Sustainable agriculture expansion: estimation and reduction of nitrogen impacts, case study of China

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Abstract

Industrialization of agriculture has a number of comparative advantages however adverse implications such as environmental impacts, health hazards, GHG emissions establish the need to identify pathways to sustainable agriculture. In this paper we propose an integrated model for long term and geographically explicit planning of agricultural activities to meet demands under resource constraints and ambient targets. Environmental, resource and production feasibility indicators permit estimating impacts of agricultural practices on environment to guide agricultural policies regarding production allocation, intensification, and fertilizer application while accounting for local constraints. Physical production potentials of land are incorporated in the model, together with demographic and socio-economic variables and behavioral drivers to reflect spatial distribution of demands and production intensification levels. The application of the model is demonstrated with a case study of nitrogen accounting at the level of China counties. We discuss current intensification trends and estimate the ranges of agricultural impacts on China's environment under plausible pollution mitigation scenarios with a particular focus on nitrogen sources and losses.

Keywords

Sustainable agriculture, integrated modeling, robust allocation, environmental indicators, health exposure, nitrogen fluxes

1. Introduction

Economic growth, increasing demand for food, feed, fiber and biofuels speed up industrialization of agricultural activities characterized by new technologies, specialization and concentration, higher mechanization, increased chemical and fertilization. Production intensification is primarily guided by profit maximization principles and has a number of comparative advantages. However, there are risks and costs which are often not factored in the production planning process, e.g., loss of food and producers diversity, GHG emissions, environmental and water pollution, problems related to human health and livestock diseases, degradation and decrease of socio-economic conditions in rural areas, poverty, rural-urban migration, loss of cultural heritage, etc.

Adverse implications of production intensification, in particular, environmental impacts and health risks establish the need to identify pathways towards sustainable agriculture planning. Estimation of impacts and mitigation measures to reduce agricultural pollution over large territories is a challenging task. It requires a careful choice of models which is often driven by availability and quality of data on the one hand and on the other, by the reliability and robustness of conclusions. Pollution mitigation measures have to realistically account for location-specific demographic and economic indicators, demand and production, pollution and health risks. They should fulfill various goals and constraints, e.g., environmental norms, ambient targets, required levels of food supply, limits regarding population exposed to environmental risks, etc.

Models for planning agriculture development and assessing impacts are traditionally classified along two main lines. One line involves process-based modeling, which combines resource and production potentials of land with data-intensive biophysical processes and models of agricultural (point and disperse) pollution. The models estimate crop growth, soil carbon dynamics, soil temperature and moisture regimes, nitrogen leaching, and emissions of gasses on very fine spatio-temporal resolutions under alternative local agricultural practices (Li et al., 1992, Leonard et al., 1987). Availability of spatio-temporal data, its harmonization and further calibration of the underlying biophysical processes even at local scales is a complex task and the results are essentially subject to underlying uncertainties, data quality and model structure. Cross-comparison of process-based models often shows substantial variability and discrepancies both among the modeled outputs and in comparison to field measurements (Frolking et al., 1998). As pointed out by Bellocchi et al. (2010), the calibration and validation may require using interdependent multiple criteria, interpolation and statistics for tailoring the validation requirements to the specific objectives of the application.

The second line of models focuses on the socio-economic and behavioral aspects of agricultural producers and consumers, aggregate demand and supply. Models such as IMPACT (Rosegrant et al., 1999) perform on the level of major world regions. Resources like land, water or climatic conditions are described by scenarios in rather aggregate terms. With the focus at global or regional problems, the location-specific trends and heterogeneities can often be overlooked. Within the limits of the natural resources, sustainable land exploitation is largely determined by location-specific anthropogenic factors, i.e. demand concentrations, availability of infrastructure, market access and the complex interaction of behavioral, socio-economic, cultural and technological factors.

In this paper we discuss an integrated agriculture planning model that explicitly combines the two lines. The model employs up- and down-scaling probabilistic robust procedures (Fischer et al., 2006) that permit to match the spatio-temporal resolutions of the biophysical (process-based) models with the resolutions of the socio-economic, behavioral and optimization models, scenarios, and data to produce decisions at scales suitable for policy analysis and implementation. The model simulates different scenarios of demand increases inducing respective location-specific production adjustments. In some locations, the indicators characterising status of environment, socio-economic conditions, and humans' exposure to adverse impacts may already exceed admissible thresholds, signaling that further production growth in these locations should not take place. The question then becomes how to plan expansion of production facilities to meet demand without exacerbating the problems. For this, the model uses indicators defined by various interdependent factors including the spatial distribution of people and incomes, the current levels of crop and livestock production and intensification, and the conditions and current use of land resources. These indicators are used to discount production locations by the degree of their diverse risks and production suitability. The risk-based preference structure is then used in production allocation algorithms to derive recommendations regarding sustainable and robust production expansion, allocation and intensification. ANNEX 1 summarizes production allocation algorithm when inherent risks and contingencies are characterized by ambient constraints. In more general cases of risks and contingencies, ANNEX 2 describes an algorithm for production allocation in multi-producer environment under environmental safety and food security constraints in the form of multidimensional risk measures having direct connections with Value-at-Risk (VaR) and Conditional Value-at-Risk (CVaR or expected shortfalls) type indicators. Similar algorithm has been elaborated in the case study of rural developments in Ukraine (Borodina et al., 2011 forthcoming).

The proposed integrated agriculture production planning model is applied to the analysis of plausible agricultural pollution projections in China through 2030 (Ermolieva et al., 2005) under alternative scenarios of population, economic growth, and technological innovations. The objective of the study is to address the following questions:

- What will be the demand for agricultural products, particularly for meat, under plausible economic, demographic and urbanization development paths to 2030?
- How will increased demand for feed and food translate into the livestock numbers and crop production?
- How much nitrogen will become available from livestock manure as a consequence of livestock production intensification? How much mineral fertilizer will be needed in addition to local manure supply?
- What environmental loads, GHG emissions, and water pollution are expected as a result of agricultural production intensification?
- What improvements can be achieved by production planning based on risk indicators that jointly reduce environmental pollution through water and air contamination in different stages of agricultural production chain, i.e., from nutrients losses in livestock houses to emissions and nutrients losses on crop fields?

Because of its major role in food production and environmental sustainability, we identify nitrogen as the key nutrient in this study. On the basis of FAO projections ('World Agriculture: Towards 2015/203), it has been concluded (Eickhout et al., 2006) that "… despite improvements in the nitrogen use efficiency, total reactive nitrogen loss will grow strongly in the world's increasingly intensive agricultural systems. In the 1995–2030 period emissions of reactive nitrogen from intensive agricultural systems will continue to rise, particularly in developing countries. Therefore, the increase of nitrogen use efficiency and further improvement of agronomic management must remain high on the priority list of policy makers".

