported will come from the European Union (60 Tg), NZ (9.8 Tg) and the United States (8.2 Tg). These regions will economically benefit from the milk export; the value of the additional milk exported by the EU is roughly 25 billion US$/year (Fig 4.4b). By contrast, milk import will hinder the development of the dairy industry in China, and will lead to 12 million fewer job opportunities compared with scenario PA (Fig S7). Further, it may become increasingly difficult to feed all dairy cattle in the milk exporting countries, due to the limited area of productive land, and significant competition with other land uses (food, fuel and fiber production and nature conservation). The farmland area for cattle-feed production in New Zealand would have to increase by about 57% (from 2.2 to 3.5 million ha) and that in the European Union by about 39% (from 38 to 53 million ha), and GHG emissions and Nr losses would increase roughly proportionally with the increase of farmland in both regions (Fig 4.4b,c). The European Union and New Zealand may significantly have to increase land productivity and dairy productivity, and/or increase the import of concentrate feed (Table 4.4). The results of the IM scenario suggest that GHG emissions from dairy production will increase by about 39% in the European Union, and the Nr losses will also increase by a similar proportion.

Table 4.4. Import of maize and soybean and alfalfa from United States of America (USA) and Canada (CA), Brazil (BR) and Argentina (AR), for dairy production in China (CN), European Union (EU) in 2010, and for scenarios producing all additional milk domestically (PA) and import all of the additional milk (IM) in 2050.

			2010		PA		IM	
			CN	EU	CN	EU	CN	EU
Feed, Tg yr^{-1}	USA and CA	Maize and soybean	1.0	1.2	4.2	1.2	1.2	1.7
		Alfalfa	0.9		6.2		1.3	
	BR and AR	Maize and soybean	1.0	1.9	4.3	1.9	1.2	2.5
		Alfalfa						
Land, million ha yr^{-1}	US and CA	Maize and soybean	0.31	0.42	1.3	0.42	0.36	0.57
		Alfalfa	0.17		1.2		0.27	
	BR and AR	Maize and soybean	0.33	0.60	1.4	0.60	0.39	0.82
		Alfalfa						

Note: New Zealand also imports small amounts of feed from Australia, which are not shown. PA, produce all the milk domestically in China; IM, import all the milk from leading export regions.

4.3.5 Expected impacts of increased milk consumption – Scenario DPI

In the Dairy Production Improvement (DPI) scenario, dairy-related impacts will be reduced compared to BAU, both in China (GHG emissions: 35%; land requirements: 51%; Nr losses: 34%) and for the global dairy sector examined here (GHG emissions: 12%; land requirements: 22%; Nr losses: 33%), due to the improved milk production performance in China (Fig 4.2a–c, 4.3a–c). This illustrates the huge scope for improving the dairy production efficiency, through meeting EU, NZ and US standards. However, the area of arable land in China used for feed production will have to increase significantly (+54%), and the imports of cereals (+72%) and alfalfa (+414%) will also increase greatly, compared to 2010 (Fig 4.3c,d). This indicates that improvements in the productivity and efficiency of dairy production alone may not be sufficient to relieve the pressure on land.

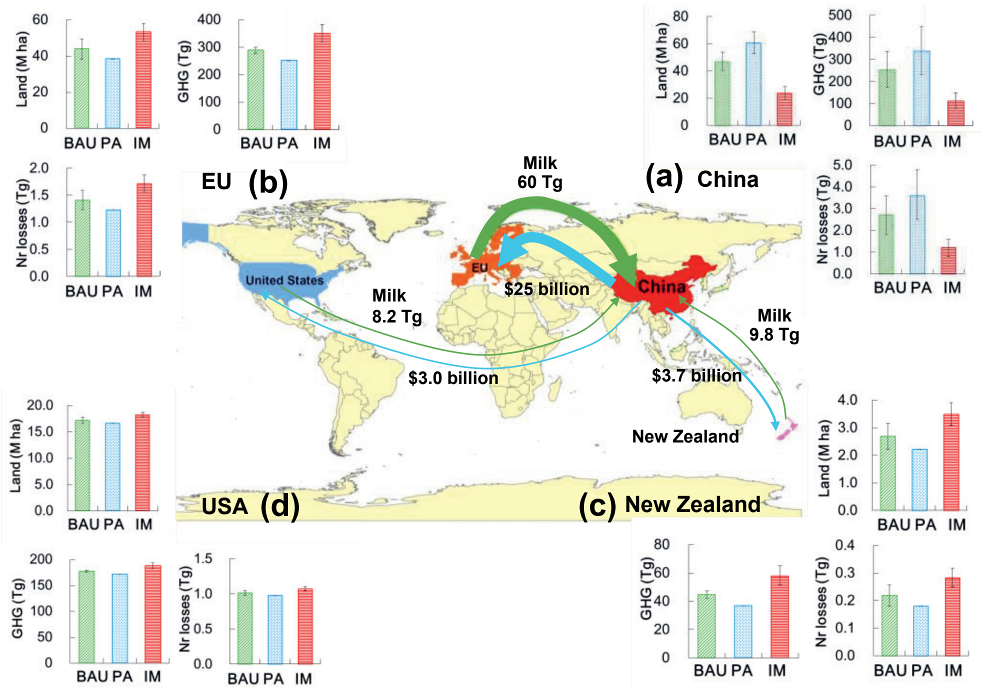

Figure 4.4. Import of milk from the world's top three milk exporters to China, and the economic return (indicated by arrows), for scenario IM in 2050. The bar graphics show the changes in agriculture land area, GHG emissions, and Nr losses in China and the three exporting countries EU, NZ and USA for the scenarios BAU, PA and IM.

Scenarios: BAU (SSP2): Business as usual, with a milk self-sufficiency of 75%; PA (SPP3): Produce all additional milk in 2050 domestically; IM (SSP5): Import all additional milk in 2050. PA represents the same production level in 2010 for EU, NZ and USA.

4.3.6 Expected impacts of increased milk consumption – Scenario FSI

The FSI scenario aims at better utilizing suitable land and closing the manure nutrient cycle, through the integration of crop – livestock production systems spatially. Scenario FSI has the potential to reduce the requirement for domestic agricultural land by 72% and the import of feed (concentrates: 4.4 Tg; forage: 4.6 Tg), compared to scenario BAU, because of the expected increases in land productivity (Fig 4.2c, Table 4.4). Meanwhile, the global GHG emissions could be reduced by 36% and Nr losses reduce by 68% (Fig 4.2a). Although the FSI scenario showed similar GHG emissions and 4%–7% higher feed demand and Nr losses compared to DPI at the global level, FSI reduced the global dairy related land use by 11% compared to DPI (Fig 4.2c). This would leave more land for arable food production and natural ecosystem services, including species rich native grasslands. However, FSI still increased GHG emissions by 19% while saving land use by 8% compared to 2010, part of these land savings will provide potential for carbon stock and compensate for the increasing GHG emissions (Fig 4.2a, c).

4.4 Discussion

The increasing demand for milk in China will have significant impacts on global dairy related GHG emissions, land use and milk, and feed trade. We show for China that producing additional milk domestically will reduce the environmental performance of global dairy production, for example, increase in GHG and Nr emissions and feed import. Importing additional milk from the leading milk exporting regions will reduce global dairy-related GHG emissions, but the environmental burden is then transferred to these countries, which may conflict with the objectives of their environmental protection policies. Improving domestic feed and dairy production efficiencies in milk-demanding countries to the level of the leading milk exporting countries seems the preferred pathway.

4.4.1 Future milk consumption

The traditional lower milk consumption countries of South and East Asia and sub-Saharan Africa are experiencing significant increases in milk consumption due to population growth and higher levels of income (Alexandratos and Bruinsma, 2012). It is projected that global milk consumption will increase by 60% between 2010 to 2050 (Alexandratos and Bruinsma, 2012), and more than 60% of the additional milk demand will come from the traditional lower milk consumption regions (<100 kg milk capita^{-1} year^{-1} in 2010), that is, East and North Africa, sub-Saharan Africa, South Asia, and East Asia, with China having the largest potential future milk demand.

We assumed that the average milk consumption in China will be 82 kg/capita in 2050, which is similar to the current level of milk consumption in Japan. Japanese and Chinese share a similar level of lactose intolerance (Mattar et al., 2012) and China's average GDP in 2050 may have caught up with Japan's 2016 level (World Bank, 2016). Yet, future milk consumption in China may be much higher, as the national guidelines for a healthy diet suggest 300 g capita^{-1} day^{-1}, which is equivalent to 110 kg capita^{-1} day^{-1} (CSN, 2014). Former Chinese Prime Minister Wen Jiabao once said he had a dream that "all Chinese, especially children, can drink a half liter of milk per day" (Xinhua News, 2006). If his dream were to be realized, the average milk consumption would be 180 kg capita^{-1} day^{-1}, still much lower the current US and EU levels (FAO, 2016). As China, has now abolished the one child policy, population may increase faster in the next few years, which may also further increase the total milk demand in the future. Evidently, the predicted mean milk consumption in 2050 has a large uncertainty range.

4

4.4.2 Domestic production or importation

Our results show that production of the additional required milk domestically without large improvements within the dairy industry will increase global dairy related GHG emissions compared to import of this milk. The average GHG emissions was 2.9 kg CO_{2eq} kg^{-1}. milk in China in 2010, compared with 2.1, 1.6 and 1.9 kg CO_{2eq} kg^{-1} milk for New Zealand, the European Union and the United States, respectively (Opio et al., 2013). The higher GHG emissions in China are due to less efficient feed and milk production. Furthermore, the GHG emissions associated with the transportation of milk are much smaller than those associated with domestic production (feed and milk), with the net effect of milk import decreasing total GHG emissions (Table 4.1). This was the same for N losses, since the average Nr loss was 34 g N kg^{-1} milk in China, which is 1.8–2.8 larger than that in the leading milk exporting regions (Table 4.1). Nitrogen losses associated with dairy production are much smaller in milk exporting countries than in China (Bai et al., 2013, 2016).

Production of all the extra milk (PA) domestically without improvement of dairy and feed production will face several domestic and international constraints. In total 5.5 million ha of domestic arable land and 28 million ha of grassland will be required additionally in the PA scenario, equal to 4.5% of arable land and 7.0% of grassland area in China, respectively (National Bureau of Statistic of China (NBSC), 2016). However, additional land area is not available domestically. Recently, the areas for arable land and grassland were slightly decreased (Fig S8). Furthermore, environmental regulations have become stricter in China, with an environmental protection tax due to be implemented at the beginning of 2018. Also a tax will be collected from high polluting dairy farms (National People's Congress of China (NPC), 2016). The PA scenario also requires import of 8.5 Tg concentrates and 6.2 Tg of alfalfa. Such high levels of import may become increasingly difficult, in part also due to pressures from the outside world. For example, the drought-stricken western United States shipped more than 0.2 billion m^3 of water embedded in alfalfa to China in 2012, which would be enough to supply annual household needs of half a million families (Culp and Robert, 2012) and soybean exports from Brazil have been linked to deforestation of the Amazon (Morton et al., 2006).

Global dairy-related GHG emissions and Nr losses will be 7% and 28% lower if all additional milk is imported compared with domestic production. However, there will be strong physical and environmental constraints in the leading milk export regions. For example, 1.3 and 15 million ha additional agricultural land would be required in New Zealand and the European Union, which is equivalent to 12% and 8% of their agricultural land in 2010, respectively (FAO, 2016). These land requirements exceed local land availabilities, so New Zealand would need to cut down the land used for sheep and beef production, or explore marginal land which is sometimes too steep or too close to watercourses for dairy production (Ministry for Primary Industries (MPI), 2012). Besides the physical limitations, environmental protection policies may also constrain large dairy production increases in the European Union and New Zealand. The results of the IM scenario suggest that Nr losses and GHG emissions from dairy production will increase by around 39% in the EU, which will obstruct environmental targets (United Nations Framework Convention on Climate Change (UNFCCC), 2015; Westhoek et al., 2014). Strong increases in milk production in NZ will also be met with resistance (MPI, 2012) The environmental constraints on drastic increases of dairy production in exporting countries suggest that changes in the balance of supply and demand will shift the global market price of dairy products to higher levels. A rise in global dairy price will make investments in domestic dairy production more attractive.

Improving domestic feed and dairy production efficiencies may be a preferred pathway for many milk-demanding countries, including China where the prospects are relatively large for improving feed and dairy production efficiency according to the DPI and FSI scenarios (Fig 4.2). This needs to be achieved not only through an increase in production and in the economic and environmental performances of China's dairy sector, to the level of leading milk export regions (DPI), but also through a redesign of the dairy production systems, to increase the contribution from grassland and to integrate dairy production systems spatially with feed production and cropland (FSI). For example, grassland covers 3/4 of the agriculture land in China. Most of this land is not suitable for intensification of feed production due to low rainfall, poor soil quality, overgrazing and desertification. However, some areas can be utilized to supply forage (1 to 3 Mg ha^{-1} $year^{-1}$) for dairy cattle when properly managed, grazed, irrigated, and fertilized (Kang et al., 2007). A further benefit of developing well-managed grazing systems is to also to contribute to grassland restoration whilst maintaining emphasis on natural ecosystem services and biodiversity in native grassland areas (Ren et al., 2016). Achieving this also requires governments, farmers, ecologists, industry, and researchers to work together to develop transition plans for different regions and farms (Eisler et al., 2014; Zhang et al., 2016). Likewise other emerging countries may face the same situation and problems of China, and may also need to improve their dairy and feed production yield, and integrate dairy and feed production together to meet their milk demand in a more sustainable manner.

4.4.3 Policy implications

Searching for the other alternatives, such as soybean milk, can be other possible options to alleviate China's high milk demand, and impacts on global sustainability. Strategies for improving feed production, dairy production, and manure management have to be embedded in coherent governmental policies with proper incentives. The Chinese government is already supporting dairy production via providing subsidies for the construction of industrial feedlots. For example, for the construction of a dairy farm with 300–1,000 dairy cattle a lump sum subsidy of 0.8–1.7 million RMB is available (300–400 US$ per dairy cow; MOA, 2014). Investments in manure management and forage production are also supported by government but less compared to dairy production. There is a need for a more coherent government policy for developing an efficient and sustainable dairy sector. Governmental support for the dairy sector has to be embedded in policies aimed at improving both the production and environmental performance. These policies should include clear regulations on manure management to ensure that all manure from housed animals is properly collected, stored, and subsequently applied to arable land and grassland, instead of being discharged to landfill or water systems as has happened for the past 60 years for in pig production industry, which have greatly decreased N use efficiency at the system level and increased manure losses to water in China (Bai et al., 2014; Strokal et al., 2016).

The Chinese government recently introduced new legislation, and has set goals to establish a waste recycling system for livestock enterprises through scientifically evidenced regulation and a clear responsibility for producers to minimize nutrient losses (State Council of China (SCC), 2017). The central government also invests 0.3 billion each year to subsidize farmers growing alfalfa.

Recently, milk processing factories banned the collection of milk from small household dairy farms, mainly due to concerns about milk quality. It has been estimated that some 100,000 small dairy farmers have stopped farming each year since 2010 (MOA, 2015). This contributes to a redesign of dairy production in China, through conversion of traditional dairy production systems to medium sized household systems. Currently, some of China's dairy companies invest overseas rather than in domestic production, due to eroded public confidence in the quality of domestic milk, low production efficiency, and high production cost (Sharma and Rou, 2014). Hence, it is of great importance to regain the consumers and investors' confidence in the Chinese milk sector, through implementing strict milk quality control and fine policies, such as the Food Security Law issued in 2015 (NPC, 2015).

Overall, the ever-growing thirst for milk in China comes with significant challenges, and impacts on global trade of milk and feed, land use, GHG emissions and Nr losses. In 2050, producing all additional required milk domestically with current technologies and management will require annual imports of 8.5 Tg concentrates and 6.2 Tg forages, and will increase GHG emissions of the global dairy sector by 41% and land demand by 40% compared to 2010. In contrast, importing all additional milk will transfer the environmental burden from China to milk exporting countries (e.g. EU, NZ and the US). The optimal option is to produce the additionally required milk in China, but with greatly improved technology. The prospects and challenges of improving the local dairy production efficiency, manure, and grassland management, and of the integration of crop–dairy production systems are large. Closing the productivity gaps in domestic dairy and feed production, accompanied by dairy production system adjustment, greater utilization of grassland resources along with feed ration improvement and strict milk quality control systems appears to be the preferred pathway. This pathway should be guided through governmental policies, mainly focused on improving manure management, feed production, crop-livestock system integration, and grassland restoration whilst maintaining emphasis on natural ecosystem services, and biodiversity in native grassland areas.

4.5 Acknowledgements

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CHAPTER 5

Nitrogen, phosphorus, and potassium flows through the manure management chain in China

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Abstract

The largest livestock production and greatest fertilizer use in the world occurs in China. However, quantification of the nutrient flows through the manure management chain and their interactions with management-related measures is lacking. Herein, we present a detailed analysis of the nutrient flows and losses in the "feed intake−excretion−housing−storage−treatment−application" manure chain, while considering differences among livestock production systems. We estimated the environmental loss from the manure chain in 2010 to be up to 78% of the excreted nitrogen and over 50% of the excreted phosphorus and potassium. The greatest losses occurred from housing and storage stages through NH_3 emissions (39% of total nitrogen losses) and direct discharge of manure into water bodies or landfill (30−73% of total nutrient losses). There are large differences among animal production systems, where the landless system has the lowest manure recycling. Scenario analyses for the year 2020 suggest that significant reductions of fertilizer use (27−100%) and nutrient losses (27−56%) can be achieved through a combination of prohibiting manure discharge, improving manure collection and storages infrastructures, and improving manure application to cropland. We recommend that current policies and subsidies targeted at the fertilizer industry should shift to reduce the costs of manure storage, transport, and application.

5.1 Introduction

Intensive livestock production systems have large impacts on water and air quality through emissions of greenhouse gas (GHG) and nutrients [mainly nitrogen (N) and phosphorus (P)] (Steinfeld et al., 2006; Bouwman et al., 2013; Gerber et al., 2013; Sutton et al., 2013). The N and P emissions originate mainly from livestock excrements. Total livestock excretion in the world is about 80−130 Tg of N per year, of which only 20−40% is efficiently utilized for fertilizing cropland (Sheldrick et al., 2003; Oenema et al., 2005). The remainder of the manure N excreted is emitted to the atmosphere, groundwater, or surface waters. However, there are significant differences in manure management systems throughout the world, depending on livestock production system, environmental conditions, and governmental policy measures. For example, on average, 65% of excreted N in animal housing is recycled back to agricultural land in the European Union, which is, in part, a result of strict regulations (Oenema et al., 2007). At the farm level, there can be large differences. A survey in Africa showed that 6−99% of collected manure is recycled to cropland (Rufino et al., 2006). In China, a large proportion of manure N from pig production is lost through direct discharge into water bodies or landfilling (Bai et al., 2013; 2014).

Recent studies have used material flow and nutrient footprint approaches to quantify N and/or P losses and to use efficiencies for whole livestock production systems at the global level (Sutton et al., 2011), regional level (European Union) (Leip et al., 2011; 2014), and national level (United States) (EPA et al., 2013). Several models have been developed to estimate the nutrient flows and losses in livestock production, for example, MITERRA-Europe (Velthof et al., 2009), NUFER (NUtrient Flows in Food Chains, Environment and Resources Use) (Ma et al., 2010) and GAINS (Greenhouse Gas−Air Pollution Interactions and Synergies) (Klimont and Brink, 2004). Some other models have been developed to estimate single-pollutant losses from livestock systems, such as ammonia (NH_3) emissions from manure management (Reidy et al., 2008). However, no models are yet available that can calculate N, P, and potassium (K) flows and losses in detail for each step in the “feed intake−excretion−housing−manure storage−manure treatment−manure application” chain in a consistent way for the different animal categories and production systems at the regional level. A further understanding of the manure nutrient flows and losses is important, because previous studies showed that the effects of feeding regimes and manure management practices strongly differ among livestock production systems (Bai et al., 2013; Herrero and Thornton, 2013). For example, N losses from traditional dairy production systems in China mainly occur through NH_3 emissions, whereas in industrial dairy feedlots, they occur mainly through direct discharge of manure to water bodies or landfill (Bai et al., 2013). Information about the effects of management-related technical measures and their interactions on manure nutrient recycling and subsequent chemical fertilizer needs is still lacking. Such insights are needed to achieve low-emissions livestock production systems and sustainable agriculture in China (MOA, 2015).

China is a major contributor to world livestock production, and both extensive and intensive systems exist (MOA, 2013; FAO, 2015). Industrial-scale livestock operations are rapidly increasing in their contribution to total livestock production, mainly because of their large production capacity and high feed use efficiency on the farm. However, most of these industrial systems are landless and have limited opportunities for recycling manure nutrients back to cropland. Recently, the Chinese government initiated a plan to stabilize fertilizer consumption by 2020, the so-called “Zero Fertilizer Growth by 2020” goal. The consumption of K fertilizers is large and there have been no studies in which the potential of replacing fertilizer K by manure K has been assessed (FAO, 2015).

The aim of this study was to estimate the manure N, P, and K flows and losses in the feed intake−excretion−housing−storage−treatment−application chain for different animal categories and production systems in China for the year 2010 using a modified version of the NUFER model. We developed a K module and included it in the NUFER model. The analyses were carried out for different animal categories (pigs, layers, broilers, dairy cattle, beef cattle, buffaloes and draft cattle, and sheep and goats) and different production systems (mixed, grazing, and landless systems). In addition, five scenarios were explored to assess the potential for reducing manure N, P, and K losses and for replacing fertilizer inputs by manure nutrients by the year 2020.

5.2 Material and methods

The NUFER model was further developed and used to estimate N, P and K flows and losses in each step of the manure management chain (Ma et al., 2010). This model uses a mass balance approach; it starts with an estimation of total feed nutrients intake rate for the different animal categories. The calculation methods and parameters used are presented in detail in the Supplementary Information (SI) (https://pubs.acs.org/doi/suppl/10.1021/acs.est.6b03348/suppl_file/es6b03348_si_001.pdf). Below a summary is given.

5.2.1 Description of livestock production systems

In total, six animal categories and three typical (for China) production systems were distinguished. The six main animal categories include pigs, layers, broilers, dairy cattle, other cattle (beef cattle, buffaloes and draught cattle), and sheep and goats. Together these categories generate most of the manure in China (Chadwick et al., 2015). For each animal category we distinguished three different production systems, according to the feeding regimes, manure management practices and available statistical data, i.e. mixed cropping-livestock system, grazing production system and landless production system (Table S1). Mixed cropping-livestock systems are basically the traditional production system; the solid part of excretion is collected and mainly applied to cereal crops, while the liquid fraction is only partly collected and the remainder is lost by leaching into the subsoil and wider environment. Grazing systems are mainly found in Gansu, Xinjiang, Ningxia, Tibet, and Inner-Mongolia provinces (Ma et al., 2010), see also Fig S1. Most of the excretion is directly dropped to grassland, during the grazing period. The solid part of the excretion is collected when the animals are kept in confinement; however, the liquid part is mainly leached to the subsoil. The industrial production systems are landless; a large fraction of the manure produced in landless systems is discharged into surface waters, with or without some treatment, or dumped into landfills. A part of the solid manure is exported to nearby farms growing vegetables and fruits following composting treatment. The livestock production structure (i.e. the percentage of each system per animal category) in 2010 was derived from national statistics (MOA, 2013). Information about the definitions and animal population of each production system is listed in Tables S1 and S2 in the SI.

5.2.2 Feed and nutrient intake calculation

Total feed intake was calculated from the number of livestock for each category and the feed requirements per animal category. Feed intake was estimated on the basis of the energy requirements for maintenance, growth (live weight gain) and production (Bai et al. 2013; 2014). The number of animals per category was derived from MOA statistics and the FAO database. For pigs and broilers, the number of slaughtered animals was used, and for dairy and layers, the numbers of producing animals (FAO, 2015). Stock numbers were used for beef cattle, buffaloes and draught cattle, and sheep and goat production (FAO, 2015). Feed intake was estimated from energy requirements per animal category and feed supply according to data from the FAO database and farm surveys (Ma et al., 2010; FAO, 2015). Feed-specific N, P and K contents were derived from literature (Table S9). The excretion of nutrients was calculated as the difference between feed nutrient intake and the nutrient retained in products (milk, eggs) and in body weight gain (meat, blood, bones and hides).

5.2.3 Nutrient retention by livestock

Nutrient (N, P and K) retention was calculated at the herd level per animal category (considering the breeding and backup animals). In a simple form (for one animal category, production system), the equation for estimating retention showed as below:

$Oa_{\text{nutrient in products}}$ = (Yield + BWG) * Animal number * Nutrient content [1]

Where, $Oa_{\text{nutrient in products}}$ is the total amount of N, P or K in animal body weight gain, milk and egg, in kg per year; Yield is the yield of animal products (milk and eggs), in kg head^{-1} yr^{-1}; BWG is the body weight gain of animals, including meat, bones, blood and hides, in kg head^{-1} yr^{-1} (Table S5-7); Nutrient content is the N, P or K content of milk, eggs and BWG, in kg kg^{-1}. The N and P contents were derived from the NUFER database (Ma et al., 2010). The K content was derived from literature (Table S8).

5.2.4 Nutrient excretion

Nutrient excretion was calculated as the difference between total nutrient intake and nutrient retention in milk, egg and BWG. In a simple form:

$Oa_{\text{nutrient excretion}} = Ia_{\text{nutrient intake}} - Oa_{\text{nutrient in products}} - Oa_{\text{nutrient in dead animals}}$ [2]

Where, $Oa_{\text{nutrient excretion}}$ is the amount of nutrient (N, P and K) excreted per animal category, in kg yr^{-1}; $Oa_{\text{nutrient in products}}$ is the amount of N, P and K output in milk, eggs and BWG per animal category and production systems, respectively, in kg yr^{-1}; and $Ia_{\text{nutrient intake}}$ represents the amount of N, P and K in feed intake per animal category and production system, in kg yr^{-1}. Corrections were made for animals that died during the production cycle ($Oa_{\text{nutrient in dead animals}}$); it was assumed that the nutrients in dead animals ended up landfill, possibly following incineration.