The study benefits from a socio-economic and agricultural data base (ADB) at county level consisting of about 3000 administrative units in China for the years 1997 to 2005[1]. In our work we only partially borrow data from international assessments, while the majority of the data come from the ADB. The ADB makes it possible to distinguish the heterogeneities of agricultural practices, for example, by crop and livestock types, management systems, level of production intensification, location-specific livestock housing and manure facilities, location-specific emission levels, climatic conditions, time of fertilizer application, etc.

The paper is organized as follows. Section 2 briefly outlines the main characteristics of the integrated model with further references to related publications. Nitrogen fluxes from agriculture to environment are described in more detail. Section 3 presents the summary of the recent results for China case study in which scenarios of uncertain agricultural nutrients/nitrogen impacts are analyzed and mitigation scenarios are formulated. The model permits generating infinitely many scenarios. In this paper we restrict attention to some basic cases which allow to identify potential ranges of uncertain outcomes. General discussions and conclusions are presented in Section 4. Annexes 1 and 2 include the structure of basic production allocation algorithms.

2. The model

Below we provide a brief overview of the model, which is described in more detail in Fischer et al. (2007-2010) and Ermolieva et al. (2009). The model is temporally and geographically explicit. It operates at different spatial scales, e.g., national, subnational, county-level, depending on the objectives of the research. This spatial flexibility allows for fine-tuning the associated policy advice to appropriately capture location-specific heterogeneities. Harmonized integration of socio-economic and demographic modeling components with proper scales of biophysical modeling (Fischer et al., 2002) simplifies production planning with limited resources and provides possibilities to improve production potentials in the presence of inherent uncertainties and risks, e.g., weather, contamination of environment, livestock diseases (Ermolieva et al., 2005).

To estimate levels of demand and agricultural production in China case study, agricultural activities are represented at 31 provinces and about 3000 counties. Relying on economic and demographic scenarios till 2030, developed in Huang et al. (2003) and Toth et al. (2003), demand increases and consumption of agricultural products are projected for cereals, four types of meat, milk, and eggs. The projections distinguishing between geographical regions, urban and rural areas, and vary with income.

Modeling of livestock sector dynamics is addressed with special attention. The role of livestock in global agriculture should not be underestimated. It has been recognized as an important part of a multi-faceted, integrated approach to agriculture production planning, rural community development, and environmental sustainability (Steinfeld et al., 2006; Borodina et al., forthcoming). In many countries, livestock is among the main sources of income. Livestock is growing faster than any other agriculture sector. In contrast to developed countries where livestock consumption has stabilized, in the developing countries, annual per capita consumption of meat has doubled since 1980, from 14 kg to 28 kg in 2002. Development of the livestock sector has been most dynamic in East Asia and China.

[1] The work has been conducted within EU FP6&7 projects on “Policy Decision Support for Sustainable Adaptation of China’s Agriculture to Globalization” (CHINAGRO), “Chinese Agricultural Transition: Trade, Social and Environmental Impacts” (CATSEI), “Atmospheric Composition Change, the European Network of Excellence” (ACCENT), and “Integrated Nitrogen Management in China” (INMIC, an activity of IIASA’s Greenhouse Gas Initiative). The ADB is prepared and used in CHINAGRO (Keyzer and van Veen 2005, Fischer et al. 2006, 2007), CATSEI (Fischer et al. 2006, 2008), and INMIC (Ermolieva et al. 2009, Fischer et al. 2010).

Currently, China accounts for 57% of the increase in total meat production in developing countries (Steinfeld et al., 2006). Further developments of the sector are of high priority for Chinese government.

To reflect the importance of livestock sector in Chinese agriculture development, the numbers of animals are projected from the base year consistently with economic and demographic scenarios and demand increase at the county level in such a way that production is assumed to meet the demand. Accounting for essential spatial, cultural, and climatic heterogeneities of China, the main livestock categories include poultry, pigs, dairy, cattle, buffaloes, yaks, sheep and goats, and other large animals (combing horses, donkeys, camels). Essential for nitrogen pollution, the model differentiates three main animals' management systems: traditional, specialized/industrial, and grazing. Information on the systems' shares in total livestock production is available at county level for the base year. The following assumptions estimate the mix of management system beyond the base year:

- Projections of the livestock distribution for confined[2] traditional systems are linked to the projected decrease in rural population.
- Industrial livestock systems are modeled to meet the provinces' projected demands for livestock products. These systems compensate for the decreases in traditional systems and evolve consistently with the demand growth at provincial level.
- The geographical distribution of pastoral livestock is projected in accordance with the availability and productivity of grasslands.

The rapid growth of industrial livestock farming is a major contributor to the worsening environmental quality in China. As the steadily growing population and rising incomes in China speed up demand for livestock products, growth in the animal sector will keep pace. Increasing demands can be met only by further intensification of production operations. Intensification has comparative advantages, but it also creates a number of problems which require proper regulations.

Damages are created through a number of socio-economic, environmental, and health pathways which are taken into account in the model. Intensification shifts production from rural to urban and peri-urban areas with higher demand closer to feed sources, separates the source of nutrient intake from the cycle of direct nutrient replenishment, and produces high volumes of concentrated animal waste that over-burdens urban water and waste management. Large amounts of water are being consumed in industrial animal husbandry, leading to overuse of a possibly scarce resource. Most water pollution from agriculture results from the storage and disposal of animal manure and waste. Manure, often stored in tanks or in pools known as "lagoons", may contain pathogenic bacteria and/or antibiotic residues. Leaking lagoons, but also manure application on fields, may lead to the spread of harmful compounds.

The model estimates the pollution level from livestock operations and crop fertilization with the help of a few agricultural, environmental, and biophysical indicators characterizing production intensity, water, soil, and air quality. Human health risks are measured in terms of population exposure to different levels of environmental pollution. The feasible domains of the indicator variables are subdivided into sub-domains of different degrees of impact, severity or suitability. The variables may be combined in risk functions to reflect the levels of different risks in areas associated with agricultural production (Ermolieva et al., 2009, Fischer et al., 2008, 2010). Production is increased primarily through the establishment of new

[2] Confined system includes post-harvest stubble grazing, as opposed to purely grazing systems relying only on pastures.

facilities and/or the expansion of existing facilities. In some areas, especially in the vicinity of urban areas, the indicators may signal that a further allocation of production is impossible. For not exacerbating environmental and health problems, production facilities are then adjusted according to a production allocation algorithm summarizes in Annex 1 (for details see Fischer et al., 2007, 2008).

In this paper, we discuss how the model is applied to account for agricultural nitrogen only. Plants require nitrogen for growth as well as for providing protein in food or feed, which leads to the application of nitrogen fertilizer in agricultural practice (Smil, 2001). In China, nitrogen fertilizers are being widely used to increase yields on scarce land resources. A large part of the fertilizer applied is not taken up by plants, but released into the environment cascading through diverse environmental pools (Galloway et al., 2004; Erisman et al., 2007). In the model, the environmental effects of soil nitrogen are measured in terms of atmospheric emissions of NH_3, N_2O and NO, and its leaching to groundwater or surface water (Velthof et al, 2008). The model estimates nutrient losses associated with agricultural nitrogen (manure as well as mineral fertilizers) at the level of counties for:

(1) *point-source losses* in the form of emissions to the atmosphere and leaching to ground and surface water from specific release points such as livestock housing or manure storage facilities.