5.2.5 Nutrient use efficiency and manure nutrient recycling efficiency

The N, P and K use efficiencies and manure nutrient recycling efficiency were calculated at herd level as follows:

Nutrient use efficiency = ($Oa_{\text{nutrient in products}}$ / $Ia_{\text{nutrient intake}}$) * 100% [3]

Manure nutrient recycling efficiency = ($Oa_{\text{nutrient recycled}}$ / $Oa_{\text{nutrient excretion}}$) * 100% [4]

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Manure nutrient recycling efficiency is the percentage of excreted N, P and K recycled to agricultural land per animal category and production system. $Oa_{nutrient\ recycled}$ is sum of the amounts of N, P and K deposited during grazing and applied to agricultural land per animal category and production system, in kg yr^{-1}.

5.2.6 Fertilizer replacement by manure

The potential to replace fertilizer by manure was calculated as follows:

Fertilizer replacement = {(Manure application * Fertilizer value) / Fertilizer application}* 100% [5]

Where, Fertilizer replacement is the amount of fertilizer N, P and K that can be replaced by manure N, P and K, in %; Manure application is the amount of manure N, P and K applied to cropland (excluding manure N, P and K deposited during grazing because essentially no fertilizer is applied to grassland in China), in kg yr^{-1}; Fertilizer value is the proportion of the manure N, P and K available to crops in the first season, in % (Table S13); Fertilizer application is the amount of fertilizer N, P and K applied to cropland in 2020, in kg yr^{-1}. Fertilizer values differ per animal manure type and nutrient. The total fertilizer application in 2020 was estimated from the total N, P and K fertilizer application in 2015 (FAO, 2015), and the expected increase in fertilizer use during the period 2015- 2020 (MOA, 2015).

5.2.7 Scenarios for 2020

A total of five future scenarios were considered to explore the potential for reducing nutrient losses from the manure management chain and the potential for replacing fertilizer nutrients by manure nutrients, The year 2010 was used as the reference year because of data availability, and 2020 was set as a target year, because of China's aim to achieve "Zero Fertilizer Increase Use" by the end of 2020 (MOA, 2015). According to the projections by FAO, the total demand for animal products in China will increase by 17% for pork and eggs, 33% for chicken meat, 24% for milk, 24% for beef, and 22% for mutton between 2010 and 2020 (Alexandratos and

Bruinsma, 2012).

S0 – Business as usual (BAU). Production of animal products was based on linear extrapolations of the projections of FAO between 2010 and 2020 (Alexandratos and Bruinsma, 2012). In this scenario, we assumed that the increase in the demand for livestock products would be produced in land-less systems. We also assumed that the productivity, feed composition and manure management practices of each system would be the same as in 2010, which may provide a conservative estimate of manure nutrient production, flows and losses. Changes in animal numbers between 2010 and 2020 are shown in Table S2.

S1 – Prohibit the discharge of manure. This scenario builds on S0, but includes an adoption of stricter manure management regulations (SCC, 2013). We assumed that the discharge of manure to surface water or landfill will be prohibited in the industrial animal production systems.

S2 – Improving manure collection and nutrient preservation in housing and storage systems. This scenario builds on S0, and assumes in addition that the N losses (in % of excreted N) from animal housing and storage systems will decrease to current mean levels in the European Union (EU), i.e., the average N losses via NH_3, N_2O and N_2 emissions to air and via N leaching will decrease to 5%, 1%, 5% and 0% of the amount of manure N excreted, respectively (Velthof et al., 2009). Further, there will be no losses of manure P and K from industrial production system due to improved containment of manures.

S3 – Improving manure application. This scenario builds also on S0. Current crop production in China does not account for nutrients supplied by manure, especially in cash crop production (e.g. in greenhouse vegetables and orchards) in which the amounts of nutrient applied with fertilizer and manure far exceed the nutrient demand of the crop (Yan et al., 2013). We assume that the available nutrients in manures applied to cropland replace fertilizer nutrients, using manure-specific fertilizer nutrient replacement values. Further, we assumed that low ammonia emission manure application methods will be adopted, i.e. slurry injection and rapid incorporation of solid manures, by which the N fertilizer replacement value of manures will increase to 55% for cattle slurry, 75% for pig slurry and 85% for poultry manure (Table S13).

S4 – Combination of S1-S3. In this scenario, the technologies assumed for S1, S2 and S3 are combined. Further, we assumed that the technologies can be applied as complements to each other.

5.3 Results

5.3.1 Nutrient flows through the whole manure chain

The N, P, and K flows in the manure chain for the year 2010 are presented in Fig 5.1. Total feed N intake amounted to 26.0 Tg, of which 22.8 Tg of N was excreted (3.5 Tg of N was deposited in the field during grazing, and 19.3 Tg of N was excreted in housing systems). Total N losses through NH_3 emissions, denitrification, and leaching from housing amounted to 8.3 Tg. In addition, a significant amount (5.4 Tg of N) was lost through discharge to water bodies or landfill. The collected manure (5.6 Tg of N) was treated (by composting, digestion, or separation), which led to another N loss of 1.6 Tg. In total, only 4.0 Tg of manure N was applied to cropland, specifically, 68% to cash crops and 32% to cereal crops.

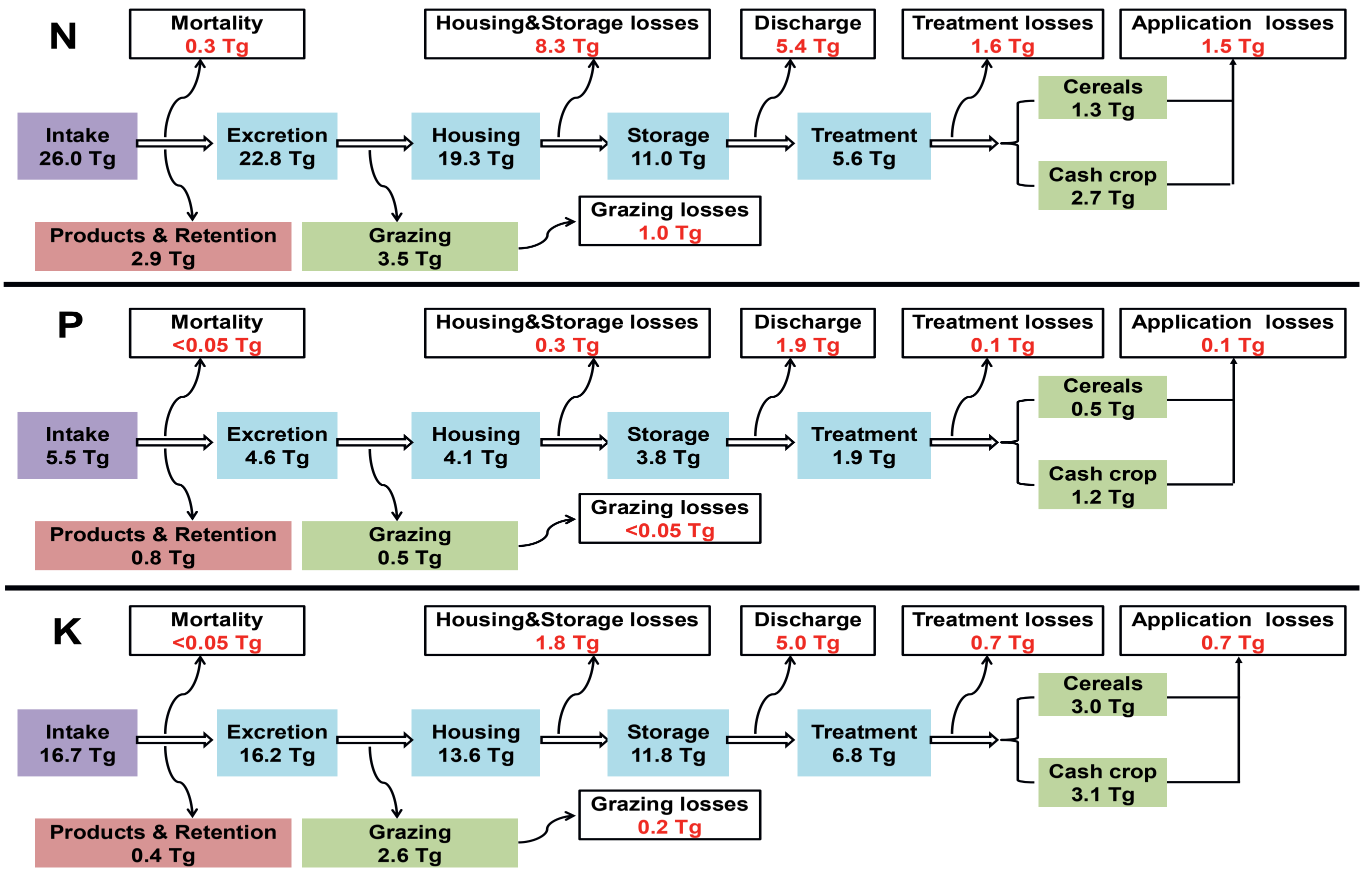

Figure 5.1. The nitrogen (N), phosphorus (P) and potassium (K) flows through the manure management chain in China in 2010. ***Note: Discharge, direct discharge of manure to water body or landfill.***

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Total N losses through gaseous emissions, leaching, runoff, and erosion during grazing and after manure application were estimated at 1.5 Tg. The remaining manure N was either taken up by crops or accumulated in the soil. In total, 78% (17.8 Tg of N, including field losses) of the excreted N was lost to the environment from the different stages of the manure management chain. The highest losses occurred from housing and manure storage (47% of total N loss), followed by discharge of manure to water bodies or landfill (30%), losses during treatment (9%), and losses following manure application (8%) and grazing (6%).

About 50% of the total amount of excreted P (4.6 Tg) and 53% of the total amount of excreted K (16.2 Tg) were recycled in pastures and cropping systems. As in the case of N, cash crops received the highest amount of manure P. However, the applied manure K was more evenly distributed among grassland, cereals, and cash crops, as most of the K came from other cattle and sheep and goat production (Fig 5.2 and 5.3).

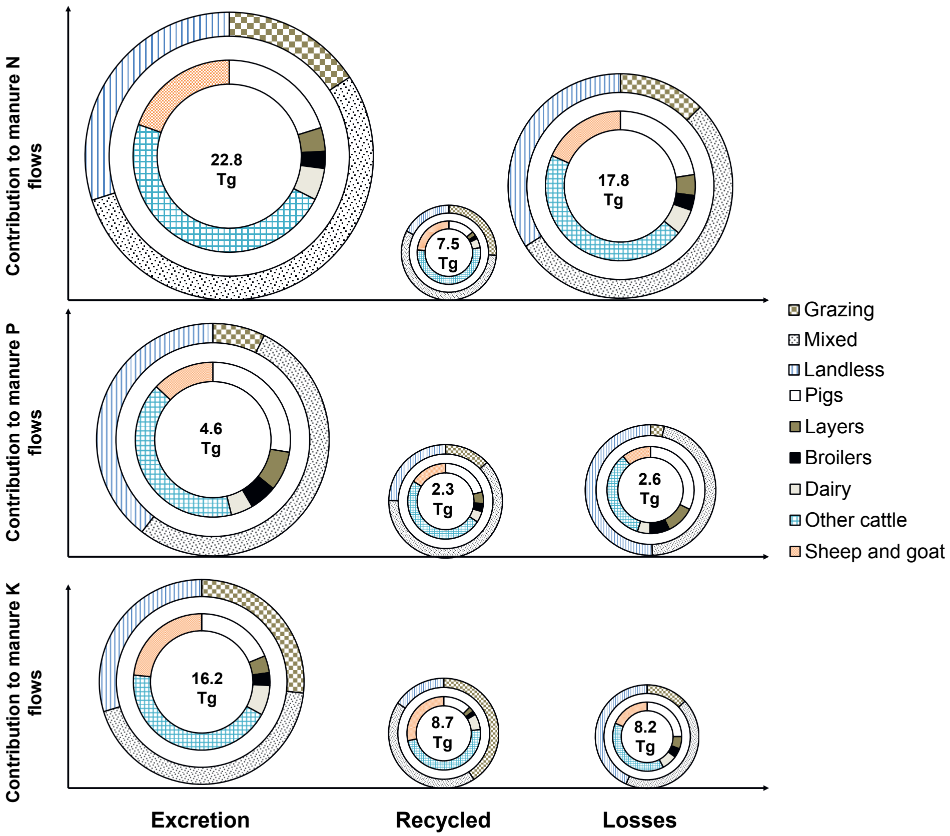

Figure 5.2. The relative contribution of different animal production systems (outside circle) and animal categories (inner circle) to the total manure nitrogen (N), phosphorus (P) and potassium (K) excretion, recycling to agriculture land and losses to the environment from livestock production in China in 2010.

5.3.2 Contributions of different animal categories and production systems

Fig 5.2 presents the relative contributions of different production systems and animal categories to total manure nutrient flows and losses. Clearly, the manure nutrient flows were largest in the mixed production systems, accounting for 53−57% of total nutrient excretion. Nutrient excretion was smallest in grazing systems. Nutrient recycling was smallest in the landless production system and largest in grazing systems. The P losses were relatively low in grazing systems (3% of total manure P losses) and high in mixed and landless production systems. Manure production was highest for the category beef cattle, buffaloes, and draft cattle, followed by sheep and goats and then pigs. Pigs excreted two times more P than sheep and goats. The contributions of layers, broilers, and dairy cattle to manure nutrient flows were relatively small (Fig 5.2).

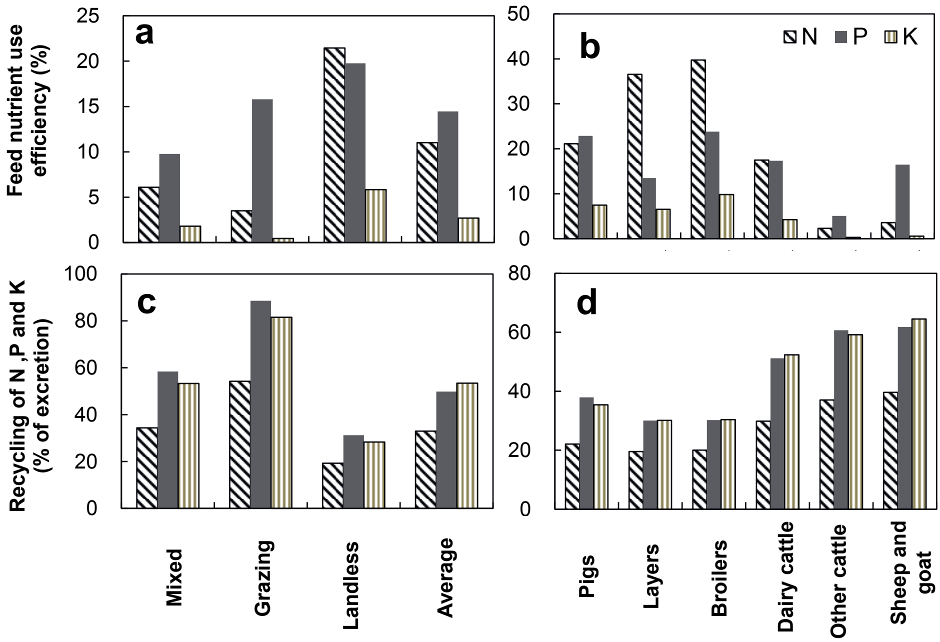

Figure 5.3. The nitrogen (N), phosphorus (P) and potassium (K) use efficiencies at the herd level for different production systems (a), and animal categories (b), recycling rate of excreted N, P and K for different production systems (c), and animal categories (d).

5.3.3 Nutrient use efficiencies at the herd level

The average feed nutrient use efficiency (UE) at the herd level, across all livestock categories, was 11% for N (NUE), 14% for P (PUE), and 2.7% for K (KUE). The low KUE was related to the low K contents of meat, milk, and eggs. The landless production systems had much higher nutrient use efficiencies than the other two production systems (Fig 5.3a). For example, the NUE of landless systems was 21%, almost six times that of grazing systems. The mixed production system was more efficient than the grazing system in converting feed N and K into products (Fig 5.3a), but the grazing system had a higher PUE than the mixed system.

The NUE at the herd level was highest for broilers (40%) and lowest for beef cattle, buffaloes, and draft cattle (2.3%). Similar results were found for PUE and KUE. The monogastric animals (broilers, layers, and pigs) were more efficient than the ruminant livestock categories (dairy cattle, beef cattle, buffaloes and draft cattle, sheep and goats) in utilizing feed nutrients (Fig 5.3b).

5.3.4 Manure nutrient recycling efficiency

The manure N recycling efficiency decreased in the order grazing (54%) >mixed (34%) > landless (19%) production system. Similar trends were found for the manure P and K recycling efficiencies (Fig 5.3c,d). The manure nutrient recycling efficiency of grazing systems was about twice that of the landless system (Fig 5.3c). The manure nutrient recycling efficiency ranged from 20% to 40% for N, from 30% to 62% for P, and from 30% to 65% for K, depending on animal category and production system. The manure nutrient recycling was higher for ruminant animals than for monogastric animals (Fig 5.3).

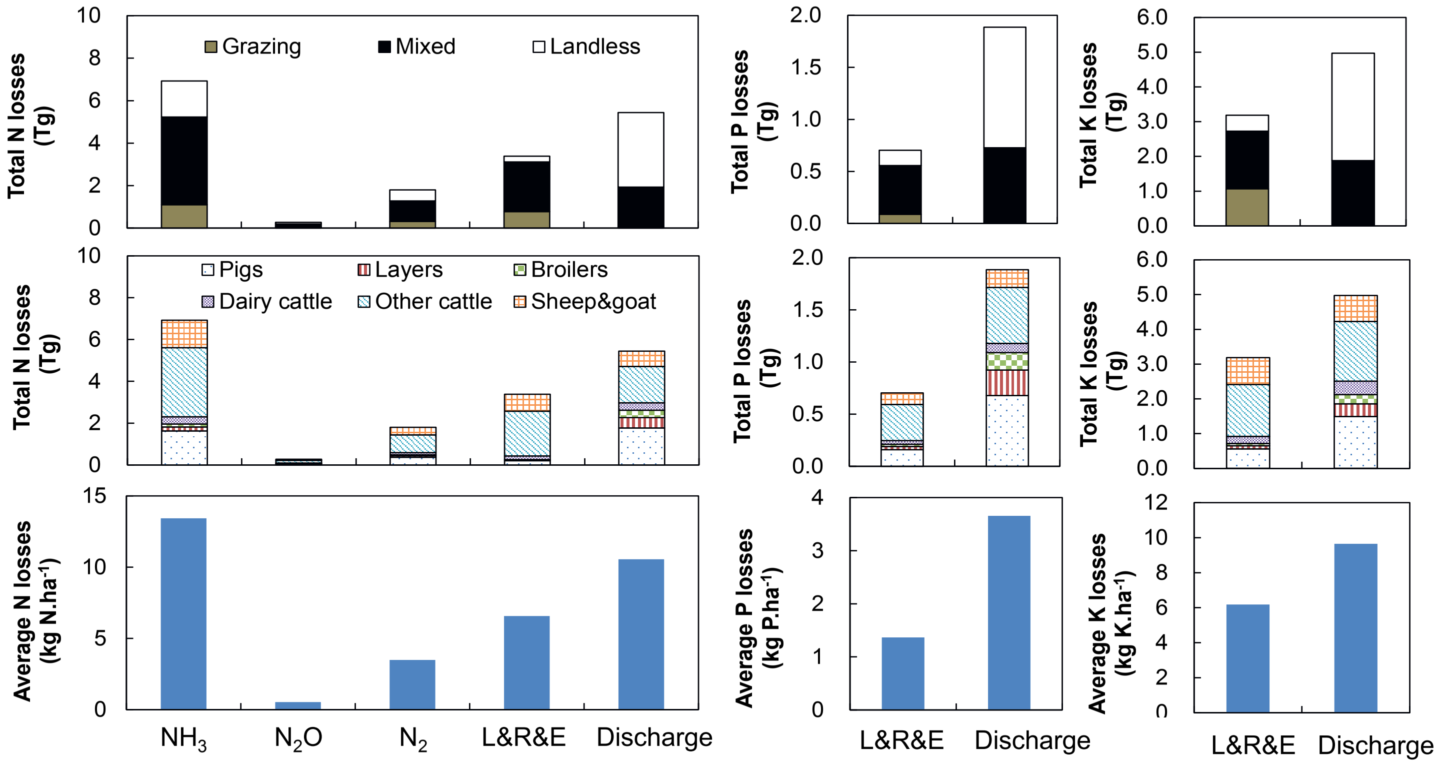

Figure 5.4. The contribution of the different animal categories and production systems to the total nitrogen (N), phosphorus (P) and potassium (K) losses through the manure management chain in China in 2010, and different scenarios in 2020.

L&R&E are losses by, leaching, runoff and erosion.

5.3.5 Losses of nutrients from the manure management chain

Total N losses from the whole manure management chain amounted to 17.8 Tg in 2010 (Fig 5.4). Emissions of NH_3 [6.9 Tg, 13 kg of N (ha of agricultural land)$^{-1}$] and discharge of manure to surface waters or landfill [5.4 Tg, 11 kg of N (ha of agricultural land)$^{-1}$] were the major N-loss pathways. Losses through leaching, runoff, and erosion (L&R&E) amounted to 3.4 Tg of N [6.6 kg of N (ha of agricultural land)$^{-1}$]. Losses through denitrification and N_2O emissions were 1.8 Tg [3.5 kg of N (ha of agricultural land)$^{-1}$] and 0.3 Tg [0.5 kg of N (ha of agricultural land)$^{-1}$], respectively. Direct discharge of manure to water bodies or landfill contributed 78% to the total manure P losses and 61% to the total manure K losses. The total average N loss was 35 kg of N (ha of cropland)$^{-1}$. Average P and K losses were 5.0 kg of P (ha of cropland)$^{-1}$ and 16 kg of K (ha of cropland)$^{-1}$, respectively.

Animal production systems differed in nutrient-loss pathways (Fig 5.4). Most of the N losses through NH_3, N_2O, and N_2 emissions to air and through leaching, runoff, and erosion to water bodies occurred in mixed production systems. Direct discharge of manure into watercourses or landfill represented the largest N-loss pathways in landless production system, whereas there was no discharge of manure nutrient in grazing systems. Ammonia emissions were the largest N-loss pathway in mixed systems. The highest P losses were found in pig and poultry production and in landless production systems. Monogastric animals contributed about 50% of the N losses through direct discharge, whereas ruminant animals were the dominant source of N losses through leaching, runoff, and erosion.

5.3.6 Scenarios for 2020

In the S0 business-as-usual scenario, manure management contributed to reductions of 1% of N fertilizer, 12% of P fertilizer, and 34% of K fertilizer (Fig 5.5). The changes in fertilizer replacement were found to be relatively small in the scenario in which direct discharge of manure was prohibited (S1). This is because most of the recycled manure was applied to cash crops;

these crops are overfertilized, and more manure does not affect the fertilizer use. Also, a ban on direct discharge would not be not easy to implement, as it would require additional investments in manure storage, transportation, and spreading infrastructure, thus increasing the cost. Improving manure collection in housing and storage systems (S2) also was found to have little impact on fertilizer replacement rates, for similar reasons (Fig S2). Improving manure application strategies (S3) seems more promising than the other two single options; it was found to increase the N fertilizer replacement to 11%, the P fertilizer replacement to 43%, and the K fertilizer replacement to 76%.

In scenario S0, the total N losses from the whole manure management chain was found to increase by 15% in 2020 compared with 2010. The total P and K losses were found to increase by 12% and 22%, respectively (Fig 5.5b). Among all of the single options, scenario S1 was found to be more effective than scenarios S2 and S3 in reducing nutrient losses. The N losses could be reduced by 14%, P losses by 47%, and K losses by 40% through the prohibition of direct discharges of manure to surface water or landfill and the application of manure to agricultural land. The integrated option (scenario S4) was found to be more effective than the single options, both in reducing fertilizer input and in reducing manure nutrient losses. For the combination scenario (S4), it was found that 27% of the chemical N fertilizer, 86% of the chemical P fertilizer, and all of the chemical K fertilizer could be replaced by optimizing the management of manure. At the same time, the manure N losses could be reduced by 27%, P losses by 56%, and K losses by 53%, compared with those in scenario S0 (Fig 5.5).

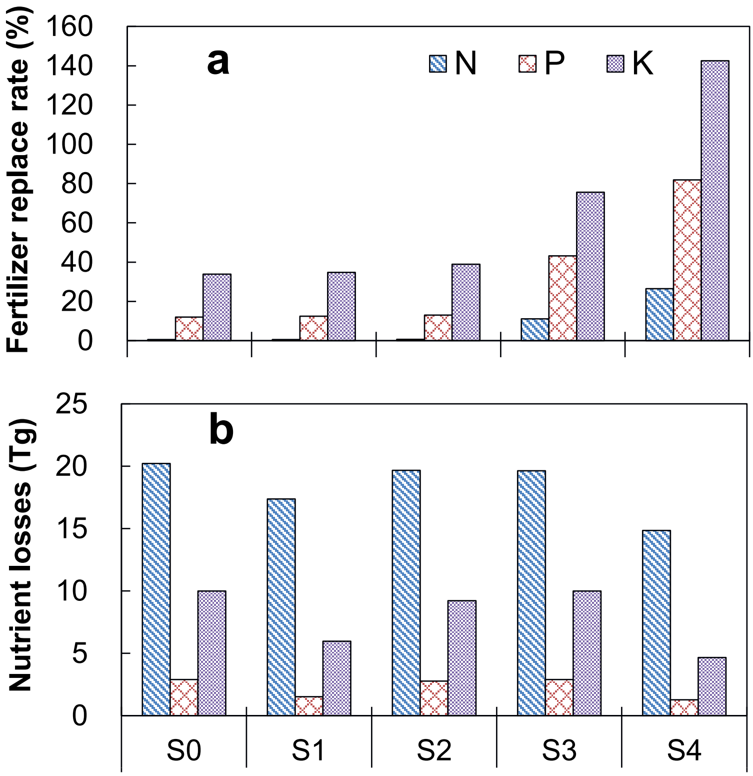

Figure 5.5. The chemical nitrogen (N), phosphorus (P), and potassium (K) fertilizer replacement by manure (a), and N, P and K losses in the manure management chain (b) in different scenarios in 2020.

S0, business as usual in 2020; S1, promote application of manure and prohibit discharge of manure; S2, improving manure collection in the housing and storage; S3, improving manure application; S4, Combination of S1-S3.