(2) *non-point losses* resulting from the application of fertilizer and manure to cultivated land or from grazing livestock in pasture areas. Non-point nutrient losses have two components. The first part comprises non-effective nutrients, i.e., nutrients not reaching the crop (including losses due to emissions, runoff and leaching), which depend on the environmental setting and nutrient application practices. These losses occur independently of crop uptake capacity. The second part consists of potentially effective nutrients that reach the crop root zone. Released quantities depend on the crop's uptake capacity.

Based on the spatially explicit distribution of animals and crop production projected by the model, we apply existing schemes of nitrogen releases to assess the loss of agriculturally derived nitrogen compounds along different pathways. The scheme of this model is presented in Figure 1. As a consequence of agricultural activities, emissions of N_2O and NH_3 into the atmosphere, and of nitrate leached to groundwater are assessed.

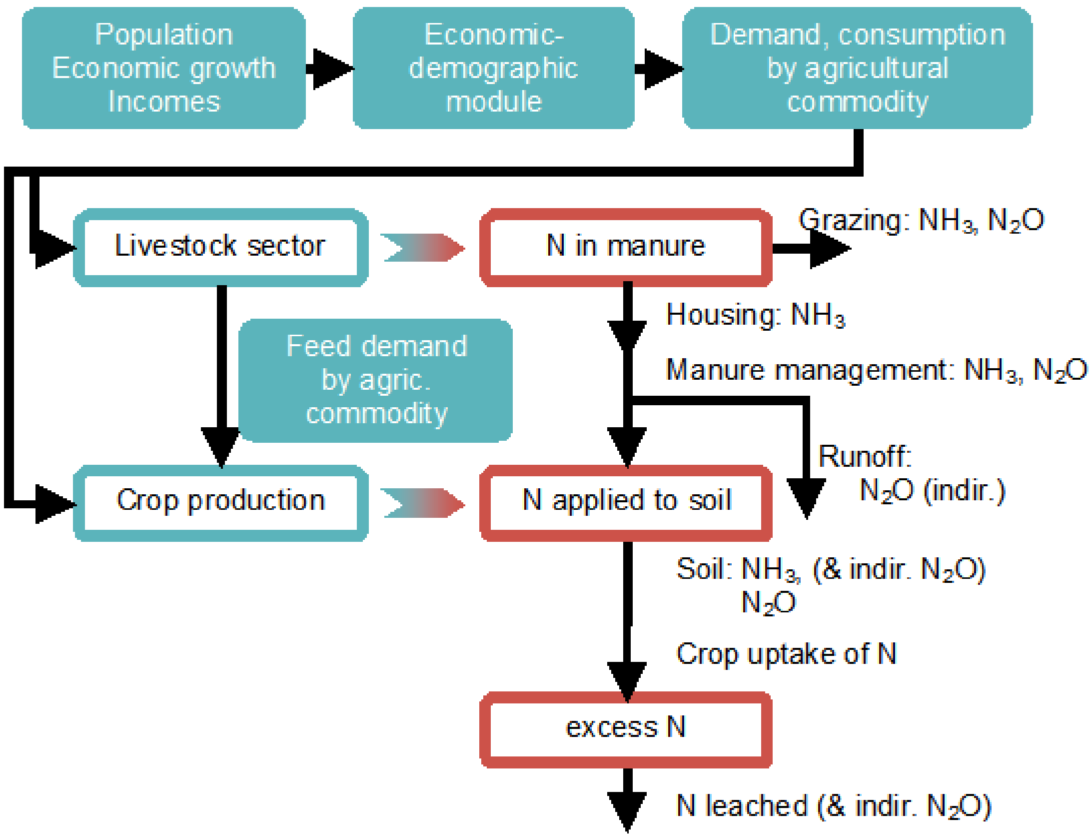


Figure 1. Nitrogen cascading: Schematic structure of the model.

For estimating nitrogen leaching to groundwater, we adopt the *MITERRA model* (Velthof et al., 2009), which applies a combined water and nitrogen balance to derive indicators of leaching for a broad range of soil types aggregated into seven classes (sandy, clay, gleyic, stagno-gleyic, peat, loam, and paddy soils) with different leaching characteristics (Shi et al., 2004; FAO/IIASA/ISRIC/ISSCAS/JRC, 2009). Soils are also distinguished by the type of crop water management, i.e., separately for irrigated and rain-fed land. For each soil class, climate condition (e.g., precipitation, temperature), and land use, the approach estimates the fraction of nitrogen surplus that moves to the ground water, i.e., the leaching fraction. Soils used for rice paddies are considered impermeable to water and assumed to have no leaching.

In a simplified way, the accounting of nitrogen applied to the field may be described as follows:

$$N\_Surpl = N_{mnr} + N_{fert} + N_{fix} - N_{upt},$$

where N_Surpl denotes surplus of nitrogen applied to the field, N_{mnr} is nitrogen in manure available for field fertilization (net of losses during housing); N_{fert} is nitrogen in chemical fertilizers; N_{fix} is atmospheric nitrogen fixed by N-fixing crops; N_{upt} is nitrogen uptake by all crops (net of nitrogen left in recycled crop residues). The nitrogen surplus, together with the leaching fraction derived according to soil and climate parameters, allows for the estimation of nitrogen leaching, in kilograms N per hectare cultivated land.

For estimating the N_2O emissions to the atmosphere we employ the methodology developed for IIASA's Greenhouse gas–Air pollution INteractions and Synergies (GAINS) model (Winiwarter, 2005). GAINS applies IPCC default emission factors (IPCC, 2000) recommended for national inventory submissions to the UNFCCC. Thus the derived results can be consistently compared among countries. The N_2O emissions are computed as the product of an emission factor times the respective activity data for manure management, grazing and soils. Indirect emissions (both from leaching and redeposition of gaseous releases) are implicitly covered in the emission factor applied, and therefore the model results generally are very close to those assessed with the IPCC method.

Ammonia emissions (NH_3) from livestock production are estimated at four major stages: in animal houses; during storage of manure; when applying manure; and from livestock grazing. These stages are explicitly distinguished in the livestock model (Fischer et al. 2007; Fischer et al. 2008b) and in the GAINS methodology (Klimont, 2001; Klimont and Brink, 2004). The ammonia accounting is based on experience and parameters developed primarily for Europe. Here we are considering the impact of local manure management practices and reflect in the estimates different levels of livestock productivity (e.g., Bouwman et al, 1997; Ermolieva et al., 2005; Menzi, 2001; NuFlux, 2001).

Emissions of ammonia from mineral fertilizer application depend on multiple factors including type of fertilizer applied, soil properties, meteorological conditions, time of application in relation to a crop canopy, and method of application. The nitrogen losses from fertilizer application are region specific. It must be stressed that the uncertainty range of emission factors is large. Typically, nitrogen losses from synthetic fertilizers vary between 1 and 4 percent, with the exception of ammonium sulfate (8 percent), urea (15 to 25 percent) and ammonium bicarbonate (ABC) (20 to 30 percent). Application practice is very different between countries, and may range from mostly non-volatizing fertilizers (typical for most of Europe) to predominantly urea and ABC (China). Figures 3-4, 6-7, 9-10 demonstrate uncertainty ranges only for some basic scenarios. The outcomes of such a scenario-based uncertainty analysis can be further used for designing robust development paths and corresponding nitrogen outputs, which is beyond the scopes of this paper.

3. Numerical application: a case study of China

In the studies, several scenarios of pollution mitigation options in China were analyzed and compared with regard to excess nitrogen:

(i) a "business-as-usual" scenario, in which the increase of production is allocated proportionally to the demand increase, which is concentrated in the vicinity of densely populated urban areas;
(ii) a reallocation scenario that combines the demand driven preference structure of the business-as-usual scenario with information on population densities and urban agglomerations to reduce risks caused by livestock production;
(iii) an "optimizing fertilizer use" scenario (first apply manure; only then supply nutrients to crops with mineral fertilizer); and
(iv) a scenario consisting of optimized fertilizer use combined with technological options focused on ammonia abatement ("minimized ammonia" scenario).

The business-as-usual scenario (i) implicitly minimizes the transportation costs as production concentrates in the vicinity of urban areas with high demand. In the alternative scenario (ii), the production is shifted to more distant locations characterized by availability of cultivated land, lower livestock and population density, but at the expense of additional transportation. In addition to (ii), scenario (iii) focuses on reducing fertilizer application, while scenario (iv) represents a scenario of drastic ammonia emission reductions. Formally, the scenarios correspond to different priors in the allocation procedure summarized in Annex 1.

The effects of the four scenarios are compared in terms of nitrogen leaching fraction, (Figure 2), the quantities of leached nitrogen (Figures 3-5), and nitrogen emitted to the atmosphere as N_2O (Figure 6-8) or NH_3 (Figure 9-11). These indicators have been computed for each spatial administrative unit, i.e., county. Figures 2-10 illustrate spatial heterogeneity of the indicators for the base line scenario only. Table 1 summarizes the results for all scenarios.

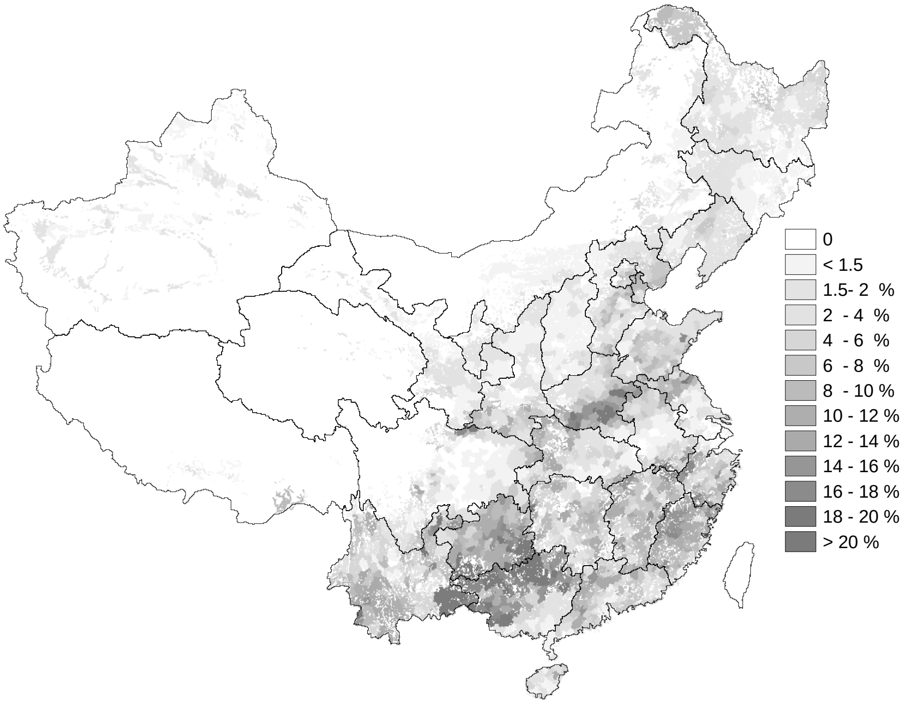


Figure 2. Nitrogen leaching fraction, in %.

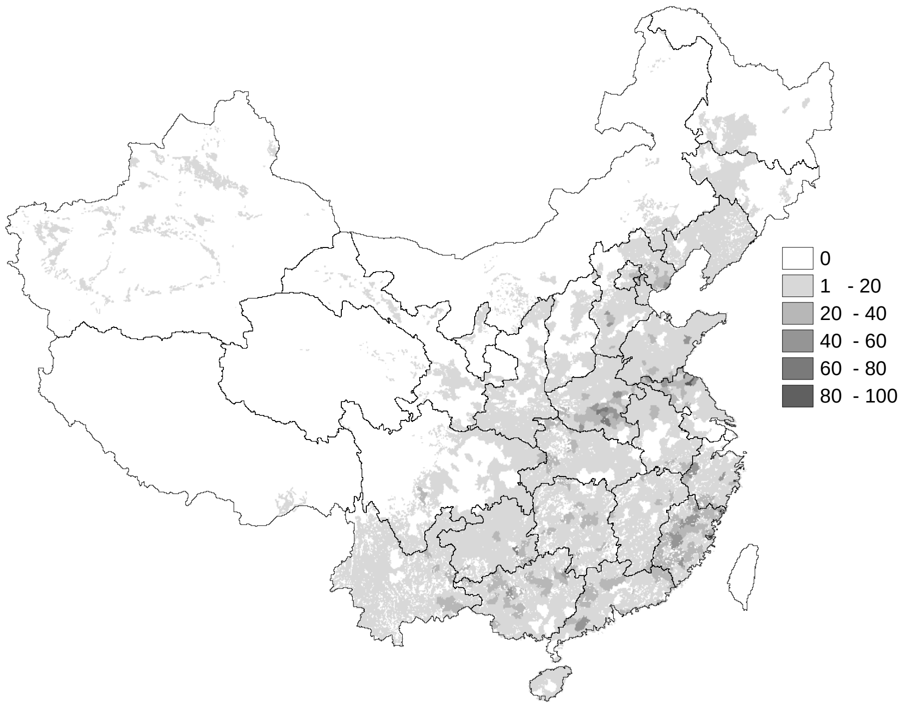


Figure 3. Leaching in kg / ha cultivated land, in 2000.