5.4 Discussion

This is the first study on N, P, and K flows through the whole manure management chain of different livestock categories and production systems in China that quantifies the impacts of management and technical measures to reduce N, P, and K losses. In 2010, 33% of the excreted N, 50% of the excreted P, and 53% of the excreted K were recycled, with the rest being lost to the environment at different stages of the manure chain. In contrast to the situations in many other countries in the world, the greatest losses occurred through direct discharge of manure to watercourses and landfill. The highest nutrient losses occurred in mixed production systems with other cattle (beef and draft cattle and buffaloes). The results of this study indicate the key stages (animal housing and manure storages), production systems (landless systems), and livestock categories (other cattle and pigs) on which policy and research on manure management should be focused in China in the future.

Through analysis of the impacts of management-related technical measures, we found that significant reductions of nutrient losses (27−56%) and inputs of fertilizers (27−100%) could be achieved by 2020. To achieve such reductions, polices for prohibiting direct discharge of manures into watercourses and landfill, improving manure collection and storage infrastructure, and improving manure application methods should be coordinated in the future. The main uncertainties in the results of this study are related to the robustness of the statistical and literature data and the emission factors used. Additional farm-level monitoring and measurements of nutrient losses are needed in the future to more accurately estimate the nutrient flows (e.g., gaseous emissions and leaching of nutrients from different livestock systems) and fertilizer replacement values of different manures.

5.4.1 Manure nutrient excretion, losses, and recycling in China

Our estimate of total N excretion in 2010 (22.8 Tg) was higher than the net N excretion of some other studies, for example, 16 Tg of N in 2010 (Chadwick et al., 2015), 17 Tg of N in 2005 (Liu et al., 2010), and 19 Tg of N in 2005 (Ma et al., 2010). These other studies presented the net excretion only; that is, they corrected the total excretion for gaseous N losses in housing and storages (Liu et al., 2010; Ma et al., 2010; Chadwick et al., 2015). The average gross N and P excretions per animal category were rather similar to the gross nutrient excretion rates reported for the European Union (Velthof et al., 2009) (Table S17). Total excretion values were 4.6 Tg for P and 16.2 Tg for K in 2010, which are rather similar to those of previous studies, namely, 5.2 Tg of P in 2010 (Chadwick et al., 2015), 4.4 Tg of P in 2005 (Ma et al., 2010), and 14 Tg of K in 2005 (Liu et al., 2010).

The estimated NH_3 emissions (6.9 Tg of NH_3-N) and N_2O emissions (0.3 Tg of N_2O-N) were higher than the estimates of other studies (5.3 Tg of NH_3-N in 2006 and 0.2 Tg of N_2O-N in 2007) (Chen et al., 2010; Huang et al., 2012). These differences are partly because net N excretion (excluding gaseous N losses from housing) was used in one of the previous studies and partly because N losses following manure application were not included in the other study (Chadwick et al., 2015). In our study, 7.5 Tg of manure N (representing 33% of excreted N) was either deposited in grassland by grazing livestock or applied to cropland (Fig 5.1). This amount is similar to the estimate provided in the study of Ma et al (2010), who reported that 32% of the excreted N was recycled in 2005, but lower than that of Liu et al (2010), who reported that more than 50% of the excreted N was recycled in 2002. The latter study did not consider N losses in all stages of the manure management chain and, therefore, overestimated the amount of manure recycled. About 50% of the excreted P was found to be recycled, which is similar (45%) to the value reported by Liu et al (2016). Our results suggest that about 8.7 Tg of K was recycled, out of the 16 Tg of K excreted in 2010. Nearly similar amounts of excreted K were found to be lost through direct discharge of manure to water bodies or landfill (61%) and leaching to the subsoil (39%) (Fig 5.4). The leaching coefficients for K were derived from composted manure (Table S12) and applied to the housing, storage, and treatment sector for the other manure types of manure because of a lack of data. As composted manure contains relatively lower amounts of K in solution, our estimation for K leaching might be relatively low.

Table 5.1. Manure nitrogen (N) and phosphorus (P) excretion, utilization in crop land, and losses in United States of America (USA), Europe (EU), Japan and China.

Year	USA		EU		Japan		China**	
	2002[1]	2007[2]	2000[3]	2000[4]	2005[5]	2005[6]	2010	2010'
	N	P	N	P	N	P	N	P
Excretion (Tg)	6.8	1.7	10.4	4.2	0.73	0.19	22.8	4.6
Utilization (Tg)	5.0	1.3	8.4	4.2	0.51	0.15	7.5	2.3
Losses (Tg)*	1.8	0.4	2.0	0.01	0.23	0.04	15.3	2.3

1. (EPA, 2011); 2. Suh and Yee, 2011; 3. Leip et al., 2011; 4. Ott and Rechberger, 2012; 5. Mishim, 2002; 6. Mishim et al., 2010. *Not including losses after manure application to crop land. **This study.

5.4.2 Comparison with manure management in other countries

About 33% of the excreted N and 50% of the excreted P were utilized in China in 2010 (Fig 5.1). Table 5.1 shows that these percentages are smaller than those reported for the United States (75% of the manure N and P is recycled) (EPA, 2011; Suh and Yee, 2011) the European Union (80% of the manure N and almost 100% of the manure P is recycled) (Leip et al., 2011; Ott and Rechberger, 2012), and Japan (70% of the manure N and 80% of the manure P are recycled) (Mishima, 2002; Mishima et al., 2010). The differences between China and these other countries are mainly related to environmental regulations. Manure storage and application to agricultural land is regulated in the European Union (e.g., Nitrates Directive, Water Framework Directive, and National Emission Ceiling Directive) (Oenema et al., 2007). The Nitrates Directive regulates the use of N in agriculture, especially through the designation of "nitrate vulnerable zones" and the establishment of Action Plans in these areas (e.g., the maximum N applied through livestock manure shall not exceed 170 kg^{-1} ha^{-1} $year^{-1}$, and there are "closed periods" for manure and fertilizer applications), as well as the obligation for leak-tight manure storages with storage capacities per farm of 6−9 months (Oenema, 2004). The zero-discharge manure systems in the United States regulate manure storage, land application, and whole-farm nutrient management planning, resulting in a higher manure recycling efficiency than in China (Centner, 2001; Centner and Feitshans, 2006) (Table 5.1). Incentives and taxes can have an important role in some of these policies; for example, excess phosphate on farms in The Netherlands was taxed at €9.08 for each additional kilogram that exceeds a defined limit until 2006 (Brinkhorst and Pronk, 1999). However, until a new policy was initiated in 2014, to control the environmental problems resulting from industrial livestock production systems (SCC, 2013), there had been little in the way of manure regulations in China. Direct discharge of manure to water bodies is still a major loss pathway. Discharge of manure is forbidden in the European Union and the United States because of the implementation of the Nitrates Directive and resolutions of specific member states in the

European Union and U.S. federal regulations (England Regulations, 1991; Centner, 2001; Centner and Feitshans, 2006; Velthof et al., 2014). Current polices in China mainly focus on manure processing and treatment to promote the recycling of manure (Meng et al., 2014). However, our results show that the amount of manure that is treated and subsequently applied to crops represents only a small part of the amount of nutrients excreted (Fig 5.1). Hence, the effectiveness of these policies is still low.

5.4.3 Implications for future manure management

Grassland is extensively managed in China, barely receiving any fertilizers. Hence, improved recycling of manure from animal confinement in grazing systems will not replace fertilizer, but it might yet improve pasture production. In contrast, fertilizer and manure P application rates of 261 and 310 kg of P ha^{-1} $year^{-1}$, respectively, have been reported for greenhouse vegetable production systems (Yan et al., 2013). Although production is high in these systems, the high fertilizer applications rates suggest that manure applied to these cash crops has not replaced much fertilizer. Only manure nutrients applied to cereal crops were considered to replace fertilizer nutrients in this study. However, there is a large potential for replacing fertilizer N, P, and K by manure N, P, and K.

In 2020, the total nutrient excretion is projected amount to 26 Tg of N, 5.2 Tg of P, and 19 Tg of K. The results presented in Fig 5.5 indicate that the objectives of zero fertilizer growth by 2020 can be achieved and that fertilizer use can be reduced by 27% for N, 82% for P, and 100% for K if the integration of manure management options (S4) can be implemented successfully. Even more fertilizer can be saved if over-fertilization is decreased through the implementation of balanced fertilization (Ma et al., 2013; Luesink et al., 2014).

Not only is recycling of manure environmentally beneficial, through the mitigation of nutrient losses, it is also economically profitable. The annual value of manure nutrients in China is equivalent to 190 billion Yuan (roughly €25 billion), based on the prices of N, P, and K in fertilizers

(Chadwick et al., 2015) and the availability of N, P, and K present in animal manure. However, the cost of manure application to cropland is higher than the cost of fertilizer application. In 2015, about 100 billion Yuan of subsidies (equivalent to €13 billion) was provided to the fertilizer industry in China. Redirecting these subsidies for the fertilizer industry toward manure storage infrastructure, manure transportation, and manure application would promote more sustainable use of manure nutrients in the future.

There are also other barriers to recycling manure nutrients to agricultural land effectively and, hence, reducing losses, such as a lack of information about manure nutrient contents and their bioavailability, a shortage of machines to transport and apply manure, inefficient extension services (Chadwick et al., 2015), and a poor infrastructure in terms of housing and storage systems. More studies are required to improve the accuracy of the estimations of nutrient losses and nutrient efficiency throughout the whole manure chain. In particular, information is needed on the fertilizer replacement value and nutrient availabilities of different manure types applied to the major crops in typical soils and climates in China, to provide the evidence base for recommendations regarding a manure management system. Further investments are needed to improve the infrastructure and management of farms.

Extensive livestock systems are generally found in remote areas in China, such as in the Northwest, where land degradation is serious, in part because of overgrazing. Intensive livestock production tends to cluster in locations with cost advantages (often close to cities) where insufficient land is available for the recycling of waste from livestock. This can lead to an overload of manure nutrients in these regions. A region-specific analysis is needed to propose region-specific strategies for effective manure use and mitigation of nutrient losses (Jia et al., 2015). There are also other potential environmental problems related to manure management; for example, antibiotics and heavy metals are of concern. Residues of veterinary antibiotics have been detected in manures and surface waters around livestock production farms (Zhao et al., 2010; Wei et al., 2011) and even in the urine samples of children (Wang et al., 2015). These problems should also be considered through strategies for manure-chain management.

To conclude, only 33% of the excreted N is recycled, which is less than has been estimated in previous studies. Further, 78% of the N excreted by livestock in China is lost to the environment (including losses after manure application to land, which account for 11% of total N excretion). Nutrient use efficiencies and manure recycling efficiencies differ greatly among systems and animal categories. There is considerable potential to reduce N, P, and K losses from the manure management chain; to increase the amount of manure applied to cropland; and to replace fertilizer N, P, and K through the adoption of integrated options. However, to improve manure utilization, large changes and investments in the livestock farm infrastructure, namely, animal housing, manure storage, and facilities for manure transportation and application, are needed. An integrated manure and fertilizer nutrient recommendation system has to be developed that takes account of the total nutrient and available nutrient contents of manures. Finally, the resulting improved knowledge needs to be disseminated to farmers.

5.5 Acknowledgement

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CHAPTER 6

Spatial planning needed to drastically reduce nitrogen and phosphorus surpluses in China's agriculture

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Abstract

China's fertilization practices contribute greatly to the global biogeochemical nitrogen (N) and phosphorus (P) flows, which have exceeded the safe-operating space. Here, we quantified the potentials of improved nutrient management in the food chain and spatial planning of livestock farms on nutrient use efficiency and losses in China, using a nutrient flow model and detailed information on >2300 counties. Annual fertilizer use could be reduced by 26 Tg N and 6.4 Tg P following improved nutrient management. This reduction N and P fertilizer use would contribute 30% and 80% of the required global reduction, needed to keep the biogeochemical N and P flows within the planetary boundary. However, there are various barriers to make this happen. A major barrier is the transportation cost due to the uneven distributions of crop land, livestock, and people within the country. The amounts of N and P in wastes and residues are larger than the N and P demand of the crops grown in 30% and 50% of the counties, respectively. We argue that a drastic increase in the recycling and utilization of N and P from wastes and residues can only happen following relocation of livestock farms to areas with sufficient cropland.

6.1 Introduction

Human pressures on the earth-system have increased to unprecedented levels, with many of these pressures having severe impacts on the stability of the earth-system. Nine intrinsic biophysical processes that regulate the stability of the earth-system have been identified, and four out of these nine have breached their boundaries. The biogeochemical flows of nitrogen (N) and phosphorus (P) have been considered to even reach a high-risk zone (Rockström et al., 2009; Erisman et al., 2015; Steffen et al., 2015). N and P are indispensable elements for all life on earth, and thus for food production. However, increasing inputs of N and P to agriculture have decreased the utilization efficiency of N and P in food production, and have led to increased losses of N and P to the environment and to pollution of surface waters and air (Liu et al., 2010; MacDonald et al., 2011; Sutton et al., 2011; Erisman et al., 2015; Lun et al., 2018). It has been estimated that the total N and P fertilizer input to agriculture needs to be reduced by at least 50% globally to be able to keep the global geochemical N and P flows within the suggested planetary boundaries (de Vries et a., 2013; Steffen et al., 2015). Most of the environmental effects of N and P become visible on the local to regional range, which increases the incentive to also perform measures at such spatial dimensions.

China will have an important role in achieving planetary boundaries for N and P flows, as China consumed around one third of global N and P fertilizers during the past decade (FAO, 2019), and it faces serious water and air pollution due to low N and P use efficiencies (Liu et al., 2013; Yu et al., 2019). The central government has set a “zero increase target” for N and P fertilizer use between 2016 and 2020 to alleviate the environmental pollution (MOARA, 2015). Although a big step for farmers and industries, this target is far below the requirement to reduce N and P losses to acceptable levels. Several additional measures have been discussed, including more efficient fertilization (Ju et al., 2009; Cui et al., 2010), improved livestock manure management, improved linking of crop production and livestock production (Bai et al., 2016; Garnier et al., 2016; Zhang et al., 2019), diet manipulations, and reduced food wastages (Ma et al., 2013; Liu et al., 2016). Large potentials to reduce both N and P fertilizer inputs have been estimated. However, these measures focused only on certain sectors of the agro-food system, and neglected significant amounts of nutrients in the whole "soil−crop−livestock−food processing−food consumption" chain, that are potentially available for recycling. Earlier studies have shown that N and P use efficiency in the food chain was low and that N and P losses were high in China (Ma et al., 2010; Ma et al., 2019). This indicates that there is a need to consider the potential to recycle N and P from all wastes and residues of the food chain, and to estimate the potential N and P fertilizer savings.

It is well-known that not all N and P contained in recycled organic resources from the food chain are readily available to crops; for example, only 10% to 70% of the nitrogen in livestock manure is available following application to cropland, depending on the type of manure (Jensen, 2013; Webb et al., 2013). If synthetic fertilizer is replaced by manure without consideration of the bioavailability of the manure, then there may be negative impacts on crop yield and possibly on food security. Hence, the bioavailability of nutrients in recycled organic resources has to be considered, also how the bioavailability is impacted by nutrient management practices, such as ammonia mitigation measures (Bittman et al., 2014). Such considerations have not been conducted yet in N and P fertilizer use projections for China.

Previous studies discussed the potentials to reduce fertilizer inputs at the national level, while ignoring the geographic disconnections between crop production, animal production, and urban areas; the availability of organic resources, such as livestock manures and household residues, is often limited in rural areas, despite its abundance in and around urban areas. Other studies have pointed out that a subnational spatial linking of cropland and livestock agriculture are needed, combined with a strategy to replace mineral fertilizer by manure (Nesme et al., 2015; Bai et al., 2018; Swaney et al., 2018; Svanbäck et al 2019). This indicates that the potentials for recycling of N and P from manures and wastes has to be examined at regional and local levels.

Here, we explored the potentials to recycle N and P from manure and wastes from the food chain in crop land at the county level, and thereby the potentials to reduce N and P fertilizer use in China. The updated NUtrient flows in Food chains, Environment Resources use (NUFER) county model was used, which contains data and information on more than 2300 counties (Wang et al., 2018; Chen et al., 2019). The potentials to recycle N and P from manure and wastes from the food chain in crop land were examined at county level, and national level; the difference between the two estimates indicates the current geographic barriers for recycling N and P from manures and wastes, and for reducing fertilizers input.

6.2 Material and Methods

6.2.1 NUFER model

The modified NUFER-county model was used to quantify the N and P flows in the whole food chain. The original NUFER model simulates the N and P flows in the "soil−crop−livestock−food processing−food consumption" chain at the national level in China, but the county version is able to estimate the N and P flows in the food chain at county level. Both model versions consider the food chain as a steady state for one particular year. NUFER comprises an input submodule (human activity, agricultural production activity), a calculation module, and an output module (different type of nutrient losses, food export, nutrient accumulation in soil). The NUFER-county model covers 2333 counties (including districts in the urban area), but does not cover counties in Xinjiang, Tibet, and Qinghai provinces, due to lack of available data. These regions contribute <3.6% to the total crop production and fertilizer use in China, and therefore have limited impacts on the results at the national and county level (NBSC, 2019).

County-specific model input data were used, including (i) human activities in the food chain, (ii) transformation and partitioning coefficients to match the data at county, provincial and national levels, and (iii) N and P contents and loss factors. Data on human activities were derived from county statistical reports. The NUFER-county model was further improved by including crop yield dependent biological N fixation for legume crops (Lassaletta et al., 2014a).

$$N_{fixed} = N_{dfa} \times (Y \div NHI) \times BGN \qquad (1)$$

where N_{fixed} is the amount of N fixed by crops (kg N $ha^{-1}yr^{-1}$), N_{dfa} is the percentage of N uptake derived from N fixation (%), Y is the harvested yield (expressed in kg N $ha^{-1}yr^{-1}$), NHI is the N harvest index (dimensionless), defined as the ratio of N in the harvested material to the total N in aboveground production, and BGN is a multiplicative factor taking into account the contribution to total N_2 fixation of belowground fixation associated with roots and nodules production as well as to rhizodeposition via exudates and decaying root cells and hyphae (dimensionless) (Lassaletta et al., 2014a).

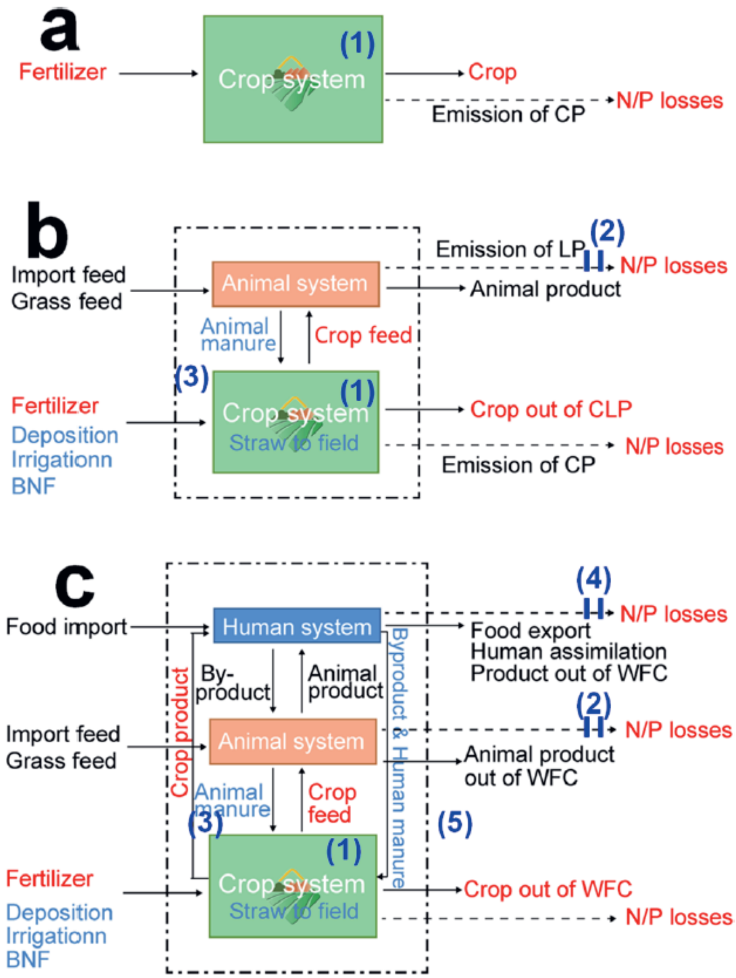

Figure 6.1. System boundaries for the different strategies considered in this study: S1 and S1-IM (a), S2 and S2-IM (b), and S3 and S3-IM (c). Note: S1: Balanced fertilization in crop production; S2: S1 + integrated nutrient accounting in crop-livestock production; S3: S2 + integrated nutrient accounting in the whole food chain; S1-IM: S1 + improved soil management; S2-IM: S2 + improved soil management + emission mitigation control; S3 -IM: S3 + improved soil management + emission mitigation control + improved recycling. *CP, crop production; LP, livestock production; CLP, crop-livestock production system; WFC, whole food chain; BNF, biological nitrogen fixation. The values with brackets represent the improvement of*

nutrient management of different system. (1) Increasing of soil fertility; (2) improved livestock manure management with low ammonia emission; (3) abandon discharge of manure and increase recycling of livestock manure; (4) improve nutrient management of human excretions with low ammonia emission; and (5) new system to recycle human excretion and food waste.

The N and P losses via surface runoff, erosion and leaching were estimated as a function of land use, precipitation, soil depth, soil type, temperature, soil texture, and soil organic matter content at the county scale. The detailed method has been described in Zhao et al (2019). The data and parameters were derived from the Data Center for Resources and Environmental Sciences, Chinese Academy of Sciences (RESDC, 2019), or estimated via the spatial interpolation methods applied by RESDC.

6.2.2 Strategies to reduce synthetic N and P fertilizer use

We developed two main strategies to reduce the required N and P fertilizer input: (i) recycling of N and P from manures, wastes, and residues in the food system, to substitute the synthetic fertilizer; (ii) improved technologies to reduce nutrient losses and to increase the bioavailability of N and P in recycled organic resources, and reduce synthetic fertilizer toward matching crop needs.

Three levels of system boundaries have been considered: crop production, crop-livestock production, and the whole "soil−crop−livestock−food processing−food consumption" chain. These system boundaries are represented in Fig 6.1a−c. For each level of system boundaries in spatial optimization, two sets of technology have been explored, one reflecting a business as usual situation, and one of improved technologies. The resulting six strategies provide an illustrative comparison of possible impacts to the base situation. Hence, no changes in crop and livestock production yield and structure were assumed with respect to the reference year situation of 2012. Also, there were no changes in feed and feed harvest from natural areas within China, and imports of food and feed from other countries were also assumed to remain constant (2012 level). All strategies were simulated for the national and the county scales.

The year 2012 was used as a reference year, because of the availability of data and parameters. Possible changes in the recycling of N and P from manures and wastes in the food chain, and the possible replacement of synthetic N and P fertilizers by recycling N and P were also estimated for the year 2012.

6.2.3 Description of strategies

Strategy S1. Balanced N and P fertilization in crop production (Fig 6.1a). Balanced fertilization was defined as "total available N (or P) from synthetic fertilizers equals total crop N (or P) uptake corrected by a crop N (or P) uptake factor". The crop N (or P) uptake factor reflects that not all applied fertilizer N (or P) can be taken up by the crop effectively, also because there are always "unavoidable" losses of N and P to the wider environment. The crop N (or P) uptake factor was introduced to ensure no reduction of crop yields, and fits in the "food security first" policy in China. The N and P uptake factors of different crop species are listed in Table S1 of the Supporting Information (SI). The required synthetic N (or P) fertilizer input was estimated as follows:

$$Ic_{fertilizer} = \sum_{i=1}^{n}[(Oc_{main\ product,i} + Oc_{straw,i}) \times UF_{crop,i}] + Oc_{managed\ grass} \times UF_{managed\ grass} - Ic_{soil\ mineralization} \quad (2)$$

where $Ic_{fertilizer}$ is the total input of synthetic N (or P) fertilizer, in kg N (or P); $Oc_{Mainproduct,i}$ and $Oc_{straw,i}$ are the amounts of N (or P) in the main crop product and straw per county, respectively, in kg N (or P); $Oc_{managred\ grass}$ is the amount of N (or P) in harvested grass from managed grassland per county, in kg N (or P); UFcrop,i and $UF_{managred\ grass}$ are the uptake factors for crop species and grass, respectively (dimensionless) (Table S1); $Ic_{soil\ mineralization}$ is the net release of N (or P) from the mineralization of soil organic matter per county, which were derived from maps from the Ministry of Agriculture and Rural Affairs. The average net N (or P) mineralization rate was dependent on the soil organic matter content and cropland area; soils with a high soil organic matter content (>4.0%) may release 43 kg N per ha, while soils with a medium (2.5%−4.0%) and low soil organic matter content (<2.5%) may release 27 and 11 kg N per ha per year, respectively (DEFRA, 2010). Requirement for P addition was calculated using soil Olsen-P content: At soils with high Olsen-P content (>40 mg kg^{-1}), 100% of crop uptake was considered to be replenished by fertilizer addition, while this value increased to 110% and 120% of crop P uptake in soils with a medium (20–40 mg kg^{-1}) and low (<20 mg kg^{-1}) Olsen-P content, respectively (MOARA, 2015). Further, we assumed that balanced fertilization reduced ammonia emission, runoff, erosion, and leaching factors by 40% relative to the reference situation (Oenema et al., 2009; Velthof et al., 2009). Note that S1 does not consider other N (or P) additions as from manure, seed, or crop residue material, atmospheric deposition, or biological fixation, which all are being maintained constant. Hence significant excess application may still occur.

Strategy S2. Balanced fertilization and improved nutrient accounting in the crop–livestock production sector (Fig 6.1b). A number of recent studies emphasized the need to recouple crop and livestock production. This would allow us to increase nutrient recycling, and hence reduce the external new nutrient input in the agricultural system (Nesme et al., 2015; Bai et al., 2018; Zhang et al., 2019). Here, we assumed that N (or P) inputs from animal manures, atmospheric deposition, biological N_2 fixation, and irrigation were taken into account in the N (or P) accounting. The required synthetic N (or P) fertilizer input was estimated as follows:

$$Ic_{fertilizer} = \sum_{i=1}^{n}[(Oc_{main\ product,i} + Oc_{straw,i}) \times UF_{crop,i}] + Oc_{managed\ grass} \times UF_{managed\ grass} - Ic_{soil\ mineralization} - Ic_{deposition} - Ic_{BNF} - Ic_{irrigation} - \sum_{i=1}^{n}\left(Ic_{straw\ back\ to\ field,i} \times AF_{straw\ back\ to\ field,i}\right) - \sum_{i=1}^{n}(Ic_{animal\ manure\ back\ to\ field,i} \times AF_{animal\ manure\ back\ to\ field,i}) \quad (3)$$

where Icdeposition is the deposition of atmospheric N (kg N), Ic_{BNF} is the N input via biological N fixation (kg N), $Ic_{irrigation}$ is the N input via irrigation water (kg N), $Ic_{straw\ back\ to\ field,\ i}$ is N (or P) input via crop straw return (kg N (or P)), $Ic_{animal\ manure\ back\ to\ field,\ i}$ is the N (or P) input via animal manure (kg N (or P)). $AF_{straw\ back\ to\ field,i}$ and $AF_{aniaml\_manure\_back\_to\_field}$, i are the mineral fertilizer values of straw and manure, respectively (dimensionless) (Table S2). Since manure P is almost 100% available to crops, mineral fertilizer values for P in manure were assumed to be constant (set at 1.0) for all strategies. Mineral fertilizer values of atmospheric N deposition, BNF, and N in irrigation were also set at 1.0.

Strategy S3. The whole food system strategy; balanced fertilization, improved nutrient accounting in the crop-livestock sector, and improved nutrient accounting of N (or P) inputs from the recycling of food waste and human excreta to crop land (Fig 6.1c). The required N (or P) fertilizer input was estimated as follows:

$$Ic_{fertilizer} = \sum_{i=1}^{n}[(Oc_{main\ product,i} + Oc_{straw,i}) \times UF_{crop,i}] + Oc_{managed\ grass} \times UF_{managed\ grass} - Ic_{soil\ mineralization} - Ic_{deposition} - Ic_{BNF} - Ic_{irrigation} - \sum_{i=1}^{n}\left(Ic_{straw\ back\ to\ field,i} \times AF_{straw\ back\ to\ field,i}\right) - \sum_{i=1}^{n}\left(Ic_{animal\ manure\ back\ to\ field,i} \times AF_{animal\ manure\ back\ to\ field,i}\right) - Ic_{food\ byproduct} \times AF_{food\ byproduct} - Ic_{human\ manure} \times AF_{human\ manure} \quad (4)$$

where $Ic_{foodbyproduct}$ and $Ic_{human\ manure}$ are the N (or P) input via recycled food waste and human excreta, $AF_{foodbyproduct}$ and $AF_{human\ manure}$ are the mineral fertilizer values of the treated (composted) food waste and human excreta (Table S2).

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Strategy S1-IM. As in S1, but with improved soil management and crop husbandry, including soil fertility management, erosion control, crop rotation, and green manuring (Fig 6.1d). We assumed that these practices will lead to a considerable improvement of soil fertility (MOARA, 2015). As a result, net soil N and P mineralization will increase.

Strategy S2-IM. As in S2, but now with improved soil management and emission mitigation in livestock production (Fig 6.1e). We assumed that ammonia emissions from livestock production will be reduced by 50%, which is in agreement with the recent target of the National Key Research and Development Program in China (MOST, 2018), through a combination of measures, including acidification of slurry, covering slurry storages, and closed manure composting technologies (Hou et al., 2015; Cao et al., 2019). As a result, the mineral fertilizer value of the N in animal slurries and manure will be significantly improved (Table S2). At the same time, we assumed a strict ban on the discharge of manure to watercourses or landfill; hence, we assumed that all the livestock manure was collected and ultimately applied to crop land.

Strategy S3-IM. As in S3, but now with improved soil management, emission mitigation in livestock production, and enhanced collection, sanitation, and utilization of N (or P) in food waste and human excreta (Fig 6.1f) (Yu et al., 2019). A new system will be built to collect human excretions which instead go to a sewage treatment system, hence, the nutrients will be preserved and recycled. The estimated mineral fertilizer value of N in composts from food wastes and human excreta are presented in Table S2.

6.2.4 Cumulative distribution of nutrient uptake and supply

We define manure N (or P) loading as the ratio between total manure N (or P) excretions and total N (or P) withdrawal in harvested crop in a county (in kg). A low manure loading ratio refers to a low manure N (or P) excretion relative to the amounts of N and P in harvested crop within a county. A high manure loading ratio refers to a manure N (and/or P) surplus within a county. For a cumulative distribution curve, all counties were plotted in a graph along the X-axis in ascending order of their manure loading ratio, with either total N (or P) withdrawal with harvest crop, or manure N (or P) excretion, or fertilizer N (or P) application on the Y-axis.

6.3 Results and discussion

6.3.1 Effects of improved nutrient accounting on synthetic fertilizer input reduction

The total input of synthetic N and P fertilizers was 31 Tg N and 6.5 Tg P in 2012 (Fig 6.2). Balanced fertilization (S1) would reduce the total input of fertilizers to 28 Tg N and 5.5 Tg P, a reduction of 15% and 9%, respectively, compared to 2012 (Fig 6.2). This will lead to strong reduction of N losses, especially from the crop production (Fig 6.3). However, N and P use efficiencies in the whole food system did not change much, as there were no improvements of nutrient management in the livestock, food processing, and consumption sectors (Fig 6.4). Note that "balanced fertilization" in S1 does not account for inputs as BNF, atmospheric deposition, and irrigation. It is a simple first step strategy, designed for local policy makers to implement at the county level, as they have as yet little knowledge about nutrient management (Gao et al., 2009).

There will be greater reductions of required synthetic N and P fertilizer inputs in the integrated crop-livestock management strategy (S2). Accounting for the N and P in animal manures, BNF, atmospheric deposition, and irrigation reduce the total required input of synthetic fertilizers to 16 Tg N and 3.4 Tg P, a reduction of 44% and 38%, respectively, compared to S1 (Fig 6.2). The strong reduction in required synthetic fertilizer input is mainly the result of accounting for the vast amounts of N and P in animal manures, even though the mineral fertilizer value of recycled manure N and P was assumed to be low due to its poor management (Bai et al., 2016). In addition, there were accountable inputs via the return of crop straw and residues from other crops (Gao et al., 2009) and atmospheric N deposition (Liu et al., 2013).

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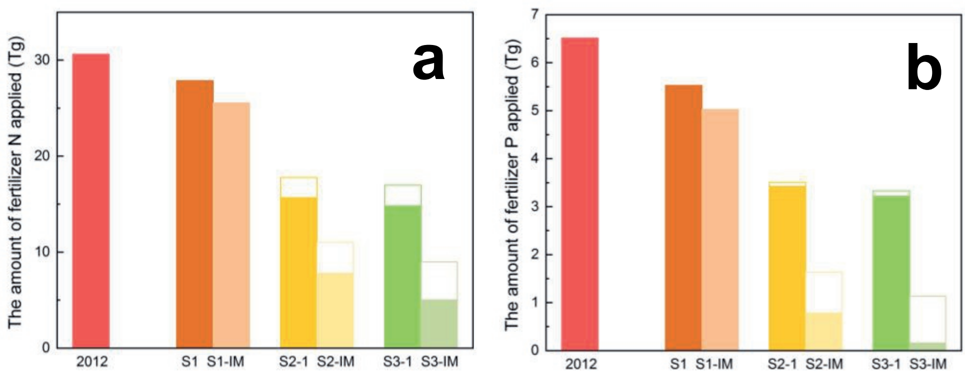

Figure 6.2. Inputs of synthetic nitrogen (N) fertilizer (a) and phosphorus (P) fertilizer (b) to Chinese agriculture in 2012, and the required inputs of

synthetic N and P fertilizers for various strategies. *The solid (filled) bars represent the required synthetic N and P fertilizer inputs, following assumptions and estimations at the national level. The blank top-up bars represent the estimated required inputs following assumptions and estimations at county level. S1: Balanced fertilization in crop production; S2: S1 + integrated nutrient accounting in crop-livestock production; S3: S2 + integrated nutrient accounting in the whole food chain; S1-IM: S1 + improved soil management; S2-IM: S2 + improved soil management + emission mitigation control; S3-IM: S3 + improved soil management + emission mitigation control + improved recycling*.

Accounting for the N and P inputs from food waste and human excreta (S3) did not further decrease the required synthetic N and P fertilizer inputs (Fig 6.2), as the N and P from human excreta and food wastes were minimally returned to crop land in 2012. Note that the required inputs of synthetic N and P fertilizers were lower when the estimations were conducted at national scale than at county scale (Fig 6.2). The estimations at county scale assumed that the recycled N and P from manures, crop residues, food wastes and human excreta were recycled within the county where they were produced, for all >2300 counties. The estimations at national scale assumed that recycling occurred within the country, but without considerations of distances between the sites of production and sites of utilization.

6.3.2 Effects of improved nutrient management on synthetic fertilizer input reduction

There are strong differences in required synthetic N and P fertilizer inputs between the current situation and the following enhanced nutrient management strategies (Fig 6.2), as technologies are implemented to increase solid and liquid manure collection, transportation, and application to crops according to the nutrient demand. Also, this strategy assumes that technology has been installed that allows one to collect and treat the sewage water, which then enables recycling of nutrients to cropland. Our estimates suggest that the required inputs of synthetic N and P fertilizers could be reduced ultimately to 5.0 Tg N and 0.16 Tg P (S3-IM) for the national scale analysis. Clearly, improved nutrient management in crop−livestock production (S2-IM vs S2) and in the whole food chain (S3-IM vs S3) greatly reduces the required input of synthetic N and P fertilizer. The differences are larger for P than for N, because P losses from crop−livestock production and from the whole food chain may be reduced more easily through improved collection and emission mitigation than N losses (Tonini et al., 2019; Withers, 2019). The estimated reductions in required synthetic N and P fertilizer inputs strongly depend on the mineral fertilizer value of the recycled nutrient resources (Table S2). There is greater uncertainty in estimated mineral fertilizer value in the short term than in the long term; overestimation of the short-term mineral fertilizer value will increase the risk of crop yield declines (Jensen, 2013; Webb et al., 2013).

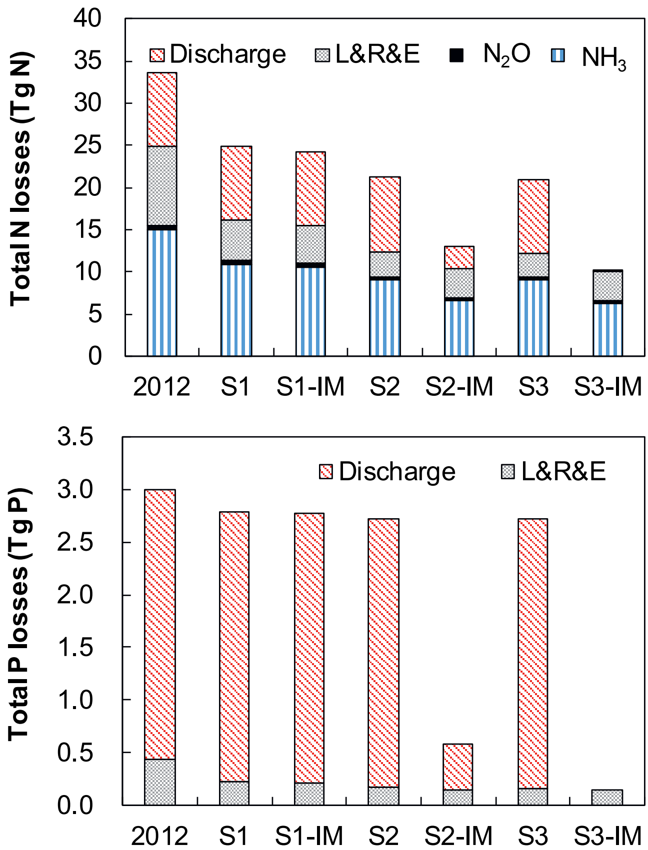

Figure 6.3. Total nitrogen (N) and phosphorus (P) losses from the whole food chain of different strategies at the national level in 2012. *L&R&E is the leaching, runoff and erosion losses.*

Improved nutrient management greatly reduces the losses of N and P from the food chain to the environment (Fig 6.3). The effects are notably large for P in crop-livestock production (S2-IM vs S2) and in the whole food chain (S3-IM vs S3), because of the strong decrease in discharges to surface waters or landfills (Fig 6.3). Conversely, N losses from the food chain are more diffuse and basically all strategies contribute to a reduction in N losses. Our estimates suggest that N losses may be reduced ultimately by ~70% and P losses ultimately by ~90%. However, these are likely overestimates because the estimations are based on national scale analyses.

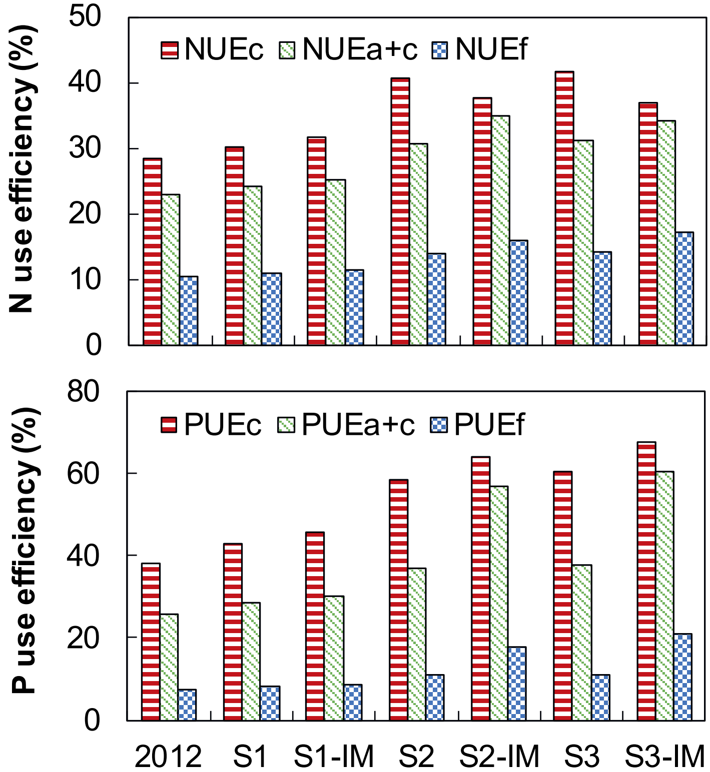

Figure 6.4. Nitrogen (N) and phosphorus (P) use efficiency in crop production (NUEc and PUEc, respectively), in crop-livestock production

(NUEa and PUEa, respectively), and in the food chain (NUEf and PUEf, respectively) in 2012 and in 2050 for different strategies.

Improved nutrient management increases the N and P use efficiency in crop production, crop-livestock production and in the whole food chain (Fig 6.4). Increases are larger for P use efficiency than for N use efficiency. Interestingly, not all strategies increase N use efficiency equally well; small decreases reflect that highly available synthetic N fertilizer was replaced by inputs of moderately available N from recycled resources. Relative increases in N and P use efficiency were largest for the whole food chain and least in crop production.

The N use efficiency in crop production increased from 29% in the reference year 2012 to a maximum of about 42% in S3, which is a modest increase. However, this modest increase hides that the N input sources have greatly altered from highly available synthetic N fertilizer to moderately available N in composts and residues. Basically, the N use efficiency in S1 is overestimated, because various possible N sources are not accounted for in the calculations. Evidently, the N and P accounting is most complete for the food chain system, and as a result the relative increases in N and P use efficiency are largest for the whole food system.

Human excreta were a main source of N (4.7 Tg) and P (0.5 Tg), but these were not used effectively in 2012 (Fig S1). Discharge of sewage water was found to be one of the main sources of N and P in watercourses in 2010 (MOEP, 2010). The central government has invested around 21 billion US $ in sewage treatment plants since 2014 to treat 49 billion m^3 sewage water per year (FAO, 2019). These sewage treatment plants were built nearby urban areas (Fig S2), and “remove” about 26% of the nutrients through treatment, while the rest ends up in watercourses (Wu, 2014; Zhao et al., 2015). Recycling of household waste and human excreta in crop land was common practice before the 1980s, but has largely vanished because of concerns about the fecal oral transmission and fecal-body transmission of communicable diseases and pathogens. Currently, there are no institutions and markets anymore for recycling of household wastes and human excreta as composts in agricultural land.

Furthermore, it has been estimated that around 20% of grains and 50% of fruits and vegetables are wasted or lost before reaching the dining table (Liu et al., 2013; Ma et al., 2019). Although some of these wastes are being used as animal feeds, most of the food wastes ends up in garbage burning installations or landfill sites (Hu et al., 2012). These wastes contain approximately 0.9 Tg N and 0.3 Tg P (Fig S1). Largest underutilized nutrient resources were animal manures in 2012. Approximately 12.2 Tg N and 2.1 Tg P were lost from the manure management chain in 2012 (Fig S1). A combination of improved manure collection and storage, appropriate emission mitigation measures and targeted application of manure to crop land may greatly increase manure nutrient utilization and decrease N and P losses from the manure chain (Bai et al., 2016).

6.3.3 Spatial disconnection of nutrient supply and demand

There is a large divide between estimations of the nutrient recycling potentials at national scale and at county scale. The nutrient recycling potentials and hence the fertilizer input reduction potentials in the S2 and S3 strategies were much smaller when the estimations were made at county scale than at country scale. For example, the required N fertilizer input in the S3-IM strategy was about 5.0 Tg when based on national-scale analyses and about 9.0 Tg when based on county-scale analyses. The difference is even bigger for P, the county-aggregated demand of P fertilizer was 1.1 Tg, which was more than 5 times that of the national-scale analysis in the S3-IM strategy. The main difference between the county and national scale analyses is that the county analysis excludes cross-county border transportation of nutrient resources. Although this is a gross simplification of reality, especially along borders of counties, this analysis accounts for the barriers involved with long-distance transport of wastes such as the high transportation cost and the risk of the transmission of pathogens. For example, the average profit of pig production ranged between 12 and 24 US \$ head^{-1} during July 2017 to January 2018, which was before the outbreak of African Swine Fever (MOARA, 2020). Each slaughtered pig produced around 1 ton of manure, for which the average transportation cost was around 0.30 US \$ km^{-1}. Transport of manure to farms 40–80 km away will neutralize all profits of pig production, a distance typically still within the county border (MOARA, 2020). The provincial level results are showed in Fig S3, and are not in-depth described here.

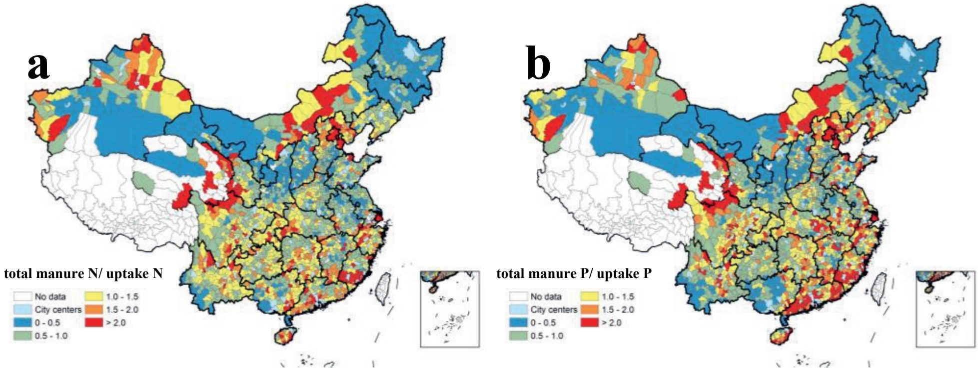

Figure 6.5. Map of the distribution of the manure N loading (a) and manure P loading (b) at county level in 2012. The manure N (or P) loading is defined as the ratio of the total excretions of N (or P) by livestock and humans and the N (or P) withdrawal with harvested crops.

The main reasons for the large differences between county and national level analysis in nutrient recycling potentials is due to the uneven distributions of productive crop land, livestock, and human population in China. The total amounts of N in livestock manure and human excreta distributed on arable systems exceeded the total uptake by crops in many counties in 2012 in the S3-IM strategy, especially in the Yangtze River Basin, which covers Sichuan, Chongqing, Hunan, Jiangxi, and Zhejiang provinces (Fig 6.5a). These provinces are mountainous and have a high density of watercourses. Livestock farms are often near villages and urban areas, and spatially disconnected from cropland by mountains and water courses, which hinders the transport of the voluminous livestock manures to crop land. The mismatch between demand and supply is even larger for P in some counties; the supply of P in livestock manures exceeds crop demand in the Yangtze River Basin, the Pearl River Delta and Fujian province (Fig 6.5b). Further differences were introduced by excluding Xinjiang, Gansu, and Tibet from the calculations, for which county level data were unavailable. As their contributions were relatively small (<3.5% of total crop N or P uptake at the national level), and as the livestock and crop production are evenly distributed in these provinces, with grassland based ruminant animal production systems, the lack of data will likely not strongly affect the overall result (NBSC, 2019).

For the S3-IM strategy, the mean manure N and P loadings per county are presented in Fig 6.6 in ascending order on the x-axis, while the cumulative manure N and P loadings are presented on the y-axis. Manure N (or P) loading is defined here as the ratio of mean N (or P) supply via livestock manure and demand by the crop. A ratio of <1 means that total supply is lower than total demand within a county. About one-third of the number of counties had a manure surplus. The cumulative surplus was 3.1 Tg N and 1.0 Tg P for the counties with a surplus (Fig 6.6). This indicates that these amounts of manure N and P cannot be used effectively as a substitute for synthetic N and P fertilizers, because of the spatial disconnect between supply and demand. Surprisingly, the counties with a manure surplus used about 1/3 of the N fertilizer in 2012. This reflects overuse of both manure N and fertilizer N (Fig 6.6a). Situations were even worse for P (Fig 6.6b).

Similar but less extreme situations have been found at country level in a global study. Lassaletta et al (2014) found that increasing trade of animal feed has contributed to decoupling of crop production and livestock production; livestock manure is rarely transferred back from feed importing countries to feed exporting countries (Swaney et al., 2018). In the Baltic Sea drainage basin in Europe, a high ammonia emission intensity occurred in regions with both high mineral fertilizer N and manure N applications, suggesting that animal manures were disposed of on cropland near farms and that mineral fertilizer N applications were not much corrected for the manure N input (de Vries et al., 2011; Hong et al., 2017). An exception is perhaps The Netherlands, where the surplus manure P produced (about 25%

of total P excretion) has to be exported (WUR, 2020), mostly to neighboring countries (Germany and France), but also to far-distance countries including Ukraine, South Korea, and China. Far-distance transport increases the cost of the processed manure products and its use is restricted therefore to niche markets.

6.3.4 Required synthetic N and P fertilizer input at the county level

The required synthetic fertilizer input per county and strategy is presented in Fig 6.7 for synthetic N fertilizer and in Fig S5 for synthetic P fertilizer, and the mean values per hectare of cropland are presented in Fig S6 and S7. These maps provide total and means per county, and could be easily used by local governments as targets at the county level. However, additional field level guidance is needed for crop type and field specific recommendations; these should be based also on results of soil testing (Chadwick et al., 2015; Zhang et al., 2015; Xu et al., 2017; Xu et al., 2019; Chawick et al., 2020). Largest inputs are required in the Northeast Plain, North China Plain, and the middle- and down- stream of the Yangtze River (Fig 6.7; Fig S4). These are major grain, vegetable, and fruits producing areas (NBSC, 2019). The relatively large required synthetic fertilizer input in the Northeast Plain and southwest Xinjiang is partly due to its large area of cropland per county (NBSC, 2019).

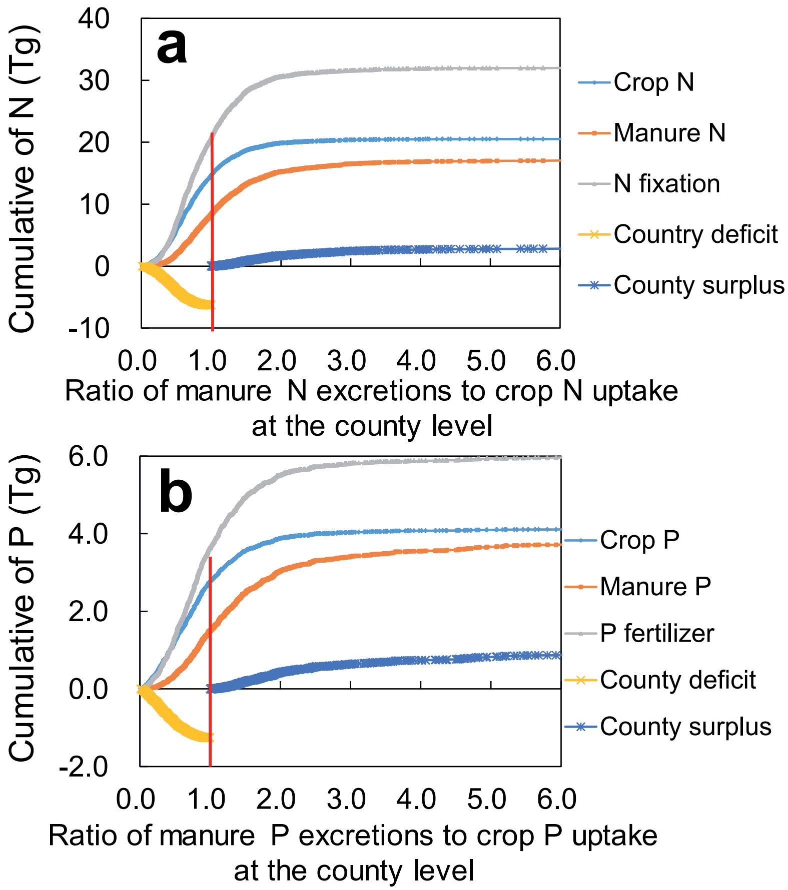

Figure 6.6. Cumulative distribution curves of N (or P) withdrawal in harvested crops, livestock N (or P) excreta, use of N (or P) fertilizer, and the surplus (or deficit) livestock N (or P) relative to the N (or P) withdrawal in harvest crops of counties in 2012. *N, nitrogen; P, phosphorus. All the counties were put into the X-axis in the ascending order of their manure N (or P) loading capacity, and their cumulative contributions to the total production or use were shown in the Y-axis. County surplus is the cumulative positive differences between total livestock N or P excretions and crop uptake. County deficit is the cumulative negative differences between livestock N or P excretions and crop uptake.*

Interestingly, around 30% of the counties appear to have no need for synthetic N fertilizer input, and 50% of the counties appear to have no need for P fertilizer input in S3-IM, because the supply of N and P from livestock manure, crop residues and human excreta exceeds on average the N and P demand by the growing crops in these counties (Fig 6.7; S4). The N and P surpluses in these counties also indicate a large pressure on the environment, especially water quality. These regions either have to invest in manure treatment and manure export to other regions, or will have to relocate livestock farms to other regions. There are several technologies for manure treatment, but economic costs are often high, such as the production of the struvite, the incineration, and the closed continuous composting technologies (Tonini et al., 2019; Withers, 2019; Liu et al., 2020).