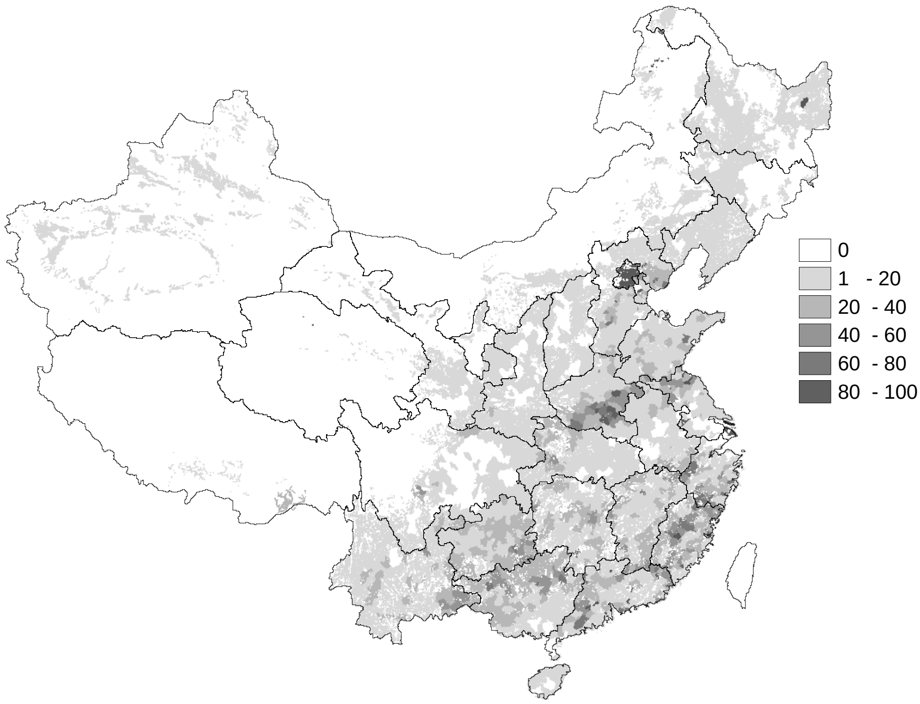
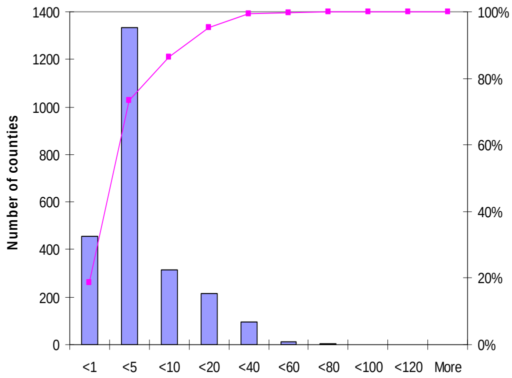
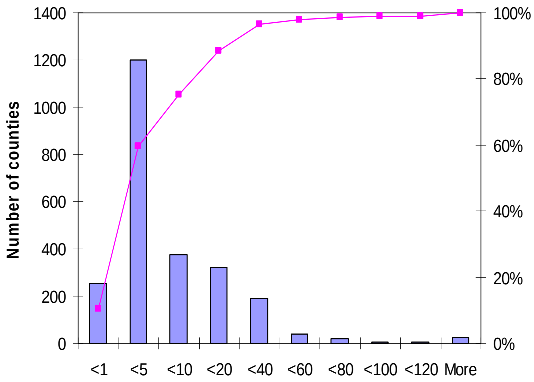


Figure 4. Leaching in kg / ha cultivated land, in 2030.



a)



b)

Figure 5. Leaching in severity classes by number of affected counties, in kg / ha cultivated land, in 2000 (a) and 2030 (b).

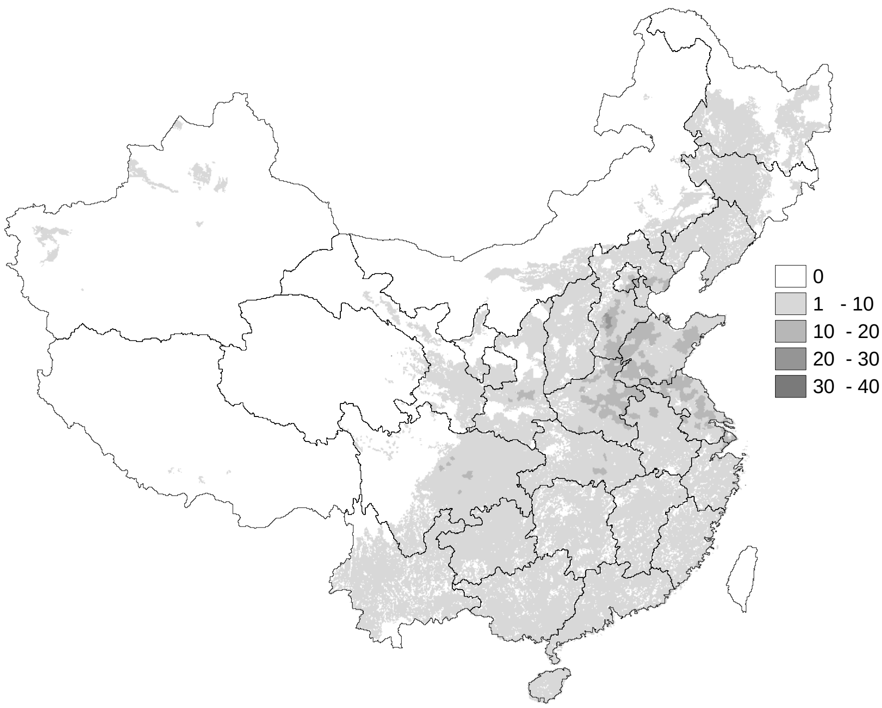


Figure 6. N_2O in kg / ha cultivated land, in 2000.