The main uncertainty originated from the mineral fertilizer value of livestock manures and organic wastes, which were estimated to range from 0.10 to 1.0. The manures and wastes provide huge amounts of N and P compared with the N and P withdrawal with harvested crop (Fig 6.6), but the fraction of total N that is available for crops is highly uncertain, because the mineral fertilizer value is highly sensitive to weather conditions, crop type and cropping system (single and doubling cropping systems), and soil properties. Hence, small changes of the mineral fertilizer value of manures and wastes have large impacts on the availability of manure and waste N to growing crops, and also had a large impact on the results of our study. Due to lack of data, estimates of the mineral fertilizer value were partly derived from Chinese data (Zhang et al., 2020) and partly from European studies (Jensen, 2013; Webb et al., 2013).

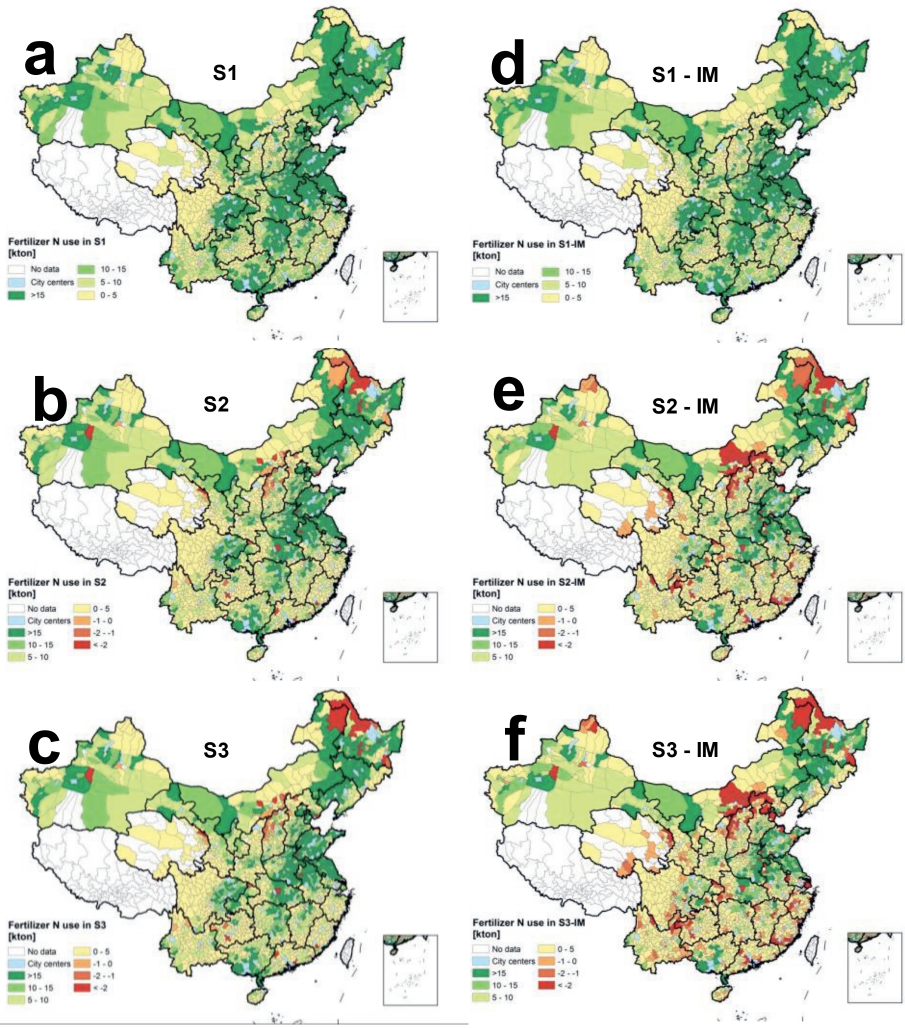

Figure 6.7. Mineral fertilizer (N) demand at the county level under the respective strategies (see Fig 6.1 for definitions). *Blue shades (negative numbers) designate areas where availability manure N already exceeds plant requirements. There might be negative values for the requirement of synthetic N and P fertilizers at the national and county level, due to high available of N and P in the recycled nutrients.*

6.3.5 Suggestions for further steps

The required input of synthetic fertilizer N and P strongly depends on strategy; the required input decreases in the order S1> S1-IM > S2 >

S3 >S2-IM > S3-IM (Fig 6.2). The planetary boundaries for biogeochemical N and P flows at the global level have been estimated a 62 Tg yr^{-1} for N and 6.2 Tg yr^{-1} for P (Steffen et al., 2015). The total global inputs in 2012 were 150 Tg for N and 14 Tg for P (FAO, 2019). If all the required reduction would have to come from synthetic fertilizers, then the total N and P fertilizer inputs need to be reduced by 88 Tg and 7.8 Tg, respectively. In the best strategy (S3-IM), China could save as much as 26 Tg synthetic fertilizer N and 6.4 Tg synthetic fertilizer P by 2030, which is equivalent to around 30% and 80% of the estimated required N and P fertilizer reduction to keep biogeochemical N and P flows within the suggested planetary boundaries at the global scale. However, only a fraction of this potential reduction in fertilizer input can be achieved at short notice, as there are major barriers for such drastic reductions. Our study indicates that improved spatial planning of livestock production is key to fully utilize the potential to recycle livestock manures and wastes.

On the basis of the results of this study, we formulated two complementary recommendations for policy makers in China to achieve the potential improvements in the recycling of N and P from manures, wastes, and residues, and to drastically reduce the inputs of synthetic fertilizers simultaneously. First, improvement of nutrient management in the food system is suggested by the results of the six strategies. There are large opportunities for improving nutrient management practices and for reducing nutrient losses to the environment, but these improvements require investments in knowledge, technology, and institutions. Above all, it requires training of the farmers and their advisors. A series of technologies and polices are needed to efficiently recycle manure (Bai et al., 2016; Bai et al., 2018; Chadwick et al., 2020). Recently, demonstration programs have been established in 100 counties to boost manure recycling, and there are plans for another 200 counties (Chadwick et al., 2020). In addition, zoonotic disease problems of livestock manure need to carefully considered to avoid spread of African Swine Fever or other diseases. The estimated investment needed for building the recycling system for human excreta is comparable to the investment needed to build and manage sewage treatment plants (Yu et al., 2019). However, additional treatment will be needed to prevent and control the transmission of communicable diseases and pathogens, which are major health concerns in the recycling of livestock and human excreta.

The second recommendation relates to improved spatial planning: livestock production must be spatially reconnected again with crop production, to be able to recycle manure nutrients effectively and efficiently. Recently, there has been a relocation of pig farms from south to north, to solve water pollution problems in the south, which has not been without side-effects (Bai et al., 2019). Spatial planning of livestock production areas must be considered from environmental, social, and economic points of view. In any case, excessively high densities of livestock production should be avoided. The regional self- sufficiency of animal-source food production was recently emphasized by the Ministry of Agricultural and Rural Affairs. The cost of implementing changes considering spatial planning of livestock maybe very low after the wide outbreak of African Swine Fever. This was because around 22% of pig production had to be closed down, and it is easy to regulate the geographic site and manure treatment facilities of the newly constructed pig farms, which will with lower additional cost when compare with completely shut down farms in one region and build new one in another region. A new 3-year plan was launched to recover the pig production from the decline through the incidence of African swine fever. The plan proposes a strict spatial planning of pig production away from water courses, but includes the target that >70% of the pork consumption must be produced locally (MOARA, 2019). We argue that additional restrictions are needed related to a maximum pig density per unit of surface area. In addition, major investments are needed in knowledge, technology, and institution to be able to achieve the suggested reductions in fertilizer use through enhanced manure and waste recycling.

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CHAPTER 7

Reallocation of 5-10 billion animals to tackle hot-spots of nitrogen pollution in China

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Abstract

The geographic segregation of crop and livestock production in China has been neglected in efforts to tackle nitrogen pollution of air and water. Here, we analyzed detailed statistical data available from 2300 counties for 1990 and 2012, and evaluated changes in nitrogen losses in response to changes in spatial distributions of livestock, using scenario analyses. Over 45% of livestock production was in counties with a livestock density ≥4.0 livestock units ha^{-1}, and 60% of people were exposed to excessive ammonia emissions in 2012. Reallocation of livestock to reduce the exposure of people to ammonia led to increased total nitrogen losses, because of less favorable production conditions in other counties. Reallocation of livestock according to crop-livestock integration criteria reduced total nitrogen losses, but increased the exposure of people to ammonia. Spatial planning is a powerful policy instrument to tackle nitrogen pollution and exposure of humans to ammonia, but requires clear criteria agreed by stakeholders.

7.1 Introduction

China is suffering from severe environmental pollution caused by nitrogen (N) emissions from agriculture, industry and traffic. These emissions contribute to high ammonia (NH_3), nitrogen oxide (NO_X) and particle matter ($PM_{2.5}$) concentrations in the air (Wu et al., 2016; Liu et al., 2019), and to eutrophication of surface waters and groundwater (Han et al., 2016; Yu et al., 2019; Zhang et al., 2021a). A significant fraction of the N losses to air and water bodies originates from the livestock production-consumption chain, which accounted for more than 50% of food system N losses in 2010 (Gu et al., 2015; Bai et al., 2018).

The increasing decoupling of crop production and livestock production at farm level is one of the main reasons for the current high N losses from livestock production systems (Jin et al., 2021a). Other studies reported that poor manure storage and application contribute to high N losses from livestock production systems (Chadwick et al., 2015). Improved technology for different sections of the manure management chain (Chadwick et al., 2015) and incentives for recoupling of crop and livestock production systems have been proposed to reduce N losses (Bluemling and Wang, 2018; Fan et al., 2018; Zhang et al., 2019). However, the geographic segregation of crop and livestock production systems and the concentration of livestock production in few regions have received less attention as such.

Livestock farms are unevenly distributed in China relative to the area of cropland. Uneven distribution of livestock was defined in terms of the ratio of livestock manure N excretion and the maximal manure N application to cropland at regional level (see below). Similarly, human populations are also unevenly distributed, with 94% located on 43% of the total land area, i.e., at the right side of the so-called Huhuanyong Line (Hu Line) (Wang et al., 2019). Further, livestock density has greatly increased in the eastern provinces during 1980 to 2010, i.e., in those areas where most people are living (Bai et al., 2018). However, concentration of livestock production usually means that the manure nutrient excretion exceeds the nutrient uptake capacity of the nearby cropland (Gerber et al., 2005; Wei et al., 2016; Uwizeye et al., 2020).

Transportation of manure to far-away cropland is costly, due to its high-water content (low dry matter content), and hence unprocessed manures are rarely transported over large distances. As a result, manure nutrients are often not efficiently utilized in China, also because a lack of effective regulations related to the low-emission collection, storage and application of the manures (Chadwick et al., 2020). Instead, manure is over-applied to cropland, accumulating in evaporation lagoons, or dumped into the environment with or without prior treatment. These practices are conducive to large gaseous carbon and nitrogen emissions to air (including NH_3, N_2O, CH_4) and large nutrient (including N, phosphorus, potassium and other nutrients) losses to watercourses (Chadwick et al., 2015; Bai et al., 2018; 2021). An uneven distribution of livestock production, especially a high livestock density near urban areas, may lead to a high proportion of the population exposed to these pollutants in the air and in water courses.

The demand for livestock products will increase in China in the following decades (Wang et al., 2018; Lassaletta et al., 2019; Ma et al., 2019), and it is important to know where the additional livestock should be raised. It has been suggested that a geographical-explicit recoupling of crop and livestock production systems minimizes the risk of high nutrient losses, creates a situation with a better and more uniform environmental quality, and keeps livestock farms at safe distances to residential areas (Hu et al., 2020; Jin et al., 2021a, b). This requires spatial planning of livestock production across and within regions. As a co-benefit of livestock spatial planning, the recycling and utilization of phosphorus and other nutrients, as well as organic matter can be made more efficient (Bradford et al., 2008; Reijnders, 2014). However, little quantitative information is available yet about the impacts of an improved spatial distribution of livestock farms. The objectives of our study are i) to analyze the driving forces and impacts of spatial distributions of livestock production in China for the period 1990 to 2012; and ii) to explore the possible impacts of a spatially more even distribution of livestock production on N use and losses in 2050. We focus on N, because of the severity of the N pollution in China, and because of brevity reason. The NUtrient flows in Food chains, Environment and Resources use (NUFER) model was adjusted and a multi-criteria based livestock spatial planning concept was developed for the purpose of this study.

7.2 Material and Methods

The NUFER (NUtrient flows in Food chains, Environment and Resources use) model was used to quantify the environmental impacts of different spatial distributions of crop and livestock production at county level in China (Wang et al., 2018). NUFER includes data of 2300 counties (out of 2850 counties), which accounted for 94% of the total crop production and for 98% of the total livestock production in China in 2012 (Jin et al., 2020b, NBSC, 2021; Zhao et al., 2021). NUFER is a food system model, it consists of four compartments: crop production, animal production, food processing and food consumption. The crop and animal production compartments include 18 crop types and 11 animal categories. Nutrients are entering the food system via synthetic fertilizer, biological N_2 fixation, atmospheric deposition and food and feed import. Nutrients are leaving the system via food export and losses (ammonia emission, nitrous oxide emission, denitrification, nitrate leaching, runoff, discharge of manure to watercourse or landfill). There are many internal nutrient cycling flows in the model, including manure return to cropland and domestic feed production (Wang et al., 2018). Data mainly refer to 1990 and 2012 due to the availability of county level data; during this period rapid changes in livestock production have taken place.

NUFER calculates the ammonia emission and N losses to watercourse from the whole food production and consumption chain. These results allow us to quantify the relationship between livestock density and N losses to air and water courses at county level. NUFER also provides information about fertilizer N use and total N flows per county; the N surplus includes all N losses to the environment and possible changes of soil N sequestration. For more information see Wang et al (2018).

7.2.1 Geographic distribution of livestock production

The geographic distribution of livestock production was characterized by the livestock density per county and the ratio of manure N excretion and crop N uptake per county. County level animal numbers were derived from Wang et al (2018), which originally came from county, city and provincial level statistical data (census data). Numbers of animals of different livestock categories were converted to livestock units (LSU); one LSU equals to a 500 kg dairy cow (Liu et al., 2017).

Livestock density was calculated below:

$$\text{Livestock density} = \text{LSU} \div \text{Agricultural land area} \tag{1}$$

Where, Livestock density is livestock density per county, in LSU ha^{-1}; LSU is the total livestock unit per county; Agricultural land area, is the total agricultural land area per county, in ha^{-1}.

The geographic (dis)connection between crop and livestock production was characterized by combinations of manure N excretion, synthetic fertilizer N input and crop N uptake per county. Two ratios have been used: i) ratio of manure N excretion and harvested crop N uptake; ii) ratio of synthetic fertilizer N input and harvested crop N uptake. Data were derived from Wang et al (2018), which include county level data for 1990 and 2012. For comparison, we used also data from United States for 1990 and 2012; these were reproduced from the Net Anthropogenic Nitrogen Inputs (NANI) toolbox (Swaney et al., 2018).

7.2.2 Panel model analysis

Two contrasting years of county level data allowed us to estimate changes over time in the relations between (i) livestock population and (ii) human population, GDP, availability of feed, accessibility to feed, and water pollution sensitivity, using panel model analysis. The fixed effect panel model is a frequently-used method in panel analysis (Jin et al., 2020a). It controls for all time-invariant differences between the individual samples, so the estimated coefficients are not biased due to the omission of a time invariant feature. If an unobserved variable does not change overtime, any change of the dependent variables must be attributed to an effect other than the fixed characteristic. Therefore, the fixed effect panel model is chosen to estimate the driving forces of uneven distribution of livestock production while controlling for location (Jin et al., 2020a). In total, 3488 pairs of data have been used; this is less than the potential number because names of some counties were changed between 1990 and 2020 and data of these counties could not be matched therefore. The following variables were used in the panel analysis. Human population per county (https://data.cnki.net/Yearbook/Navi?type=type&code=A) was derived from Statistics China; GDP per county (https://data.cnki.net/Yearbook/Navi?type=type&code=A) was derived also from Statistics China; Availability of feed refers to total maize and soybean production per county and calculated by NUFER; Accessibility to feed refers to the road length per county and was derived from the website https://www.webmap.cn/commres.do?method=result100W; Water pollution sensitivity reflects the total length of surface rivers per county, and was derived from the website https://www.resdc.cn/data.aspx?DATAID=226.

7.2.3 Estimations of N requirement in crop production

Total N requirement in crop production per county was calculated as follows:

$$\mathrm{Ic}_{\mathrm{N\ requirement}} = \sum_{i=1}^{n}\left[\left(\mathrm{Oc}_{\mathrm{Main}_{\mathrm{product},i}} + \mathrm{Oc}_{\mathrm{Straw},i}\right)\right] * \mathrm{Uf(i)} + \mathrm{Oc}_{\mathrm{Managed\ grass}} * \mathrm{Uf} \quad (2)$$

Where, $\mathrm{Ic}_{\mathrm{N\ requirement}}$ is the N requirement per county for crop production under balanced fertilization criteria, i.e., N application is equal to crop N demand; $\mathrm{Oc}_{\mathrm{Main}_{\mathrm{product},i}}$ and $\mathrm{Oc}_{\mathrm{Straw},i}$ are the amounts of N in the main crop product and in straw at harvest per county, respectively, in kg N; $\mathrm{Oc}_{\mathrm{Managed\ grass}}$ is the amount of N in harvested grass from managed grassland per county, in kg N. These values were corrected by the uptake factors (Uf_i) of each crop species under balanced fertilization (Table S1).

We assumed that there is transport of manure within counties and no transport of manure between counties, because of the high cost of long-distance transport. Further, we assumed significant improvements of nutrient management in the whole food system by 2050, including a reduction of NH_3 emissions from the manure management chain by 60%, a ban on the discharge of manure to watercourses and to landfills, and enhanced collection, sanitation and utilization of N in food waste and human excreta (Jin et al., 2020b).

The N balance was calculated per county as follows:

$$Ic_{\text{manure N surplus}} = Ic_{\text{N requirement}} - Ic_{\text{deposition}} - Ic_{\text{BNF}} - Ic_{\text{irrigation}} - Ic_{\text{straw to field}} - Ic_{\text{manure to field}} - Ic_{\text{food}_{\text{byproduct}}} - Ic_{\text{human}_{\text{manure}}} - Ic_{\text{soil mineralization}}$$
(3)

Where, $Ic_{\text{N surplus}}$ is the difference between N demand and N supply per county (excluding the synthetic N fertilizer), defined as manure N surplus; if $Ic_{\text{N surplus}}$ has a negative value it means there is oversupply of N, if it has a positive value it means there is a relative shortage of N. Then, all the negative values were added together, to evaluate the cumulative effects. $Ic_{\text{deposition}}$, Ic_{BNF} and $Ic_{\text{irrigation}}$ are the N inputs via atmospheric N deposition, biological N_2 fixation (BNF) and irrigation water per county. $Ic_{\text{straw to field}}$, $Ic_{\text{manure to field}}$, $Ic_{\text{food}_{\text{byproduct}}}$ and $Ic_{\text{human\_manure}}$ are N inputs via organic resources per county, i.e., crop straw, animal manure, food and kitchen waste and human excreta. $Ic_{\text{soil mineralization}}$ is the N supply from net soil mineralization.

7.2.4 Scenarios for 2050

Business as usual (BAU) scenario. This scenario follows the SSP2 storyline towards 2050 (Zhao et al., 2021). China's total demand for agricultural products, including food, feed, and biofuel, is projected to increase by 25% in per-capita calorie demand in 2050 relative to 2010 in the BAU scenario. Per-capita demand of animal sourced calories is projected to increase almost twice as fast as for total calories compared to 2010. The total projected food demand and production in China are listed in Table S1. The increase in food production by 2050 was evenly allocated to each county according to their shares to the total in 2012.

South to north transfer of pigs scenario. This scenario (SNT) specifically deals with pig production, which is by far the main livestock species in China. It follows the BAU scenario, however, with a different geographic distribution of pig production and includes a ban on the discharge of manure to watercourses, following the Ten Water Protection Action (Han et al., 2016). The Ministry of Agricultural and Rural Affairs divided provinces into four zones in 2017 according to their vulnerability to surface water pollutions and their potential for pig production, namely (i) potential development (PD) zones, (ii) key development (KD) zones, (iii) moderate develop (MD) zones, and (iv) constraint development (CD) zones (Bai et al., 2019a). We assumed that the annual increase of pig production in the PD zone was two times higher than that in the KD zone, and four times higher than that in the MD zone. There was no increase of pig production in the CD zone. These assumptions allowed us to calculate the total pig production for different pig production zones in 2050 (See supplementary information).

Integrated technology improvement scenario. This scenario (ITI) was developed on the BAU scenario, but with significant technological improvements in nutrient management throughout the whole food system. The technologies were specific for three sectors, i.e., crop production, livestock manure management chain, and food waste and human excreta. The technologies aimed to reduce ammonia emission, and to increase the recycling of manure and human excreta, and food wastes. Details about the key technologies and potential effects on N recycling and ammonia emission reduction are presented in supplementary information.

Spatial planning under the NH_3 emission intensity criteria scenario. This scenario (SP) was developed on the ITI scenario, but with comprehensive spatial planning of livestock production (Fig 7.5). Two contrasting variants were developed: i) to reduce the exposure of most people to below the United Nation threshold for the NH_3 emission intensity (SP-NH_3); ii) to optimize the coupling of crop-livestock production in terms of N requirement and supply (SP-CLI). The detail descriptions of the procedures of livestock spatial planning under the two different variants are described in supplementary information.

7.3 Results

7.3.1 Unevenly distributed livestock production

Livestock population increased by 80% between 1990 and 2012, from 240 to 430 million in terms of standard livestock units (LSU). A standard livestock unit equals a 500 kg dairy cow (Liu et al., 2017). About 98% of the increase occurred at the right-hand side of Hu Line (Fig 7.1a, c). Main livestock production counties (i.e., counties with more than 320 thousands LSU), contributed 9.2% to the total LSU in 1990 and to 55% in 2012 (Fig 7.1a, c; S1). The geographic connection between crop and livestock production greatly decreased between 1990 and 2012, as reflected by the rapid increase of livestock density and the increasing ratio of annual manure N excretion to annual crop N uptake in some counties (Fig 7.1b, d; S2). This indicates that the manure N produced could not be utilized by the crops within these counties. Over 37% of the counties at the right-hand side of the Hu Line had a livestock density > 4.0 LSU ha^{-1} in 2012 (Fig 7.1a, c; S1). For comparison, Denmark regulated the maximum livestock density at 2.0 LSU ha^{-1} so as to limit nutrient pollution by livestock production (Willems et al., 2016). A livestock density of ≤ 2 LSU ha^{-1} is a rough indication that the produced manure nutrients can be absorbed by the growing crops in that area, depending also on soil and climatic conditions and management.

Counties with an average livestock density > 4.0 LSU ha^{-1} produced 45% of the total livestock production and applied 40% of the total amount of synthetic N fertilizer used in China in 2012 (Fig S1), showing that counties with high livestock density also had a high fertilizer N use. Most of these counties had a fertilizer N input to crop N uptake ratio larger than 2.0 (Fig S2), which implicitly means that manure N was not effectively substituting synthetic N fertilizer.

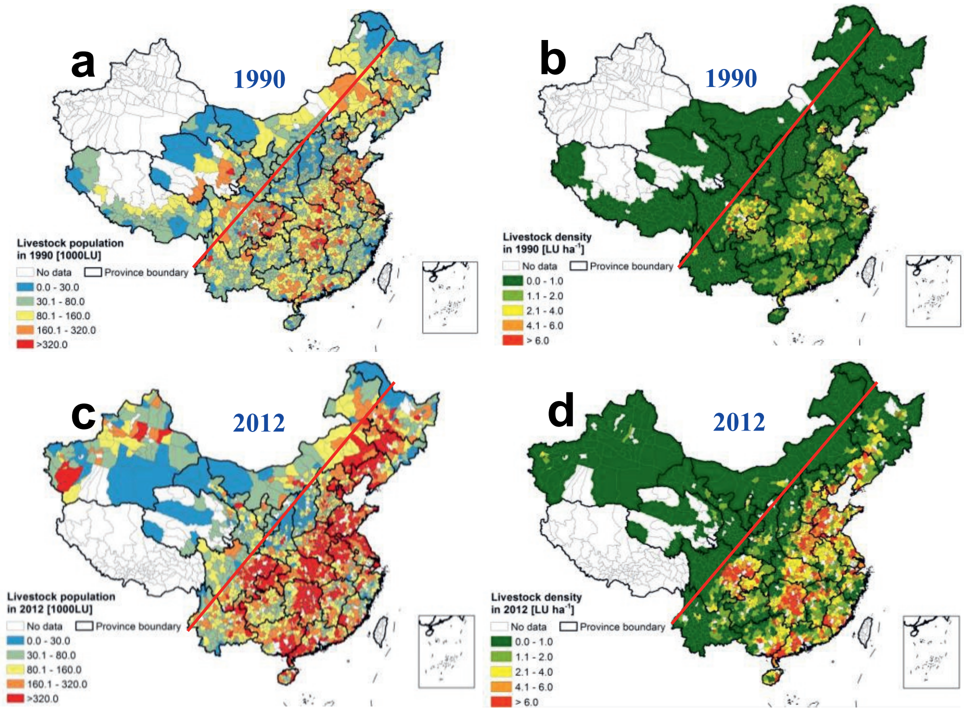

Figure 7.1. Geographic distribution of livestock production in terms of livestock units and livestock density (expressed as livestock units per hectare of agricultural land) at county level in China in 1990 and 2012. *Note, there are 34 provinces and 2850 counties in China.*

The concentration of livestock production in regions is not unique to China; it has been observed in many developed and emerging countries, including the United States of America (USA) (Fig S3). The ratio of annual manure N excretion and annual crop N uptake was > 2.0 in 11% of the 3100 counties in the USA in 1992 and in 22% of the counties in 2012. The contribution of these counties to the total manure N production has increased from 21% to 32% between 1992 and 2012 (Fig S3; Swaney et al., 2018). Though USA and China have roughly similar areas of land and cropland, the numbers of humans and livestock were much larger in China, and hence the concentrations of humans and livestock in specific regions are much larger in China than USA (Fig S2-3). The ratio of synthetic fertilizer N use to crop N uptake has a different geo-spatial trend with the ratio of manure N

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excretion to crop N uptake in USA than in China. This suggests that more manure N was recycled effectively and substituted synthetic fertilizer N in USA compared to China. This has resulted in a relatively low fertilizer N application rate in livestock-dense regions in USA compared to that in China in 2012 (Fig S3).