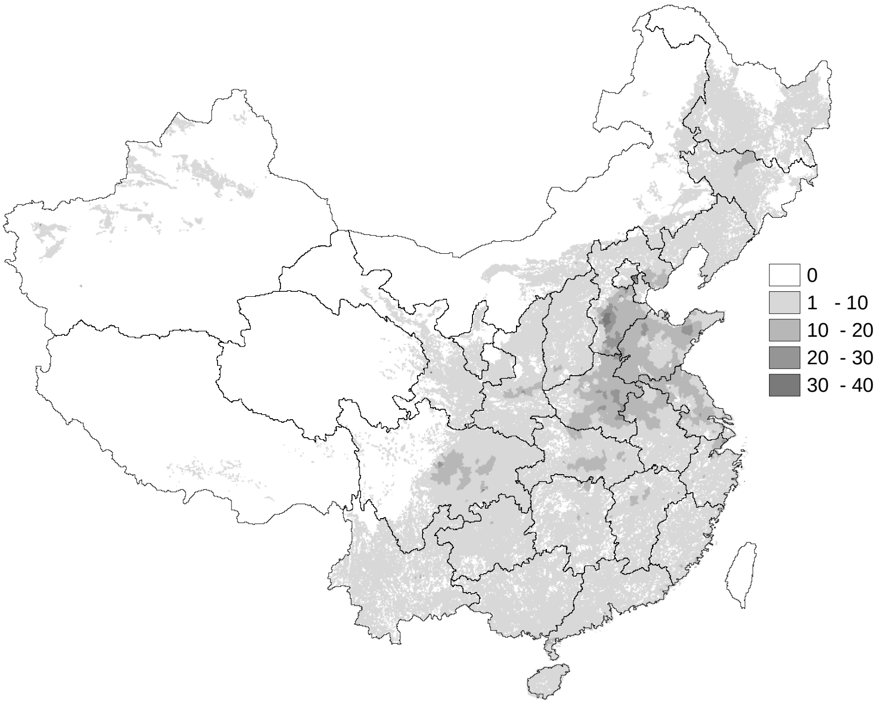
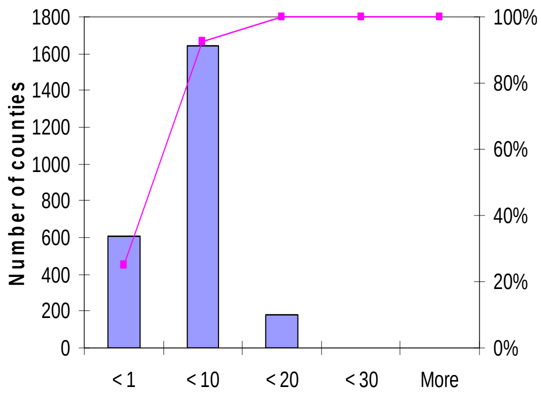
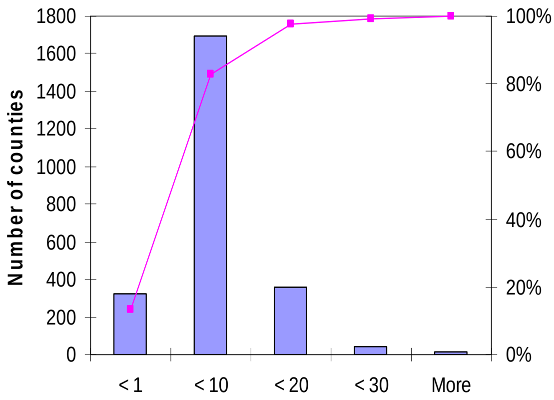


Figure 7. N_2O in kg / ha cultivated land, in 2030.



a)



b)

Figure 8. N_2O emissions by size classes and number of affected counties, in kg per ha of cultivated land, for 2000 (a) and 2030 (b).

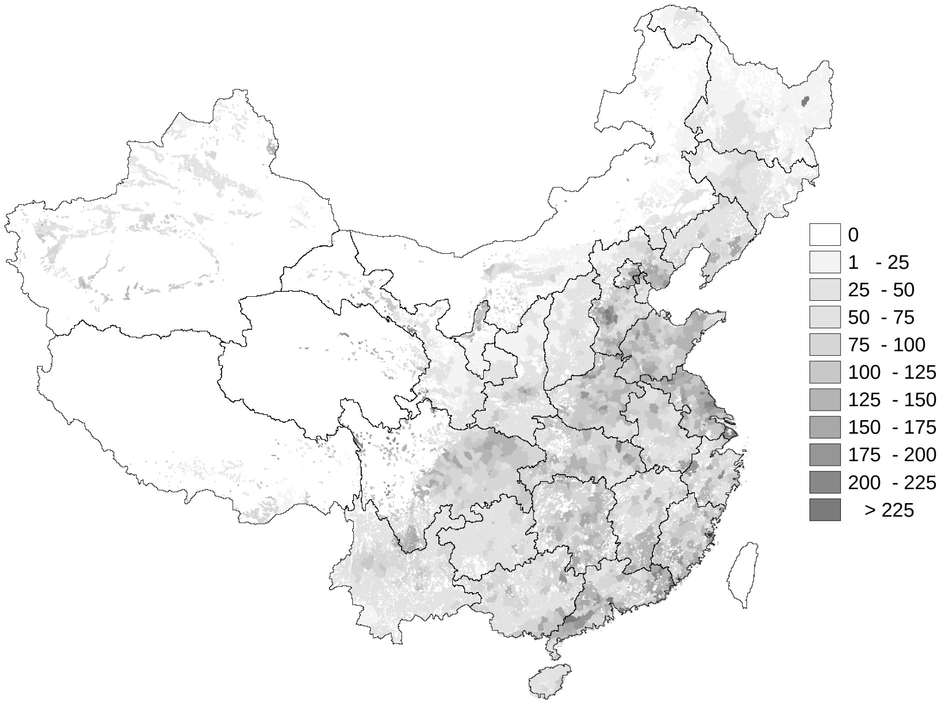


Figure 9. Ammonia emissions from agriculture (kg ammonia/ha cultivated land) in 2000.

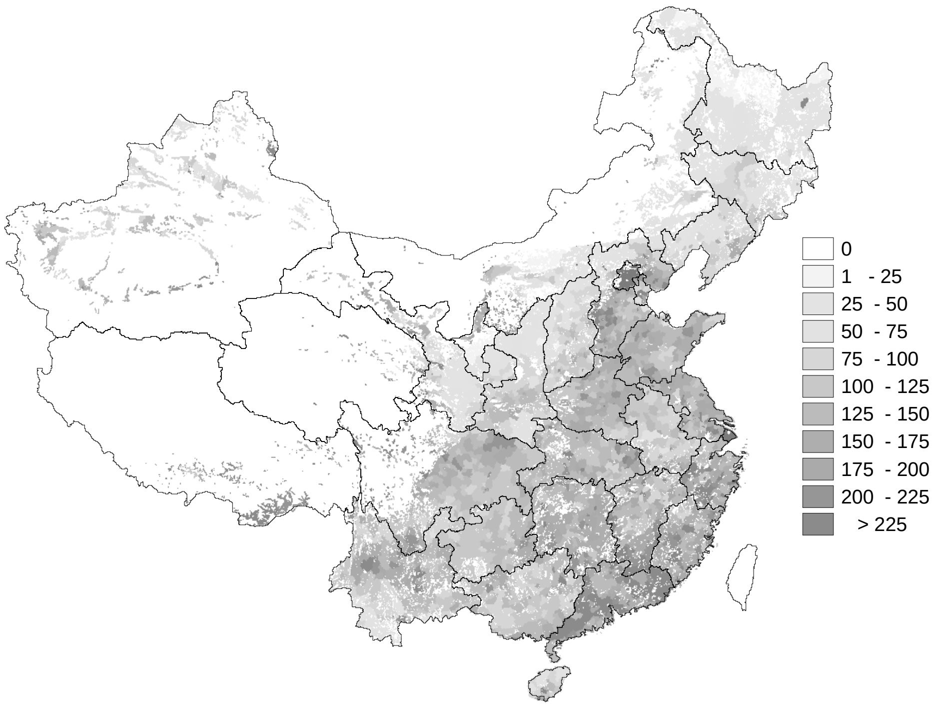
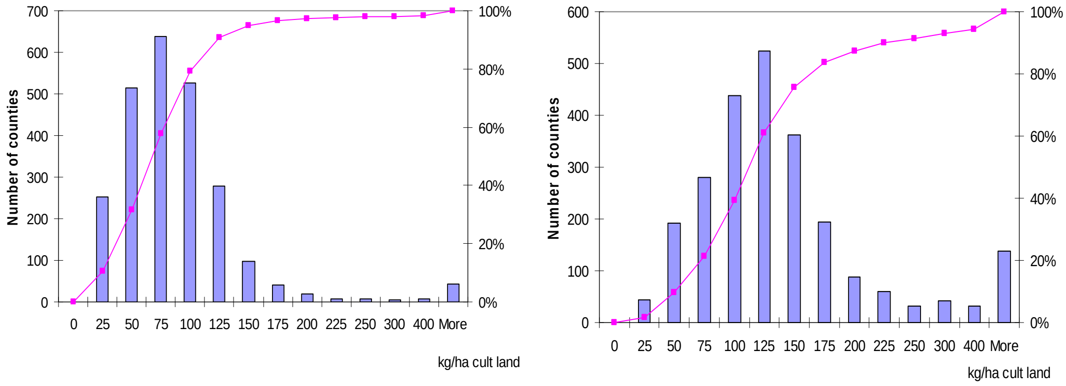