7.3.2 Driving forces of the concentration of livestock production

The location of livestock farms is influenced by several factors, such as vicinity of markets, traffic infrastructure, availability of animal feed, availability of animal health and epidemic prevention services, and slaughter and milk processing services (Gerber et al., 2005; Delgado et al., 2008; Steinfeld et al., 2013; Błażejczyk-Majka and Radosław, 2015). Concentration of livestock production near markets contributes to improving livestock production efficiency, lowering production costs and shortening the delivery distances (Drabenstott and Mitchell, 1999; Donham, 2000). Our panel data analysis showed that livestock production of a county was significantly and positively correlated with human population and gross domestic production (GDP) value, suggesting that the thickness of local demand played an important role in shaping the geo-spatial patterns of livestock production during the period 1990-2012 (Table 7.1). River intensity had significant positive correlation with livestock density in the no-fixed effect model (Table 7.1). Feed availability (measured by the production of maize and soybean of a county) and feed accessibility (measured by the road length of a county) were also not significantly related to livestock density in the fixed effect model analysis (Table 7.1).

Table 7.1. Regression coefficients of the determinants of livestock populations at county level in China according to two analyses.

Dependent variable:	(1) No fixed effect	(2) Fixed effect
Human population	0.48***	0.29***
GDP	0.10***	0.11***
River intensity	0.099***	-
Crop N uptake	0.20***	0.22***
Feed availability	0.0097	0.082*
Feed accessibility	-0.0009	0.044
Constant	0.89***	1.60**
Observations	3488	3488

*Note: In column 1 we pooled the data for 1990 and 2012, while we add the county fixed effect and estimate a panel model in column 2. Feed availability is defined as the total production of maize and soybean per county, which are the main feed ingredients for modern livestock production; Feed accessibility is defined as the road length per county, including railway and different type of highway. t statistics in parentheses; * $p<0.05$, ** $p<0.01$, *** $p<0.001$. – not applied.*

The governmental Vegetable and Basket Policy has supported the uneven distribution of livestock production indirectly (Li et al., 2008; Wang et al., 2009). Through this policy, mayors of cities established big retail markets, and large vegetable and livestock production farms in peri-urban areas, to ensure a stable supply of food to the citizens. This has led to the concentration of livestock production in the peri-urban areas of almost all big cities (Roelcke et al., 2019). Environmental protection policies have played essentially no role in shaping the geographic distribution of China's livestock production until about 2013. The first environmental regulation on livestock production was issued by the end of 2013 and enacted at the beginning of 2014 (Bai et al., 2019a).

7.3.3 Nitrogen pollution impacts of unevenly distributed livestock production

There were significant positive correlations between livestock density, NH_3 emissions, and N losses to water in counties in 1990 and 2012 (Fig 7.2). There was also a significant positive correlation between livestock density and average synthetic N fertilizer use per ha cropland in counties in 1990 and 2012, which indirectly suggests that little or no synthetic fertilizer N was replaced by animal manure N (Fig 7.2). The slightly higher slope of the linear regression between livestock density and NH_3 emission intensity in 1990 than in 2012 is probably related to a lower crop-livestock integration and to improved animal feeding in 2012 compared to 1990. The uneven distribution of livestock production at county level led to a summed manure surplus of 1.8 Tg N in 1990 and of 4.6 Tg N in 2012 (Fig 7.3b). Manure N surplus is defined here as the difference between total manure N excretion and total crop N demand, corrected for the supply of N by soil, crop residues, wastes, biological N_2 fixation and atmospheric deposition (thus without synthetic fertilizer N), at county level (for more detail see equation 3). Hence, this manure N surplus could not be recycled in cropland and thus was largely lost to air and water bodies (Bai et al., 2018). At the same time, there was a rapid increase in synthetic N fertilizer use, which hindered the effective recycling of livestock manure N further (Bai et al., 2018; Yu et al., 2019).

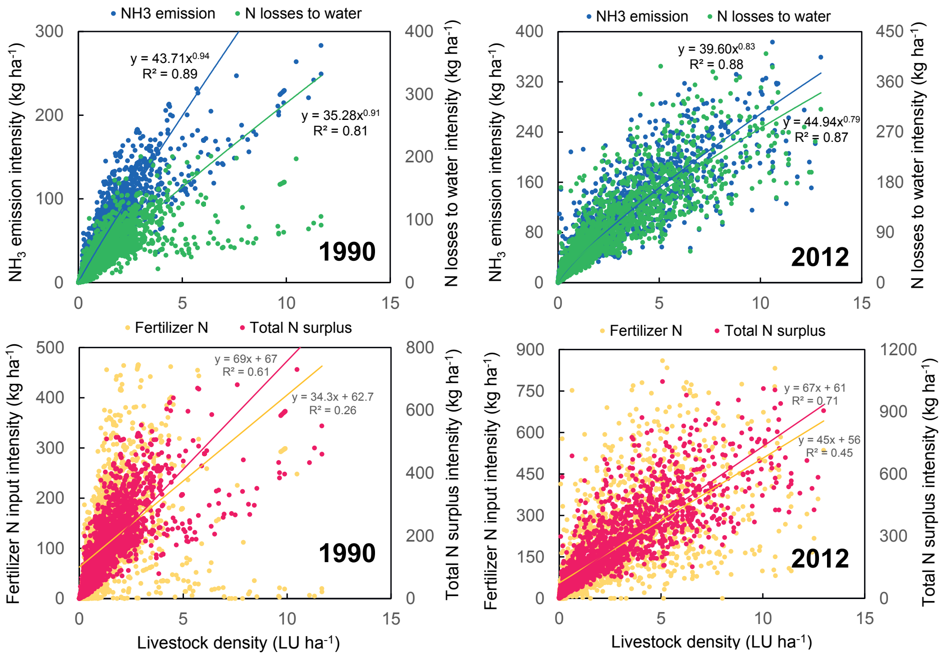

Figure 7.2. Relationships between livestock density and average NH_3 emission intensity, reactive N losses to surface waters, synthetic N fertilizer use and total N surplus (including the N from synthetic fertilizer N) at county level in 1990 (left panel) and 2012 (right panel). *Note: Counties with average livestock density >3 times of the standard deviation of average livestock density (average density was 3.7 LSU per ha in 1990 and 4.3 LSU per ha in 2012) were excluded from the analysis. Note, the calculated NH_3 emissions are not independent from livestock density, since a higher livestock density usually produces more manure N which is vulnerable to NH_3 emissions.*

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Average $PM_{2.5}$ concentrations in the air in a county were more positively correlated with livestock N excretion per county, rather than with fertilizer N input, or with SO_2 and NO_X concentrations in the air (Fig S4). Regions with high $PM_{2.5}$ concentrations in the air correlated to regions with high livestock density. High $PM_{2.5}$ concentrations contribute to respiratory diseases; there were 1.8 million people aged >65 years with premature mortality related to $PM_{2.5}$ exposure in China in 2012 (Li et al., 2018), and over 60% occurred in regions with $PM_{2.5}$ concentration significantly correlated to livestock N excretions (Fig S4).

7.3.4 Environmental impacts of livestock production in 2050 - BAU scenario

Likely, China will continue to suffer from an unevenly distributed livestock production in the near future, especially when the demand of livestock products will increase further. Domestic crop production and livestock production are projected to increase by 20% and 40%, respectively, in terms of protein production between 2012 and 2050 according to a business-as-usual (BAU) scenario (Zhao et al., 2021). As a result, total NH_3 emissions and N losses to water bodies will increase by 12% between 2012 and 2050 (Fig 7.3c-d). The modest increases of N losses to the environment in the BAU scenario reflect the balance between the expected increased production and hence increased N losses, and the expected impacts of policy and measures to decrease these losses (e.g., Zero Fertilizer Action (MOARA, 2015; FAO, 2021); improved crop husbandry practices, improved irrigation and nutrient management (Chen et al., 2014; Cui et al., 2018) and the replacement of fertilizer N by manure N (Chadwick et al., 2020). The policy and measures together will reduce the average N losses per unit of crop product and especially per unit of animal product produced.

Around 26% of the number of counties will have a livestock density > 4.0 LSU ha^{-1} and these counties will accommodate 58% of total LSU in the BAU scenario in 2050 (Fig 7.4a; S1). This will result in a summed manure N surplus of 11 Tg; a surplus that cannot be recycled to cropland in an economically feasible and environmentally sound manner (Fig 7.3b). This surplus may have to be treated to N_2 via nitrification and denitrification process technologies or treated in desired manure N products, which would allow transport between counties in economic feasible manner. The mean NH_3 emission intensity will exceed the suggested threshold of 31 kg NH_3 ha^{-1} in 56% of the number of counties in BAU; the NH_3 emission intensity threshold is one of the criteria for achieving SDG2 (Sachs et al., 2020).

Around 59% of China's population will live in regions with NH_3 emission intensity higher than 31 kg NH_3 ha^{-1} in BAU in 2050 (Fig 7.3e-f).

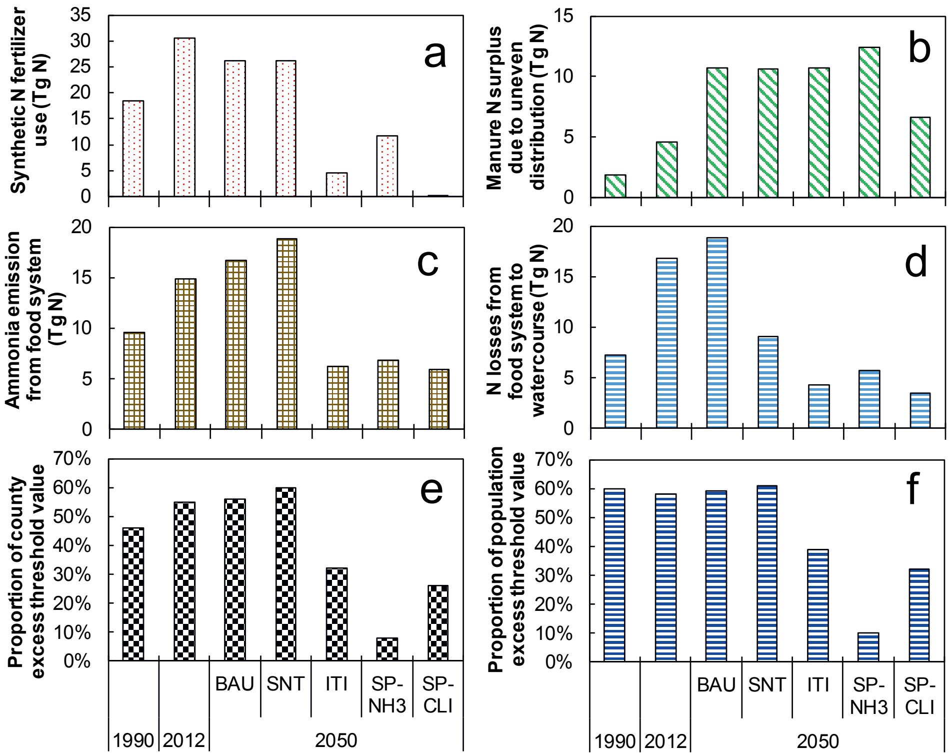

Figure 7.3. Total synthetic N fertilizer use (a), cumulative manure N surplus (expressed in N) due to uneven distribution of crop and livestock production (b), total NH_3 emissions (c) and total N losses to waters from the whole food system to water bodies (d), the percentage of counties exceeding the UN recommended NH_3 emission intensity threshold value (31 kg NH_3-N ha^{-1} agricultural land) (e), and the percentage of people living in these counties (f) in 1990, 2012 and for four different scenarios in 2050.

Note: BAU is business as usual scenario which follows the Shared Socio-economic Pathway 2 storyline; SNT, is south-to-north transfer of pig production scenario; ITI is the integrated technology improvement scenario; SP-NH_3 is spatial planning scenario using the NH_3 emission intensity criterion; SP-CLI is the spatial planning scenario using the crop-livestock integration criterion. The NH_3 emission threshold value is one of the criteria for achieving SDG2 - for sustainable and resilient food production *systems (Sachs et al., 2020). Manure N surplus was defined as the difference between total manure N excretion and total crop N demand, corrected for the supply of N by soil, crop residues, wastes, biological N2 fixation and atmospheric deposition (thus without synthetic fertilizer N), at county level (see equation 3).*

7.3.5 Impacts of pig reallocation and improved technologies - SNT and ITI scenarios

The south-to-north-transfer of pigs (SNT) scenario follows the pig reallocation policy of the national government as proposed in 2017; it will reduce N losses to water bodies by 52% when compared to BAU due to the transfer of pigs from the water courses-dense south to the semi-arid north. However, NH_3 emissions will increase by 13% in the SNT scenario compared to BAU, in part due to a higher manure N recycling, since discharge of manure from intensive livestock farms to watercourses has been banned (Fig 7.3; S5). Much of the increase of the NH_3 emissions will occur in northern China, where air pollution is most serious (Fig 7.4b, e). Thus, the SNT scenario leads to pollution swapping from south to north, and to swapping water pollution in the south to air pollution in the north (Bai et al., 2019ab).

In the scenario 'Integrated technology improvement' (ITI), synthetic fertilizer N use, NH_3 emissions and N losses to water will reduce by 82%, 63% and 77%, respectively, compared to the BAU scenario (Fig 7.3a-d). Evidently, the improved recycling of livestock and human excreta, food waste and crop residues contributed to the substitution of synthetic fertilizers, and reduced N losses greatly (Jin et al., 2020b). However, a total of 11 Tg of manure N could not be effectively recycled due to the geographic separation of crop and livestock production systems in the ITI scenario. In addition, still 60% of the population will live in counties with NH_3 emission intensity higher than the threshold value of 31 kg NH_3 ha^{-1} yr^{-1} (Fig 7.3f).

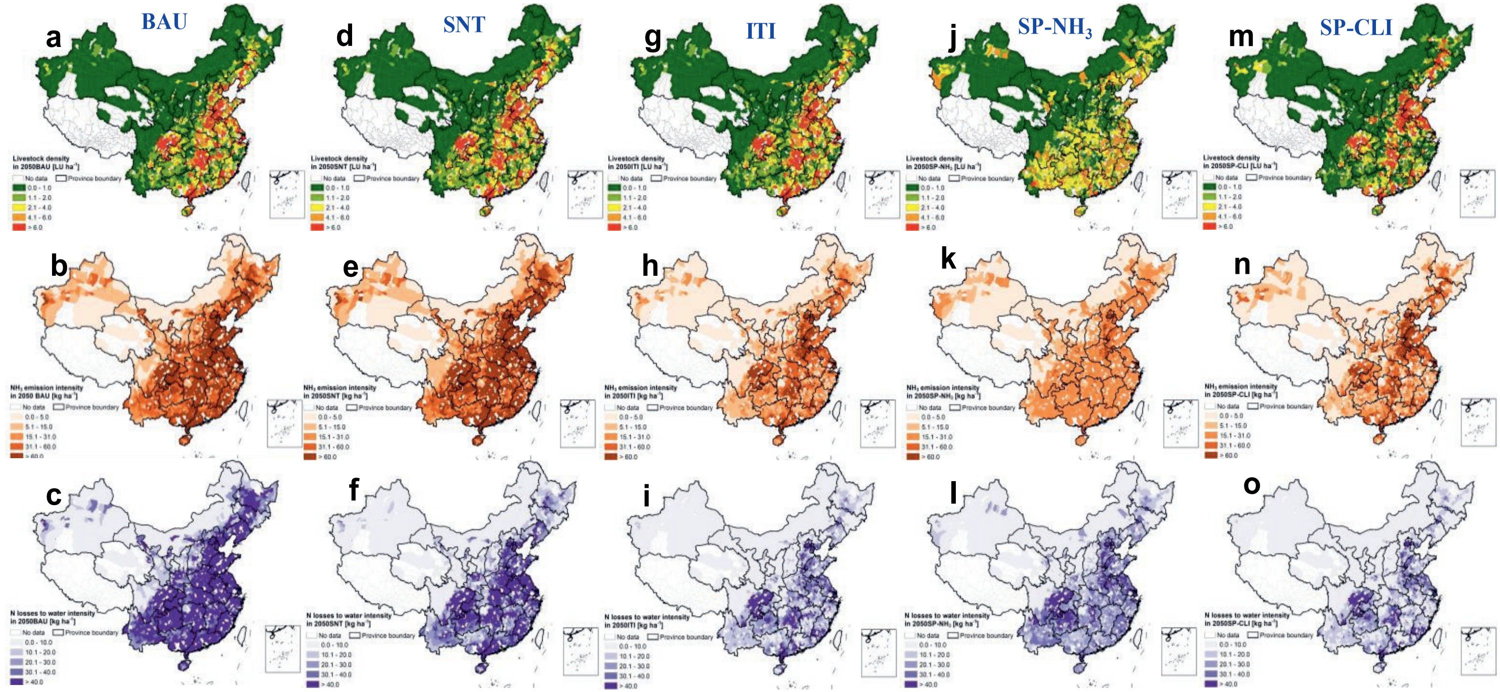

Figure 7.4. Livestock density, average ammonia emission intensity and average N losses to water bodies in 2050 for five scenarios: BAU (a-c), SNT (d-f), ITI (g-i), SP-NH_3 (j-l) and SP-CLI (m-o). *Note: For the full name of each scenario see Fig 7.3.*

7.3.6 Impacts of spatial planning - SP-NH_3 and SP-CLI scenarios

Based on the optimized spatial planning in the NH_3 emission intensity criteria (SP-NH_3) scenario, which was developed on top of the ITI scenario, we estimate that some 10 billion animals need to be reallocated to be able to meet a lower NH_3 emission intensity for most of populations in China (Fig S6). This number is equivalent to 344 million LSU, and includes 484 million pigs, 10 million dairy cattle, 30 million beef cattle, 9,300 million chicken, 146 million sheep and goat, and 1.5 million mules and donkeys (Fig S6). The number accounts for 58% of the domestic livestock population in the SP-NH_3 scenario, and is equal to the current livestock populations in USA and Russian Federation combined (FAO, 2021). Livestock need to be transferred mainly from southern and eastern China to north and southwest China (Fig S6). About 15% of the livestock need to be reallocated within provincial borders, mainly for shortening the transportation distance to ensure the stability of food supply and reduce transportation cost (Fig S7).

In the SP-NH_3 scenario, total NH_3 emissions to air and N losses to water bodies will decrease by 60 to 70% compared to the BAU scenario. However, the SP-NH_3 scenario had slightly higher total NH_3 emissions to air and total N losses to water bodies than the ITI scenario (Fig 7.3c-d). Yet, the SP-NH_3 scenario decreased the proportion of the people in counties with exposure to ammonia emissions higher than the threshold value, when compared to BAU and ITI scenarios (Fig 7.3e-f). Regions exceeding the ammonia emission intensity threshold value were mainly found in semi-arid intensive cropping areas, i.e., mainly in the North China Plain (Fig 7.4k). Manure N surplus and synthetic N fertilizer use were higher in the SP-NH_3 than in the ITI scenario (Fig 7.3a-b). This is in contrast with the objective to decrease the manure N surplus and fertilizer N use through improved manure N recycling. Evidently, there is a tradeoff here between keeping the NH_3 emission intensity below the threshold value and bringing livestock production more closely to cropland.

Spatial planning following crop-livestock integration criteria (SP-CLI) which was also developed on top of the ITI scenario, led to the need to reallocate 5.0 billion animals or 177 million LSU, including 170 million pigs, 6.5 million dairy cattle, 31 million beef cattle, 4,323 million poultry, 146 million sheep and goat, and 2.6 million mules and donkeys (Fig S8). The number of reallocated animals will be 50% less than in the SP-NH_3 scenario, and most of the animals had to be reallocated over relatively short distances in SP-CLI (Fig S7). The SP-CLI scenario resulted in 13% less NH_3 emission, 40% less N losses to water bodies, 99% less N fertilizer use and 47% less manure N wastage compared to the SP-NH_3 scenario (Fig 7.3a-d). However, still 6.6 Tg of manure N was not recycled, because the local manure N supply exceeded the local N demand by the cropland, which could be reduced through import more livestock products from global market. The SP-CLI scenario led to a different geo-reallocation of livestock production than the SP-NH_3 scenario (Fig S9). The SP-CLI scenario showed a stronger performance in reducing total N losses, but a weaker performance in reducing the percentage of the population exposed to a high NH_3 emission intensity compared to the SP-NH_3 scenario (Fig 7.3f). Evidently, there was a tradeoff between keeping the NH_3 emission intensity below the threshold value and reducing total N losses.

7.4 Discussion

7.4.1 Instruments to handle regional manure surpluses

China is the largest pork, poultry and mutton producer in the world currently (FAO, 2021). The increase in livestock production since the 1990s has been unprecedented (Bai et al., 2018). Concentrations of livestock production are found in several regions of the world, mostly in areas with concentrations of humans, i.e., near urban areas (Hou et al., 2021 and references therein), but there are also concentrations of livestock amidst extensive areas of cropland, i.e., in large confined animal feeding operations in for example North and Latin America (Franzluebbers et al., 2021). Common to all concentrations of livestock production is the difficulty to recycle manure nutrients cost-effectively in cropland, and the high NH_3 emission intensity (e.g., Liu et al., 2017; Uwizeye et al., 2020).

European countries were among the first to address regional manure N surpluses and air and water pollution resulting from animal manures. The 1991 Nitrates Directive provides a blanket manure application limit and best N management measures across European Union countries. The 1999 Gothenburg Protocol led to the approval of uniform NH_3 emission mitigation practices. However, individual countries may have specific strategies to address regional manure surpluses. Denmark anticipated on a rapid intensification of livestock production and implemented a livestock density limit of 2.0 LSU ha^{-1} already in the 1980s, which has prevented locally high manure N and phosphorus surpluses (Willems et al., 2016; Sommer and Knudsen, 2021). The Netherlands experienced a very rapid intensification of livestock production in the second half of 20^{th} century, and implemented manure production limits and a manure distribution system from the 1980s, but this was essentially too late to prevent large regional surpluses. The regulations have prevented further increases in manure production, and have contributed to an improved utilization of manure N and P (Oenema and Oenema, 2021), but NH_3 emissions are still too high and surface water eutrophication serious (De Pue and Buysse, 2020; Erisman, 2021). The Po area in Italy, Brittany in France, Catalonia in Spain, and Flanders in Belgium also experience high livestock densities and manure surpluses; here, manure treatment is a main strategy to deal with regional manure surpluses (Meers et al., 2020).

Despite all regulatory measures and economic and voluntary incentives, concentrations of livestock remain persistent hotspots of NH_3 emissions and of nutrient losses to groundwater and surface waters. Benefits of a possible reallocation of livestock from West European countries to Central European countries have been explored through simulation modelling only (Van Grinsven et al., 2015; 2018). The Netherlands government is considering a buy-out of intensive livestock farms near nature conservation areas (Erisman, 2021). However, the number of animals considered in these options and plans is very modest compared to the south-to-north-reallocation of pig production in China (Bai et al., 2019ab) and those of the current study (e.g. Fig 7.4).

The question is whether China can address the manure nutrient surpluses and NH_3 emission hotspots effectively without reallocation of livestock production. Evidently, there is large room for technical measures and emission abatement measures as the ITI scenario shows (Fig 7.4). Further improvement can be made through spatial planning as the SP scenarios indicate (Fig 7.4). We did not analyze possible changes in consumption pattern and animal-source food intake, which may have large beneficial effects on the environment indeed (Springmann et al., 2018).