Figure 10. Ammonia emissions from agriculture, in kg ammonia/ha cultivated land, in 2030.



a) b)

Figure 11. Ammonia emissions by size classes and number of affected counties, in kg per ha of cultivated land, for 2000 (a) and 2030 (b).

Table 1. Aggregated environmental pressure indices for total China by scenario. Data in kt N.

Scenario	**2000**			**2030**			**Change 2000-2030 (%)**		
	Lch	**N_2O-N**	**NH_3-N**	**Lch**	**N_2O-N**	**NH_3-N**	**Lch**	**N_2O-N**	**NH_3-N**
BAU	701	855	7469	1101	1282	10878	57	50	46
Sustainable reallocation	--	--	--	1077	1278	10848	54	49	45
Optimization	--	--	--	321	956	7487	-54	12	0
Minimized ammonia	--	--	--	328	963	6884	-53	13	-8

In all scenarios and for all years considered, the greatest part of nitrogen loss is via the ammonia pathway. NH_3 nitrogen emissions are between 40 and 50% of total nitrogen application. Loss in the form of N_2O follows next, and it may seem somewhat surprising that nitrate leaching is even less important. This can be explained by the huge losses in the gas phase, while the leaching fractions (relatively small) apply only to the quantity of soil nitrogen not lost or used elsewhere, and also do not include nitrate runoff to surface water.

The scenarios vary, but it is clear in all cases that nitrogen pollution is likely to increase over time. Even in the scenarios that provide considerable improvements, aggregate environmental pressures associated with livestock manure and use of mineral fertilizer increase between 2000 and 2030 by roughly a third to almost half.

Environmental risk indicators[3] are determined for each administrative region. However, it is not so much the area that is affected by an adverse environmental situation, but the health of the population which is at stake. In order to estimate this risk, we need to assess the exposure of people. Figure 12 presents population exposure in terms of different classes of

[3] Risk indicators are introduced in terms of ambient targets similar to norms on water or air pollution. Fischer et al. (2009) provide further discussion on VaR and CVaR risk indicators.

ammonia emissions in 2000, and Figure 13 compares the same indicator for the four alternative scenarios by China regions and aggregated for the whole country. Ammonia is selected not because it is dangerous to humans, although ammonia exposure may be harmful. As mentioned, ammonia is a major nitrogen flux, and may therefore serve as a robust indicator of the overall nitrogen load both on the environment and on humans. While we are not able to provide information on an absolute risk measured in monetary terms, we may compare between the different scenarios. The scenarios (ii), (iii) and (iv) reallocate agricultural production to minimize overall risk (see the procedure in the Annex 1 and Ermolieva et al., 2009, Fischer et al., 2010). This function, considering population density implicitly on the demand side, effectively performs reallocation away from population. The altered distribution pattern of production alone will then reduce the exposure of the population. With mitigation measures in place, the gradual improvement of the situation across scenarios (Figure 13), alleviating the exposure of people to environmental and health risk, becomes visible when comparing the incremental additional measures leading from the *business-as-usual* to the *minimized ammonia* scenario.

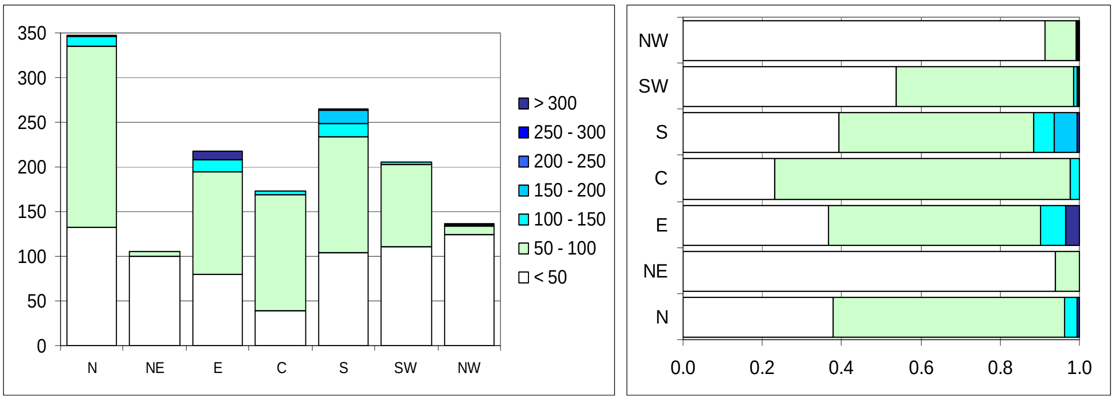
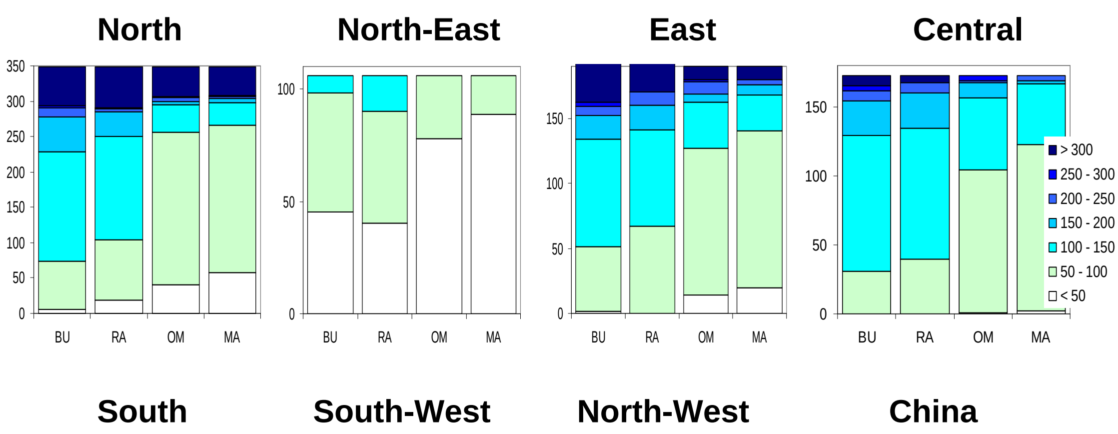