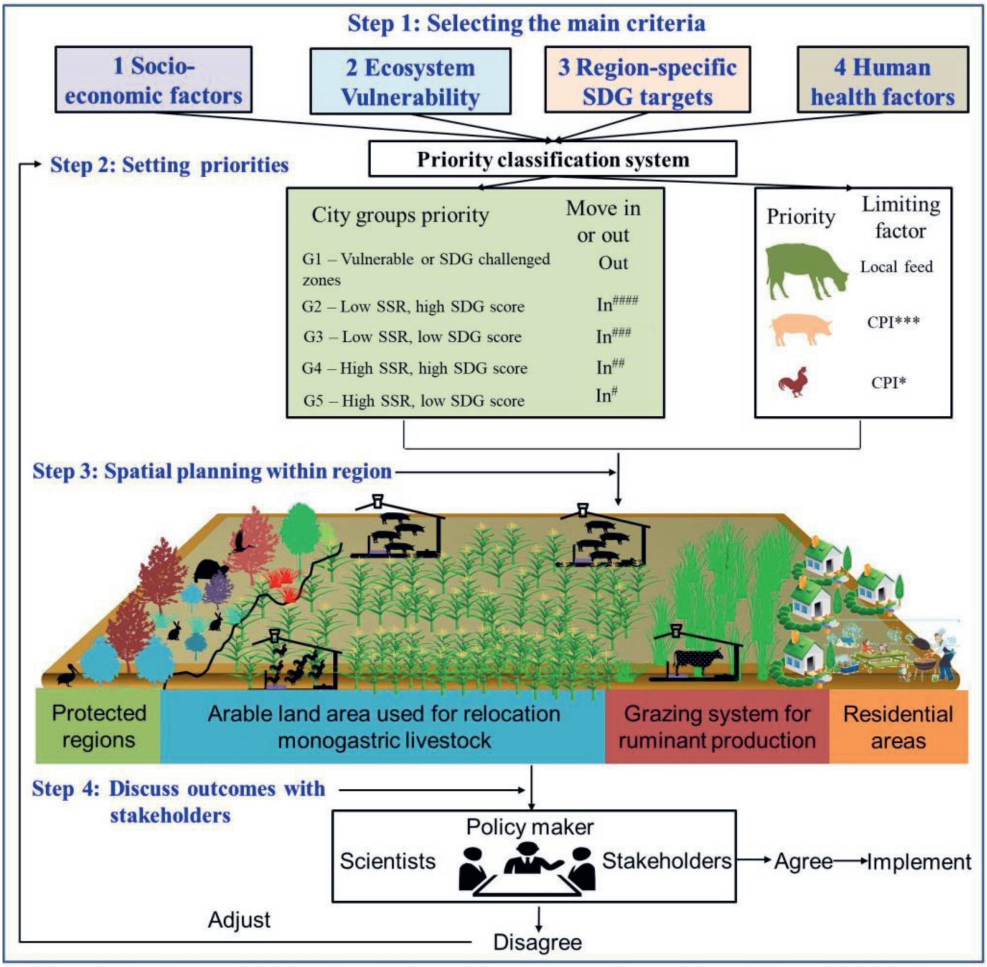

Figure 7.5. Suggested scheme for selecting criteria, setting priorities, and optimizing the spatial planning of livestock production in cities and counties in China. *Note: SSR, self-sufficient rate; CPI, consumer price index; LF, livestock farms; G1-G5, city group from 1 to 5.*

China is a likely country where spatial reallocation of large-scale livestock production can be successful, because of its large livestock concentrations, and its strong central governance. However, spatial planning of livestock production requires clear criteria, principles and procedures (Fig 7.5). China has started a top-down spatial planning of livestock production in 2014 already, and has reallocated millions of pig farms from regions in the southeast with a dense network of watercourses to north China, to reduce water pollution (Bai et al., 2019ab, 2021). As a result, large spatial changes in livestock density have been observed between 2012 and 2017; livestock density was reduced in south and increased in the north (Fig S10). However, the combination of water protection policies and the pig reallocation policy have led to unexpected effects on pork prices which have been enhanced after the outbreak of African Swine Fever in 2018-2019 and the Covid-19 pandemic in 2020-2021 (Bai et al., 2021). As a result, the central government has given up the pig reallocation plan, and has given higher priority at the Vegetable and Basket Policy (Bai et al., 2021). As a result, major new pig farms were built in high NH_3 emission intensity regions and high population density regions recently (Fig S10).

7.4.2 Minimizing trade-offs related to spatial reallocation of livestock production

Spatial planning can be successful in protecting ecosystems, when sound ecological and social-economic principles are applied together (Day, 2008; Foley et al., 2010). The Vegetable and Basket Policy (Li et al., 2008; Wang et al., 2009) may be seen as a one-sided spatial planning aimed at increasing vegetable and meet production and its fast delivery to urban consumers. The south-to-north-reallocation of pig production (Bai et al., 2019ab, 2021) may be seen as a one-sided spatial planning aimed at decreasing surface water pollution. Evidently, the last two spatial planning policies have large side-effects.

Serious trade-offs are also observed in the two spatial planning scenarios of our study. Spatial reallocation of livestock to reduce the exposure of humans to high NH_3 emissions is not effective in enhancing manure N recycling and reducing synthetic N fertilizer use in China. In contrast, spatial reallocation of livestock aimed at enhancing manure N recycling and reducing synthetic N fertilizer use is not effective in reducing the exposure of humans to high NH_3 emissions. These results are caused in part by the fact that intensive crop production systems in China have already a high NH_3 emission intensity due to the liberal use of urea fertilizers (Fig 7.3).

Pollution swapping through reallocation of livestock production can be minimized through the implementation of strict emission mitigation measures in crop and livestock production systems. For example, urea-based N fertilizers have to be deeply placed/injected into the soil or have to be replaced by nitrate-based fertilizers to reduce NH_3 emissions (Ti et al., 2019; Zhang et al., 2020; Sha et al., 2021). Similarly, low-protein animal feeding, and low-emission manure storage and application to cropland may greatly decrease NH_3 emissions associated with manure application. Implementing low-emission technology and balanced fertilization before any spatial planning action greatly reduces the risk of pollution swapping but decreases also the benefits (and costs) of spatial planning actions. An optimum balance needs to be found between the cost and benefits of low-emission technology and the costs and benefits of spatial planning actions, without compromising agricultural productivity. People in the rural areas and peri-urban areas will benefit ultimately from cleaner air and waters because of the strong reduction of ammonia emission and related health costs, but we cannot exclude that food production costs will increase (Zhang et al., 2020).

7.4.3 Sensitivities and implications of livestock spatial planning

Spatial planning of livestock production under NH_3 emission control criteria was highly sensitive to the selection of the threshold value. The stricter the threshold value the higher the number of livestock that need to be reallocated within China. The threshold value was set at 31 kg NH_3 ha^{-1} – the criteria for achieving SDG2 – for sustainable and resilient food production systems (Sachs et al., 2020). The criteria will be stricter in many European countries by 2030, because the National Emission Ceiling Directive has put regulations on total ammonia emission per country (Giannakis et al., 2019). China will have to relocate more animals when the threshold value will decrease to below 31 kg NH_3 ha^{-1}, or will have to further cut down domestic livestock production either through diet change to less meat consumption or through import of more livestock products from the global market. Spatial planning of livestock production under crop-livestock integration criteria was sensitive to the bio-availability of N from different organic resources. Less animals will need to be reallocated if the bio-availability of N was considered, as there is lower surplus of bioavailable manure N than of total manure N (Fig 7.3b; S5b). Urbanization may also play a role in shaping the reallocation of animals, because urbanization leads to less people in the country side and hence to less people exposed to cropland and livestock production related NH_3 emissions (Wang et al., 2021).

Other possible pollutants, such as heavy metals, veterinary antibiotics and greenhouse gas emissions were not considered in this study, but may have large impacts as well. This suggests that additional indicators and ecosystem threshold values have to be considered in spatial planning (Fig 7.5). Current barriers for proper manure treatment and recycling to cropland need to be fully removed to be able to recouple crop and livestock production system effectively (Zhang et al., 2021b; Tan et al., 2021). The strive for carbon neutrality will also have implications for the optimal locations of production and consumption centers.

Overall, China may have to reallocation 5-10 billion animals to tackle N pollution of air and water caused by concentrations of livestock production. This is an enormous challenge, with huge economic and societal impacts, but will yield improved N pollution control as final result. Policy makers need to carefully select the criteria for spatial planning of livestock production, as the ultimate effects of spatial planning depend on the criteria prioritized.

7.5 Acknowledgements

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CHAPTER 8

General discussion

8.1 Introduction

Improving livestock production is one of the keys to achieving more sustainable food production systems at global scale. Currently, livestock production has wide and profound influences on land use, food-feed competition for cropland (Wirsenius et al., 2010; Alexander et al., 2015), land use change and associated loss of biodiversity (FAO, 2020), air pollution (Erisman, 2021; Gu et al., 2021), greenhouse gas (GHG) emissions and climate change (Herrero et al., 2016; Rivera-Ferre et al., 2016), and eutrophication of surface waters (Jones et al., 2019; Li et al., 2021). Several studies have explored options to improve the performance of different aspects of livestock production systems, including animal breeding, animal feeding, animal housing, and manure management and treatment (e.g., genetic breeding technologies and feed supplements and precise feeding (Connor, 2015; Patience et al., 2015; Gadde et al., 2017; De Verdal et al., 2018), rapid and efficient floor cleaning (Ndegwa et al., 2008; Hou et al., 2015), and continuous closed manure composting technology (Liu et al., 2020)), as well as reducing food waste losses (Springmann et al., 2018). However, quantitative analyses that include up-stream and down-stream effects of livestock production systems in regions that face rapid and large transitions are still scarce (Steinfeld et al., 2006; 2013).

The main objective of my thesis is to increase the understanding of the global livestock transition and its environmental impacts, using China as a case, because of its large livestock production, the rapid changes in livestock production systems, and the availability of data. The studies described in Chapters 2 to 7 of my thesis were each dedicated to a specific theme and research objective. The objective of this final discussion chapter is to highlight, integrate and discuss the main findings in the broader context of the international literature, and to explicate the livestock transition. This chapter starts with the description of main findings of my thesis research (Chapter 8.2). Then follows a discussion of China's unique livestock transition, and its impacts on domestic resources use and environmental pollution (Chapter 8.3). Next, the global impact of China's increasing livestock production is discussed (Chapter 8.4). Thereafter, the importance of whole chain manure management and livestock spatial planning to reduce nitrogen (N) pollutions and over-fertilization are being discussed (Chapters 8.5 8.6 and 8.7). Finally, the main conclusions are presented.

8.2 Main findings

- A novel and systematic method – the cumulative productivity distribution curve - was developed to quantify the concentration of agricultural production in high productivity countries. In addition, two complementary indicators – the functionality of trade and the optimality level of trade - have been defined and quantified, and used to evaluate the impacts of international trade on cropland and livestock productivities and partial fertilizer N and feed N productivities from 1961 to 2017. The new method and associated indicators allow uniform assessments to be made of the impacts of international trade for importing and exporting countries, and may help to set priorities for specific countries and specific products. The new method is relatively simple, transparent and may be easily extended. More applications of the cumulative production curve approach can be envisaged, including in industry and ecology (Chapter 2).
- The NUFER model was modified and further developed to the NUFER-animal model, to analyze N and phosphorus (P) flows, GHG emissions, land use and feed consumption in the soil-feed-livestock-manure management chain. The NUFER-animal model distinguishes six main livestock species and twenty different production systems. Through the NUFER-animal model, with its extended databases, we quantified the impacts of the livestock transition on the environment and on resources use within and outside China, and we explored pathways towards more sustainable livestock production (Chapters 3-7).

- China's livestock transition was characterized as unique in the world, in terms of its driving forces, its size and rate of change, and the enormous environmental and social-economic impacts. Within 30 years, the livestock population tripled, expressed in total livestock standard units (LSU), and the number of LSU in landless industrial systems increased 70-fold. Total animal protein production increased by a factor of 5 (Fig 8.1), and the annual gross economic value by a factor of 60 during this period. The changes have been much larger and quicker in China than in any other country in the world (Chapter 3).
- My analyses indicate that the livestock transition in China has wide global impacts. Animal feed importation has increased 49 times between 1980 and 2010, and this increased import has multiple impacts in both China and feed exporting countries. For example, to meet the increasing demand of just dairy products by 2050, global dairy-related GHG emissions, N losses and the area of land needed for feed production will increase by 35%, 48% and 32% compared to 2010, respectively. Importation of both milk and/or animal feed will have significant environmental effects in exporting countries. I concluded that China needs to improve the domestic milk and feed production efficiencies, at least up to the level of the leading milk producing countries in the world. Through improving dairy production structure and optimizing the spatial distribution pattern of dairy production, a win-win situation will be created for China and the global dairy sector (Chapters 3 and 4).
- The livestock transition greatly contributed to the spatial decoupling of crop and livestock production in China, and thereby also to the increased NH_3 emissions to air and N losses to surface waters during 1980-2010 (Fig 8.1). Animal housing and manure management account for more than 80% of total N losses from the livestock sector, but this large loss has not attracted equally large attention from scientists and policy makers in China yet (Chapters 3 and 5).

- The livestock transition has also greatly contributed to the uneven geographic distribution of crop and livestock production in China. Livestock production has concentrated in and near urban areas, while crop production is mainly in the rural areas. Such geographic segregation of crop and livestock production has greatly limited the opportunities to utilize the carbon and nutrients from animal manure in crop production, to substitute synthetic N and P fertilizers by animal manure, and to reduce N and P losses. I estimated that the amounts of N and P in manure, wastes and residues are larger than the demands of N and P by the crops grown in 30% and 50% of the counties, respectively. A drastic increase in the recycling and utilization of N and P from manure, wastes and residues can only happen following relocation of livestock farms to areas with sufficient cropland (Chapters 3 and 6).
- A novel livestock spatial planning framework has been developed, with multiple criteria. Using this framework, I estimated that relocating 5 billion animals by 2050 could reduce nitrogen emissions by two thirds, and halve the number of people exposed to high ammonia emissions. Relocating 10 billion animals away from southern and eastern China could reduce the exposure of 90% of China's population to high ammonia emissions. Thus, spatial planning can be a powerful policy instrument to tackle nitrogen pollution and exposure of humans to ammonia in the air (Chapter 7).

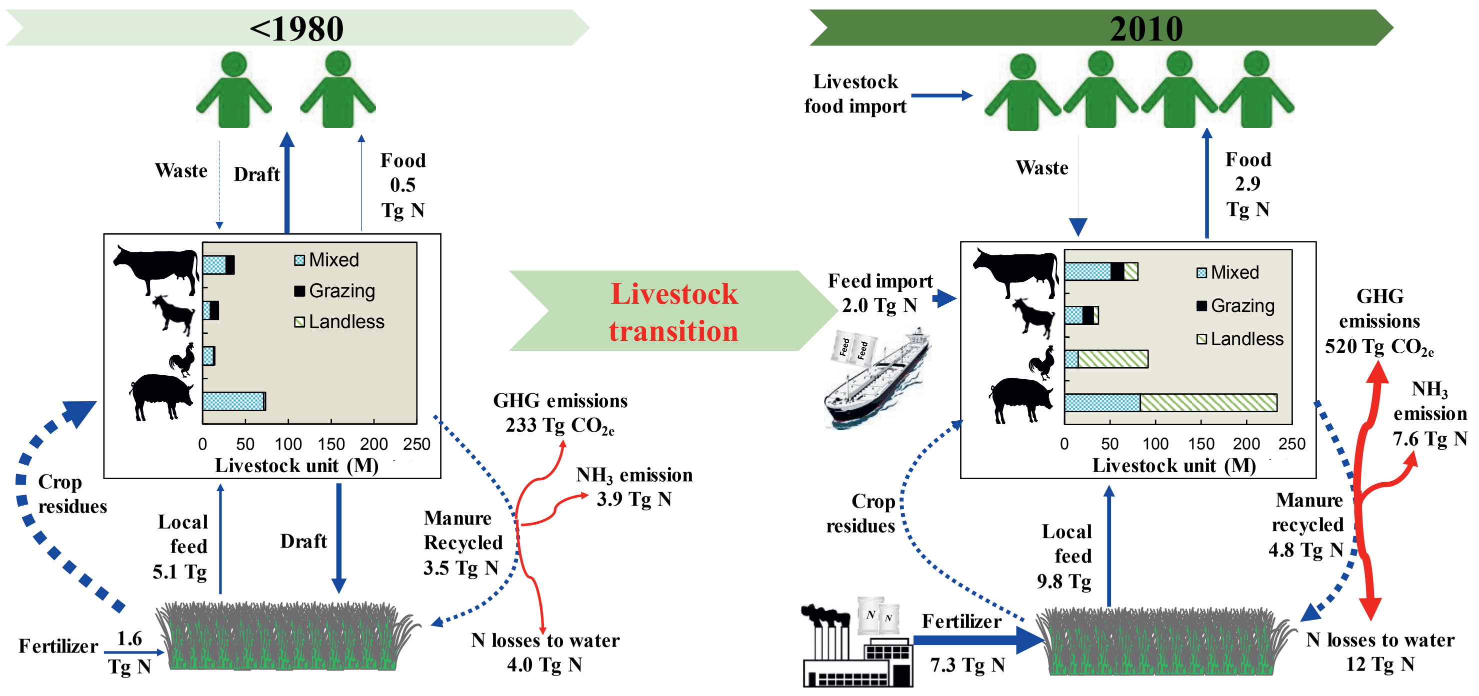

Figure 8.1. Livestock transition in China between 1980 and 2010. The graphs show the main features of crop production (bottom), livestock production (middle), and the consumption of food (top) in 1980 (left) and 2010 (right). Solid arrows represent nitrogen (N) inputs and outputs; the dotted arrows represent nutrient recycling flows. *Modified on the basis of Chapter 3 and Jin et al., 2020.*

8.3 China's unique livestock transition

Livestock functions, production categories, production systems, actors and impacts have greatly changed in China in the past few years, in response to the economic growth and development, changes in governmental policy, and the rapid increases of the human population, human wealth and urbanization. The effects of the changes in livestock production occur both up-stream and down-stream. To quantify these effects, I developed the NUFER-animal model, which is based on the food pyramid NUFER model (Ma et al., 2010). NUFER-animal includes modules for feed production, animal production and manure management. NUFER-animal model can be used to analyze N and P flows, GHG emissions, blue water requirement and land use for six main livestock species and twenty different livestock production systems (Chapters 3-5).

The term 'livestock revolution' was first coined in 1999 to describe the growing demand for livestock products in developing countries, due to population growth, rising incomes, rapid urbanization, and changing diets (Delgado et al., 1999; 2001). The livestock revolution is thought to be mainly 'demand driven' while the earlier and well-known Green Revolution was largely 'supply driven', through the supply of cheap synthetic fertilizers and new crop breeds (Delgado et al., 2001; Sumberg and Thompson, 2012). The concept of livestock revolution was readily accepted and livestock revolutions have been reported for several developing countries, including India (Khan and Bidabadi, 2004), Lao PDR (Millar and Photakoun, 2008), Africa (Mwangi and Omore, 2004), Brazil (Lundström, 2011) and China (Rae, 2008). These reports focused on poverty alleviation, diet nutrition improvement, household changes and trade of agricultural products at local level. There have been also global studies on the implications of the livestock revolution for food security (Hall et al., 2004; Neumann et al., 2010), veterinary development (Steninfeld, 2004), zoonotic coronavirus transmission (Rulli et al., 2021), biomass use (Blummel et al., 2013), grassland management (Delgado, 2005) and environmental conservation

(Paudel, 2006). These studies provide evidence of a global level awareness of the importance of the livestock revolution. However, most of these studies were qualitative, and as a result there is little quantitative cause-effect data and information about the livestock revolution, especially related to the driven forces, pressures, states, impacts and governmental responses.

Chapter 3 presents one of few studies that quantitatively analyzed the livestock transition according to the 'Driver-Pressure-State-Impact-Response' (DPSIR) framework. This framework allowed us to describe China's livestock revolution in a number of unique features. **I) The speed and scale.** Livestock number increased only moderately between 1960 and 1980, but it tripled between 1980 and 2010, while total animal protein production increased by a factor of 5 (Chapter 3). No other countries have achieved such rapid and large-scale transition during the same period (FAO, 2021). **II) Powered by two different engines.** The livestock revolution was not only incentivized by increased food demands (Delgado et al., 2001), but also by governmental incentives in China. Governmental policy was acting as a strong driving force of the livestock transition. This is rather unique as it is commonly perceived that there is one single engine for the livestock revolution (Delgado et al., 2001). Three policies contributed to the livestock revolution directly and indirectly. First, the liberation of markets and the removal of barriers, such as the autonomy right to produce, and the cancellation of meat coupons. Second, economic policies aimed at boosting crop and animal production around big cities, such as the Vegetable Basket program. Third, the loose environmental regulations (Chapter 3). **III) Changing systems.** Within 30 years, the number of LSU in landless industrial systems increased 70-fold, and the proportion of monogastric animals to total LSU increased to 74% between 1980 and 2010 (Chapter 3). In contrast, the number of ruminant animals increased more strongly than that of monogastric animals in USA between 1959 and 1980 (FAO, 2021), and the increase of the cattle population is also stronger than the increase of the pig and poultry populations in Latin American countries during its livestock transition. The contribution of monogastric animals to total animal-source food production is much higher in China than the global average (Liu et al., 2017). **IV) Increasing resources dependence.** China turned 17 million ha of cropland into maize production to address the increasing demand of feed during the past 3 decades. This change is the largest change in land use in the world, and accounted for 30% of the global increase in maize production area between 1961 to 2020 (FAO, 2021). The land use change included conversion of natural wetland in northeast into cropland (Zhang et al., 2010), and the replacement of soybean production by maize production after China joined the World Trade Organization (Liu et al., 2014). China became the world largest importer of animal feed, notably soybean, maize and alfalfa. **V) Increasing environmental impacts.** Total N loss from livestock production systems to watercourses tripled between 1980 to 2010, mainly as a result of direct discharges of manure to watercourses. Discharge of manure to surface waters is forbidden, but there is lack of monitoring, control and enforcement. Emission of NH_3 to the atmosphere from livestock housing and manure storages also increased (Chapters 3 and 5). It has been reported that 62% of the drinking water wells monitored in China were considered 'unhealthy'; the main sources of the nitrates in the wells were fertilizers and livestock manure, as identified through isotopes analyses (Han et al., 2018). **VI) Diminishing multi-functionality of livestock.** The many functions that livestock had in the past have changed into one main function – provide high quality animal-source protein (Fig 8.1). This is in part due to the livestock transition process, and in part due to the rapid development and changes in other relevant sectors during the same period. For example, the industrialization of livestock production has diminished the role of farm animals as 'waste converters and upgraders'. The increased availability of cheap, subsidized fertilizers has reduced the importance of manure as nutrient source and soil amendment. The increasing availability of tractors and other diesel-based vehicles decreased the need of animal draft, and the rapid increase of household income reduced the need of having livestock as buffer against poverty (NBSC, 2021).

A unique feature of China's livestock transition is the spatial decoupling of crop and livestock production systems, which challenges nutrient recycling (Chapters 3 and 6). This feature was not part of the original definition and concept of livestock revolution (Delgado et al., 1999; 2001). China's livestock production has increasingly concentrated around big cities, near slaughter houses and sales markets. The soybean crushing industry is concentrated in coastal areas, and feed companies in coastal and urban areas. Concentration of livestock production and spatial decoupling of crop and livestock production systems have also been observed in many other countries of the world (e.g. Hou et al., 2021).

Livestock on smallholder farms remains to have an important role in recycling wastes and crop residues, which still contribute a great proportion of feed consumption by global livestock production systems (Herrero et al., 2013). Large amounts of crop residue are currently not utilized well in China (Cao et al., 2008), but there is potential to improve the utilization of these crop residues through fermentation processes.

8.4 Global impacts of China's livestock production

The estimated reliance of imported feed protein to total feed protein intake was around 30% in China during the period 2010-2020 (FAO, 2021). In addition, China also imported increasing amounts of milk after the melamine scandal in 2008, and beef and pork after the African Swine Fever outbreak in 2018, which together accounted for 10% of the total domestic animal-source protein supply in 2019 (Bai et al., 2021; FAO, 2021). It is as yet unclear whether the import of animal feed and/or livestock products will increase in the near future, and by how much. Both imported feed and imported livestock products have relatively large environmental footprints as further discussed below.

Chapter 4 systematically evaluated the impacts of increased future dairy consumption and the options for addressing the increased consumption. Both importing all additional feed to support increased domestic dairy production, or importing all additional dairy products to satisfy the increased domestic demand by 2050 appeared not realistic, for various reasons. Importing large amounts of feed or dairy products will make China strongly dependent, and will externalize the environmental footprint of dairy production to exporting countries (Fig 8.2; Zhao et al., 2021). More sustainable solutions for satisfying China's increasing dairy consumption rely on multiple actions, including but not limited to i) closing the yield gaps, i.e., increasing dairy productivity and feed production, to the levels that currently occur in developed countries; and ii) promoting dairy production in medium-size farms, to facilitate crop residue and manure nutrient recycling.

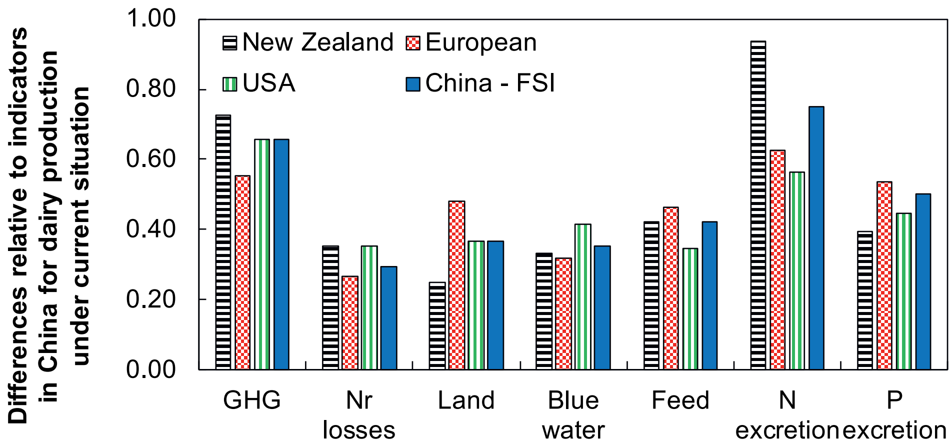

Figure 8.2 Relative indicator values for environmental footprints of dairy production in the three main dairy exporting countries in the world compared to the current situation in China (modified from Table 4.1). The relative indicator values for China-FSI indicate that the performance of dairy production may catch-up to the level of the exporting countries following significant improvements (see Chapter 4). *Note: USA, United States of American; Nr, reactive N; N, nitrogen; P, phosphorus. FSI, farm system improvement scenario.*

Chapter 4 focused on dairy production in China and its main trade partners. The results would be different if the analysis would have been conducted at the global scale, and when the whole agricultural production would have been considered. For example, land expansion for increasing forage production may compete with land for crop production in dairy exporting countries, and may reduce the overall productivity of agricultural land. Evidently, it is complicated to systematically evaluate the effects of food and feed trade on agricultural productivity and agriculture related environmental effects. Few studies quantified resources savings and impacts or air pollution through international trade through a comparison of pair-to-pair differences of commodities between importing and exporting countries (Meng et al., 2016; Liu et al., 2019). However, it is impossible to implement such method for agricultural products, since not all imported agricultural products can be produced in importing countries, for climate, soil and water availability reasons. Hence, we developed a new, novel and systematic method in Chapter 2 to solve this problem. Our results suggest that international trade of food and feed has reduced global cropland use efficiency, but increase global livestock production efficiency, in terms of energy (calorie) production. This analysis has provided new information and insights for the debated role of trade on global cropland productivity (Fader et al., 2011; Fader et al., 2013; Kastner et al., 2014). Chapter 2 also proposed two new indicators, which allowed to indicate the optimality level of trade, for different years and commodities.

8.5 Vulnerability of livestock production to shocks

China's livestock production is vulnerable to external and internal shocks. For example, domestic milk production increased by a factor 10 between 1988 and 2008, but production stagnated after the well-known melamine scandal in 2008 (Chapter 4), because several dairy processing companies imported dairy products from abroad, which were more trusted by consumers. These processing companies and retail found it more profitable to import milk, because purchasing price of imported milk is 50% lower than domestically produced milk, while the retail price of imported milk is 2–4 times higher than domestically produced milk (Zhao et al., 2018). Dairy processing companies and retail advertised imported milk (powder) as high quality and healthy. As a result, the market share of domestic milk strongly decreased, even when the quality of domestic milk has greatly improved (Zhao et al., 2018).

Consumer confidence in domestically produced pork is high, but the pork production sector is vulnerable to disease epidemics, such as the African Swine Fever (ASF), which hit China severely between 2018-2020. The ASF outbreak has reduced pork production by 12 Tg in 2019, equivalent to 22% of the production in 2017 (NBSC, 2021). The ASF outbreak has changed the pork supply and demand at the regional level, especially in Shandong and Henan, where the pork self-sufficiency was reduced by around 60% between 2017 and 2019, and to less than 20% in Zhejiang, Beijing and Shanghai (Bai et al., 2021). The shortage of pork led to an increase in the urban consume price index (CPI) for meat. Urban CPI of all commodities increased by 3%–5% in 2019; around 50% of the increase was contributed by the increases in the price of pork. In response to ASF outbreaks, government immediately ban the feeding of swill (kitchen waste and food residues from restaurants), which were then discharged to the sewage treatment system and surface waters. This has negatively affected water quality in major watercourses alongside the Yangtze River Basin and Peral River Basin (Bai et al., 2021). Several attempts are being made to make the pork sector more robust and less vulnerable to diseases.

8.6 Pathways to more sustainable livestock production

Consumption of animal-source food per capita will likely increase further during next decades (Chapter 3). A recent study suggested that importation of animal feed, dairy products and beef will increase steadily between 2010 and 2050, with significant impacts on land use and GHG emissions in New Zealand, Brazil and Australia (Zhao et al., 2021). Likely, domestic livestock production will also increase; this will challenge China's commitment to become carbon neutral, unless drastic improvements are being implemented. We argued that a new livestock transition/revolution is needed, which should at least contain the following aspects:

i) Finding a balance in farm size. The resilience of livestock production systems to climate change risks and financial risks has decreased during the livestock revolution in China. Also other societal functions of livestock have been lost, but the production function. This loss of multifunctionality of livestock is largely related to the rationalization and industrialization of livestock production. Smallholder farmers and small mixed farming systems are rapidly disappearing and replaced by specialized, large livestock farms (Jin et al., 2021). However, medium-size livestock farms seem to have advantages in feed use, herd management and health and epidemic prevention management conditions (Chapter 3) and may offer the potential to include multifunctionality again to livestock production. The latter is becoming more important recently, following the outbreak of the African Swine Fever, which wiped a big proportion of China's pig production capacity (Bai et al., 2021). Especially smallholder pig farms and backyard farms have been severely hit by the African Swine Fever. In the meantime, pork production companies built tall and large 'hotels' for pigs, which produce hundreds of thousands of fattening pigs per year in remote areas. These industrial-scale pig farms are able to keep African Swine Fever out of their buildings, but have great difficulty to recycle the manure in a proper manner in cropland. Evidently, there is a need to find compromises in farm size and farming structure, for optimal biosecurity, manure recycling and other functions of livestock production.

ii) Spatial planning needed for recoupling crop and livestock production systems. Locations of livestock farms have to be planned strategically away from watercourses, nature conservation areas and densely populated urban area, because of their sensitivity to N pollution, and because of the need to recycle manure to cropland (Chapter 6, Bai et al., 2017; 2018). Chapter 7 showed that spatial planning following crop-livestock integration criteria will strongly reduce total N losses and synthetic N fertilizer use, but this scenario showed a weaker performance in reducing the percentage of the population exposed to a high NH_3 emission intensity compared to the scenario that included strict NH_3 emission criteria. Clearly, there are tradeoffs in livestock spatial planning between different indicators. The study presented in Chapter 7 showed that spatial planning is a powerful tool to address environmental pollution, but was limited to a few indicators only. In the future, also other indicators, such as pollution by heavy metals, veterinary antibiotics and GHG emissions should be evaluated, as well adaptation to climate change effects.

iii) Improve the feed protein supply. Three food system innovations may greatly reduce China's reliance on imports: (1) microbial feed protein production to substitute imported feed protein, (2) vegetation greening and improved fodder production through grassland restoration, to increase roughage production and to reduce the import of roughages and ruminant animal products, and (3) insect based fish production and offshore marine restoration to replace meat consumption (Bai et al., 2020). These measure are technically and economically feasible, and generate high ecosystem benefits. By reducing biodiversity loss and curbing GHG emissions, these changes would also assist in achieving COP-26 targets. Moreover, these changes would put China at the forefront of developing technologies for sustainable protein production, which can then be shared with and transferred to food and feed importing countries (Bai et al., 2020). Smart diet change towards to animal sourced food which stemmed from non-edible plant products could also offset China's great demand for feed protein (Van Zanten et al., 2019; Van Selm et al., 2022).

iv) Improve manure management. Several policy measures have been implemented to improve manure management during recent years, such as a ban on manure discharge and the obligation to conduct manure treatment (anaerobic digestion, composting) on large livestock farms. However, manure management remains a weak aspect of most livestock farms, because of the spatial separation of crop and livestock farms, the high investment and operational costs of proper manure storage, treatment and utilization, and the lack of control and enforcement. There are also several barriers at crop farms that limit the utilization of animal manure (Zhang et al., 2021; 2022). Yet, improving manure management on livestock farms will greatly contribute to several policy objectives, including a decrease in synthetic fertilizer use, and a decrease in the emissions of greenhouse gases and the eutrophication of surface waters.

8.7 Conclusions

The main objective of this thesis was to increase the understanding of the livestock revolution in China on nutrient flows and greenhouse gas emissions, and to explore options for more sustainable crop-livestock production systems by 2050. In order to meet this target, six specific objectives were formulated (Chapter 1). My conclusions regarding the main objective and the five specific objectives are presented below.

The livestock revolution in China was driven both by increasing demand for animal-source food, governmental policies, and economic and technological developments. This livestock revolution has been unique in terms of scale and speed; it has greatly altered the livestock production structure, nutrient cycling, nutrient losses and GHG emissions. Total animal production has greatly increased.

A novel method was developed and used to evaluate the impacts of food and feed trade on global land use efficiency, N use efficiency, livestock productivity and feed N use efficiency. Interestingly, the overall net effects of international trade on productivity and efficiency were different when the traded products were characterized in terms of calories or in terms of protein. The massive international trade of protein-rich soybean has contributed to increased land use efficiency.

The increasing demand of dairy products has increased China's reliance on the global market to fulfill its thirst for milk. However, satisfying the increasing demand for milk continuously through importation from abroad may not be sustainable and feasible in the long term. Instead, improving domestic milk and feed production efficiencies, up to the level achieved by the leading milk producing countries in the world, and improving the dairy production structure seems a more preferred pathway because of lower greenhouse gas emissions.

Livestock manure management was poor in China; only one-third of the amount of nitrogen excreted in urine and feces ultimately ended up in cropland during the last decade. The remainder was lost to the environment. About 80% of the N losses occurred from animal houses and manure storages through ammonia emission to air and through discharge of manure to watercourses. Manure management receives too little attention from research, policy and public.

The spatial separation of crop and livestock production systems is a main barrier for increasing manure utilization in cropland and replacing synthetic fertilizer by animal manure. The costs of manure transportation from livestock farms to crop farms are simply too high. Spatial reallocation and optimization of livestock farms has the potential to greatly reduce the exposure of citizens to high emissions of ammonia, and to increase the recycling of manure in cropland and thereby to decrease surface water eutrophication.

Evidently, spatial planning is a powerful policy instrument to tackle N pollution and exposure of humans to ammonia, to reduce the demand of synthetic N fertilizer, and to reduce the total N losses to the environment. However, spatial planning and optimization requires clear criteria, agreed by stakeholders. Further, there are important tradeoffs when reallocating livestock, which must be examined further.

8.8 Outlook

Though mean animal-source food consumption is much lower in China than in European countries and North and Latin American countries, there is a need to explore demand-side measures also in China. This holds especially for the affluent consumers in urban areas. Evidently, livestock production exerts large effects on land use and the environment. Understanding changes in livestock production and understanding the livestock revolution are therefore of crucial importance for developing pathways for more sustainable food systems. There is a great need to have more in-depth understanding of how livestock production is changing in other rapidly developing countries in the world. There is also a great need to improve the productivity and efficiency of livestock production, and to improve manure management. Other rapidly developing countries with an increasing consumption of animal source food may probably learn from the changes that have occurred in China. In addition, scenarios for a livestock transition in these developing countries could be built and the relevant up-stream and down-stream impacts estimated.

The implications of livestock production in proximity of large residential areas, nature conservation areas, and watercourses are still poorly known. Also, the implications of spatial optimizations of livestock production should be explored further. The results of our study in Chapter 7 suggest that there are potential positive effects and possible tradeoffs. Studies of spatial-temporal changes of livestock production at provincial and county level scale should be explore rather than at national level. A more comprehensive livestock spatial planning framework needs to be built. This framework could be used also to explore options for a spatial optimization of livestock production at global level, while considering possible trade barriers, transnational environmental costs and economic benefits.

Clearly, China will have to increase its domestic feed production to be able to reduce its reliance on importation and global market. New technologies, such as microbial protein production, and food waste based insect protein production should be developed and explored. Improved technologies are also needed in manure management and treatment.

This thesis is largely based on modelling and database analysis and synthesis. Uncertainties in the results have been discussed in most chapters, but further analyses are needed. For example, we observed some discrepancies between the data from FAOS and NBSC databases. The databases were regularly updated and corrected, but further checks are needed. Another type of uncertainty is related to the lack of region-specific and system-specific emission parameters of crop-livestock production systems; more studies are needed here.

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Summary

Livestock production has rapidly increased and livestock production systems have rapidly changed during the last decades, notably in affluent countries and countries with rapidly developing economies. The increase in livestock production is fueled by the increasing demand of livestock products, which is driven by the increasing human populations, urbanization and wealth. The increases in livestock production and the associated changes in production structure and scale have tremendous up-stream and down-stream effects on land-use and the environment. These changes are alluded by the term 'livestock revolution'.

Public awareness and recognition of the livestock revolution and its massive upstream and downstream impacts have strongly increased during the last two to three decades. The upstream impacts are related to the increasing competition between food and feed production for natural resources. The downstream impacts relate to the increased environmental pollution, such as GHG emissions, which contribute to climate change, and nutrient losses, which contribute to air pollution, water contamination and biodiversity loss. Many of these up- and down-stream impacts are profound, and have already impacted the state and stability of our Earth ecosystem.

China is an interesting case in reflecting the livestock revolution, because of the rate and size of the changes. However, the changes in livestock production in China and its impacts have not been examined in detail yet. As a result, the full cause-effect-impact relationship of the livestock revolution, and the perspectives of future livestock production in China are not well understood. Moreover, livestock production is projected to double during the next few decades. Such changes seriously question the sustainability of future livestock production in China. Thus, there is an increasing need to better understand the causes and effects of the livestock transition for different livestock categories and production systems. Also, the perspectives of future livestock production in China need to be explored in greater detail.

Hence, the main objective of the research described in this thesis is to increase the understanding of the livestock transition in China and its impacts on livestock production, manure management, nutrient cycling and the environment. The specific objectives of this thesis are: I) to develop and apply a method for a systematic analysis of the effects of international trade of food and feed during the last five decades on global land use, fertilizer N use and feed use efficiencies and livestock productivity; II) to analyze the livestock transition in China and its driving forces, impacts and possible future consequences at national and international levels; III) to analyze the N, P and K flows and losses in the 'feed-animal-manure-cropland' chain for the various livestock production systems in China, and to explore options for increasing nutrient use efficiency and recycling; IV) and to analyze the effects of the spatially uneven distributions of livestock production in China on nutrient recycling efficiency, and explore the effects of spatial planning options for crop-livestock integration on nutrient recycling efficiency .

In Chapter 2, I developed and applied a new method for a systematic quantification of the complicated impacts of international trade of food and feed on the efficiencies of global crop and livestock production. We analyzed the productivity of cropland and livestock and the associated feed and fertilizer use efficiencies for 240 countries, and estimated these countries' cumulative contributions to imports and exports of 190 agricultural products for the period 1961–2017. Crop trade has increased global land and partial fertilizer N productivities in terms of protein production, which equaled savings of 2,270 Mha cropland and 480 Tg synthetic fertilizer N over the analyzed period. However, crop trade decreased global cropland productivity when productivity is expressed on an energy (per calorie) basis. Agricultural trade has generally moved towards optimality, that is, has increased global land and N use efficiencies during 1961–2017, but remains at a relatively low level. Overall, mixed impacts of trade on resource use indicate the need to rethink trade patterns and improve their optimality.

In Chapter 3, I conducted a comprehensive analysis of the driving forces of the livestock transition in China between 1980 and 2010, and quantified profound effects on domestic and global food provisioning, resource use, N and phosphorus losses, and GHG emissions. The number of livestock tripled in less than 30 years, mainly through the growth of landless industrial livestock production systems. Changes were fueled through increases in demand, supply of new breeds, new technology, and government support. Production of animal source protein increased 4.9 times, N use efficiency at herd level tripled, and average feed use and GHG emissions per gram protein produced decreased by a factor of 2 between 1980 and 2010. In the same period, animal feed imports have increased 49 times, total ammonia and GHG emissions to the atmosphere doubled, and N losses to watercourses tripled. It was concluded that China's livestock transition has significant global impacts.

In Chapter 4, I examined the impacts of increasing domestic milk production versus increasing importation of milk from abroad on GHG emissions, N losses, land and water use, and economic performances across the main feed and milk producing countries in the world. China has an ever-increasing thirst for milk, with a predicted 3.2-fold increase in demand by 2050 compared to the production level in 2010. Meeting China's milk demand in a business as usual scenario will increase global dairy-related GHG emissions by 35% (from 565 to 764 Tg CO_{2eq}) and land use for dairy feed production by 32% (from 84 to 111 million ha) compared to 2010, while reactive N losses from the dairy sector will increase by 48% (from 3.6 to 5.4 Tg N). Producing all additional milk in China with current technology will greatly increase animal feed import; from 1.9 to 8.5 Tg for concentrates and from 1.0 to 6.2 Tg for forages. In addition, it will increase domestic dairy related GHG emissions by 2.2 times compared to 2010 levels. Importing the extra milk will transfer the environmental burden from China to milk exporting countries; current dairy exporting countries may be unable to produce all additional milk due to physical limitations or environmental preferences/legislation. We propose that a more sustainable dairy future will rely on high milk demanding regions (such as China) improving their domestic milk and feed production efficiencies up to the level of leading milk producing countries. This will decrease the global dairy related GHG emissions and land use by 12% (90 Tg CO_{2eq} reduction) and 30% (34 million ha land reduction) compared to the business as usual scenario, respectively. However, this still represents an increase in total GHG emissions of 19% whereas land use will decrease by 8% when compared with 2010 levels, respectively.

In Chapter 5, I analyzed the manure N, P, and K flows and losses in the 'feed-animal-manure-cropland' chain in China for different animal categories and production systems for the year 2010 using a modified version of the NUFER model. We estimated the N losses from the manure chain in 2010 to be up to 78% of the excreted N, while phosphorus (P) and potassium (K) losses were over 50% of the excreted P and K. The greatest losses occurred from animal housing and manure storage stages through NH_3 emissions (39% of total N losses) and direct discharge of manure into water bodies or landfill (30−73% of total nutrient losses). There are large differences among animal production systems, where the landless system has the lowest manure recycling. Scenario analyses for the year 2030 suggest that significant reductions of fertilizer use (27−100%) and nutrient losses (27−56%) can be achieved through a combination of prohibiting manure discharge, improving manure collection and storages infrastructures, and improving manure application to cropland. We recommend that current policies and subsidies targeted at the fertilizer industry should shift so as to reduce the costs of manure storage, transport, and application.

In Chapter 6, I explored the potentials of the manure management systems in China to keep N and P use within derived 'planetary boundaries' for China. Annual fertilizer use could be reduced by 26 Tg N and 6.4 Tg P following improved nutrient management. This reduction of N and P fertilizer use would contribute 30% and 80% of the required global reduction needed to keep the biogeochemical N and P flows within the planetary boundary. However, there are various barriers to make this happen.

A major barrier is the transportation cost due to the uneven distributions of crop land, livestock, and people within the country. The amounts of N and P in wastes and residues are larger than the N and P demand of the crops grown in 30% and 50% of the counties, respectively. We argue that a drastic increase in the recycling and utilization of N and P from wastes and residues can only happen following relocation of livestock farms to areas with sufficient cropland.

In Chapter 7, I explored pathways for improved spatial planning of livestock production aimed at minimizing the negative effects of concentrations of livestock production, using two contrasting criteria for optimization of the spatial planning. Over 45% of livestock production was in counties with a livestock density ≥4.0 livestock units ha^{-1}, and 60% of people were exposed to excessive ammonia emissions in 2012. Reallocation of livestock to reduce the exposure of people to ammonia led to increased total N losses, because of less favorable production conditions in other counties. Reallocation of livestock according to crop-livestock integration criteria reduced total N losses, but increased the exposure of people to ammonia. We concluded that spatial planning is a powerful policy instrument to tackle N pollution and exposure of humans to ammonia, but it requires clear criteria agreed by stakeholders.

The main conclusions of my PhD thesis are as follows:

- The livestock revolution in China was driven both by increasing demand for animal-source food and governmental policies. This transition has been unique in terms of scale and speed; it has greatly altered the livestock production structure, nutrient cycling, nutrient losses and GHG emissions. Total animal production has greatly increased.

- A novel method was developed and used to evaluate the impacts of food and feed trade on global land use efficiency, N use efficiency, livestock productivity and feed N use efficiency. Interestingly, the overall net effects of international trade on productivity and efficiency were different when the traded products were characterized in terms of calories or in protein. The massive international trade of protein-rich soybean has contributed to increased land use efficiency.

- The increasing demand of dairy products has increased China's reliance on the global market to fulfill its thirst for milk. However, satisfying the increasing demand for milk continuously through importation from abroad may not be sustainable and feasible in the long term. Instead, improving domestic milk and feed production efficiencies, up to the level achieved by the leading milk producing countries, and improving the dairy production structure seems a more preferred pathway because of lower greenhouse gas emissions.

- Livestock manure management was poor in China; only one-third of the amount of N excreted in urine and feces ultimately ends up in cropland. The remainder was lost to the environment. About 80% of the N losses occur from animal houses and manure storages through ammonia emission to air and through discharge of manure to watercourses. Manure management receives too little attention from research, policy and public.

- The spatial separation of crop and livestock production systems is a main barrier for increasing manure utilization in cropland and replacing synthetic fertilizer by animal manure. The costs of manure transportation from livestock farms to crop farms are simply too high. Spatial reallocation and optimization of livestock farms has the potential to greatly reduce the exposure of citizens to high emissions of ammonia, and to increase the recycling of manure in cropland and thereby to decrease surface water eutrophication.

- Evidently, spatial planning is a powerful policy instrument to tackle N pollution and exposure of humans to ammonia, and reduce the demand of synthetic N fertilizer and reduce the total N losses to the environment.

However, spatial planning and optimization requires clear criteria, agreed by stakeholders. Further, are important tradeoffs when reallocating livestock, which must be examined further.

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Zhaohai Bai

Shijiazhuang, February 2022

About the author

Zhaohai Bai was born on 29 January 1986 in Yancheng city, Jiangsu province, China. Zhaohai started his bachelor in China Agricultural University in 2004 after graduation from high school. Then Zhaohai continued his study in China Agricultural University (CAU). He obtained a BSc degree in 2011 and a PhD degree in Plant Nutrition from CAU in 2015. During his PhD study, he visited Soil Quality group of Wageningen University for about three years. His PhD supervisors were Prof Qing Chen, Prof Zhang Fusuo from CAU and Prof. Oene Oenema from Wageningen University.

After he got his PhD degree from China Agricultural University in 2015, he decided to do a second PhD study at Wageningen University under the supervision of Prof. Oene Oenema. His second PhD thesis is about unravelling China's livestock transition: nutrient flows and greenhouse gas emissions. During the second PhD study, he received a position in the Center of Agricultural Resources Research, IGDB, Chinese Academy of Sciences as assistant professor. He was promoted to associated professor in 2018 and to full professor at the same CAS institute in 2020.

During his second PhD study, he published the following publications:

1. Bai, Z., Ma, W., Ma, L., et al. (2018). China's livestock transition: Driving forces, impacts, and consequences. Science Advances, 4(7), eaar8534.

2. Bai, Z., Ma, W., Zhao, H., et al. (2021). Food and feed trade has greatly impacted global land and nitrogen use efficiencies over 1961–2017. Nature Food, 2, 780–791.

3. Bai, Z#., Fan, X#., Jin, X#., et al. (2022). Reallocation of 5-10 billion animals to tackle hot-spots of nitrogen pollution in China. Nature Food, 3(2), 152-160.

4. Bai, Z#., Jin, S#., Wu, Y., et al. (2019). China's pig relocation in balance. Nature Sustainability. 2(10), 888-888.

5. Bai, Z. H., Lee, M. R., Ma, L., et al. (2018). Global environmental costs of China's thirst for milk. Global Change Biology. 24 (5): 2198-2211.

6. Bai Z, Ma L, Jin S, et al. (2016). Nitrogen, Phosphorus, and Potassium Flows through the Manure Management Chain in China. Environmental Science & Technology, 50(24): 13409-13418.

7. Jin, X.,# Bai, Z.,#* Oenema, O., et al. (2020). Spatial planning needed to drastically reduce nitrogen and phosphorus surplus in China's agriculture. Environmental Science & Technology, 54 (19), 11894-11904.

8. Zhang, N#., Bai, Z#. Winiwarter, W., et al. (2019). Reducing ammonia emissions from dairy cattle production via cost-effective manure management techniques in China. Environmental Science & Technology, 53(20), 11840-11848.

9. Ma, L#., Bai, Z#., Ma, W#., et al. (2019). Exploring future food provision scenarios for China. Environmental Science & Technology, 53(3), 1385-1393.

PE&RC Training and Education Statement

With the training and education activities listed below the PhD candidate has complied with the requirements set by the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC) which comprises of a minimum total of 32 ECTS (= 22 weeks of activities)

Review of literature (6.5 ECTS)

- Unravelling China's livestock transition: nutrient flows and greenhouse gas emissions
- Mitigation of ammonia and methane emission simultaneously during the storage of dairy cattle manure (2018)

Post-graduate courses (5.3 ECTS)

- Governmental policies related to nutrient management; WUR (2013)
- Meta-analysis; CAU (2016)
- Animal manure recycling treatment and management; CAS (2018)
- FABLE Technical Training; SDSN, IIASA (2018)

Laboratory training and working visits (5.5 ECTS)

- Impacts of trade on global land and nitrogen use efficiency; Aberdeen University (2018)
- Livestock transition in China and impacts (data analysis and joint scientific paper writing); Institute of International Applied Science Agency, Austria (2017)
- Sustainable pathway of phosphorus management in the food systems of China; INRA, France (2017)

Invited review of journal manuscripts (5 ECTS)

- Environmental Science & Technology: a network flow analysis of nitrogen metabolism in Beijing, China
- Environmental Science and Pollution Research: international food trade reduces environmental effects of nitrogen pollution in China
- Environmental Science & Technology: recoupling industrial dairy feedlots and industrial farmlands mitigates the environmental impacts of milk production in China

- Agricultural Water Management: effect of vegetable waste biogas slurry on the yield, quality, nitrogen use efficiency, and water footprint of cauliflower
- Resources, Conservation & Recycling: an institutional approach to manure recycling: conduit brokerage in Sichuan Province, China

Competence strengthening / skills courses (2.5 ECTS)
- How to apply for the NSFC funding?; CARR, CAS (2018)
- How to write SCI papers?; CARR, CAS (2018)
- EndNote demonstration session; WUR Library (2019)

PE&RC Annual meetings, seminars and the PE&RC weekend (1.5 ECTS)
- PE&RC Day (2011, 2012)
- PE&RC Last years weekend (2019)

Discussion groups / local seminars or scientific meetings (7.3 ECTS)
- Seminar on case study of meta-analysis in nutrient managemen;, Shijiazhuang, China (2015)
- Seminar on 1st food chain nutrient management modelling research network; Shizhuang, China (2015)
- Seminar on nutrient management; Wageningen, the Netherlands (2016)
- Discussion group of The 565th session of Xiangshan-science conferences: key issues in China's grassland farming and farming structural adjustment; Beijing, China (2016)
- Seminar on phosphorus flows in the food chain; Kunming, China (2018)
- Group discussion on integrated crop-livestock production; Beijing, China (2018)

International symposia, workshops and conferences (11.4 ECTS)
- 3rd International Symposium on Sustainable Agriculture for Subtropical Regions (ISSASR-3); Changsha, China (2015)
- Food security forum-sustainably feeding China's growing population and the global implication; Beijing, China (2015)
- CINAg Meeting; Rothamsted, UK (2017)
- INMS international meeting; Wageningen, the Netherlands (2017)
- Sino-German P project kick-off meeting and block seminar in China; Beijing, China (2018)
- Second FABLE consortium meeting; IIASA, Austria (2018)

Societally relevant exposure (2 ECTS)

- Interview and newspapers reporting about the global impacts of China's thirst for milk (2017)
- Newspapers reporting about the China's livestock transition (2018)

BSc/MSc thesis supervision (3 ECTS)

- Greenhouse gas emission from the aquatic products import and export in China
- Greenhouse gas emission of different aquatic production system in Yangtze River Basin in China
- Impacts of livestock production subsidy policy on the geographic distribution on pig production in China

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