Figure 12. Absolute (million people) and relative (share of total population) distribution of population according to classes of severity of environmental pressure (measured in terms of kg nitrogen in ammonia emitted per ha cultivated land), 2000. The label on the horizontal axis indicates the regions in China: N, NE, E, C, S, SW, NW stand for North, North-East, East, Center, South, South-West, North-West, respectively, business-as-usual scenario.



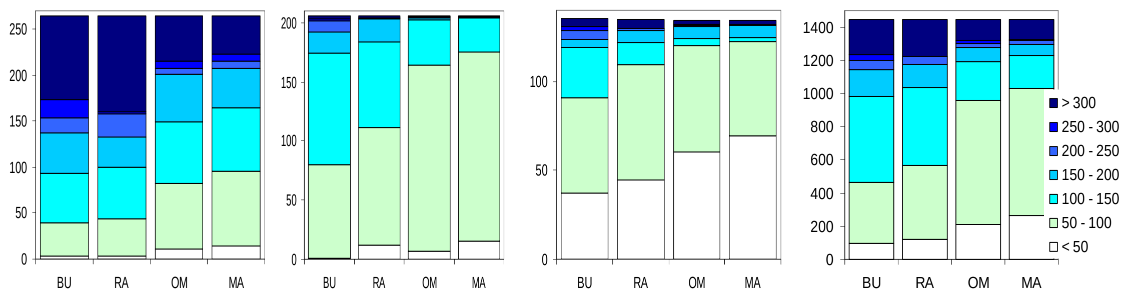


Figure 13. Number of people by classes of severity of ammonia losses (kg nitrogen in ammonia emitted per ha total area), by economic regions and scenarios (BU=business-as-usual, RA=reallocation, OM=optimized manure; MA=minimized ammonia), in 2030.

4. Concluding remarks

In this paper we discussed model-based estimation of nitrogen fluxes from agriculture in China. Proposed integrated agriculture planning model combines the advantages of the two main types of traditional modeling: process-based and aggregate socio-economic demand-supply modeling. The model derives indicators quantifying environment pollution and health exposure from agricultural activities spatially. This permits to assign a risk preference structure to specific locations and to guide recommendations regarding robust production allocation and intensification. The model has been applied to the analysis of agricultural developments in China through 2030 focusing on pollution abatement options. In this paper, agricultural pollution is measured in terms of nitrogen excess. Nitrogen deserves special attention for a number of reasons. While being the main component in the atmosphere (78%), molucular nitrogen as such is not accessible to plants. And first needs to be “fixed” to form reactive nitrogen, i.e. nitrogen in form of a range of different chemical compounds. Reactive nitrogen is indispensable in agriculture to stimulate plant growth, however recent nitrogen pollution studies show the incredible growth of nitrogen in the environment. While in 1860, humanity produced 15 million metric tons of reactive nitrogen,by 1995, that number was at 156 million tons, and increased to 187 million tons by 2005 (Galloway et al., 2004; 2008). In comparison to global CO_2 emissions of 27 billion tons annually, these numbers may seem small, but nitrogen impacts are magnified by the so called nitrogen cascade, i.e., propagation of nitrogen fluxes through the atmosphere, into the soil, into the water, into the coastal systems and back into the atmosphere, which considerably magnifies the impacts.

Some of the reactive nitrogen comes from industry, but the majority comes from agricultural activities – livestock production and crops fertilization. The most important pathways of nitrogen loss to the environmental are atmospheric emissions of NH_3, N_2O and NO, and leaching to groundwater or surface water. Nitrogen losses are expected to further increase in the future, as demand for agricultural products will continue to rise. Much of the accelerated N cycle is expected in China where remarkable increase in the production of agricultural products was observed over the last years. Four alternative agricultural scenarios of nitrogen pollution reduction in China are developed and discussed. We differentiate between abatement strategies that provide a spatial production reallocation - very effective in terms of costs and reduction of population exposure, and scenarios that provide actual reductions in nutrient (nitrogen) application and release. Environmental sustainability aspects of these scenarios are compared with respect to area and human exposure to different severity classes of risks. The derived geographically explicit results of the four scenarios indicate essential location-specific heterogeneities regarding the level of environmental risks, health exposure, and economic capacity to cope with the risks. However, it is clear that in all locations, the level of pollution is likely to increase. Even in scenarios (iii) and (iv), which carry considerable improvements, China's environmental deterioration increases by roughly one third to nearly half. Nitrogen losses in the form of ammonia pose the biggest problem by far in all four scenarios. Losses in the form of N_2O follow, while nitrate leaching is of less importance. While we do not intend to provide a full nitrogen balance for China, the results may be useful to look at the respective fluxes in perspective to gain an overall understanding of the system.

A comparison of results obtained here with literature values indicates general agreement on the magnitude of nitrogen fluxes. Because of uncertainties and variability, instead of using absolute estimates for planning pollution mitigation options we propose to employ approaches based on robust indicators. These proved to have a success in evaluating the preference structure of feasible decisions. The results relying on the indicators may then be considered valuable for policy advice. In this application our indicators guide production allocation and intensification in such a way that both the demands targets (food security) and security constraints on the environment and health exposure are met.

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ANNEX 1:

Let us briefly summarize the production allocation algorithm (for more details, see Fischer 2007; Fischer 2008). The objective is to allocate new supply facilities in the best possible way to meet the projected increase in national demand d_i for agricultural products i among the production activities/locations k, $k = \overline{1,K}$ while considering various risk indicators. In the following model, the risks are treated as constraints on production expansion (similar to ambient targets in pollution control models). Therefore, the problem is to determine suitable activity levels y_{ik} given the constraints:

$$\sum_k a_{ik} y_{ik} \geq d_i, \tag{1}$$

$$y_{ik} \geq 0, \tag{2}$$

$$\sum_i y_{ik} \leq b_k,\ i = \overline{1,m},\ k = \overline{1,K}, \tag{3}$$

where b_k denotes thresholds for environmental and health risks and imposes limitations to an increase in production of system or location k, $k = \overline{1,K}$. Apart from b_k, there may be additional limits on y_{ik}, $y_{ik} \leq r_{ik}$, which may be associated with legislation, for example, to restrict production i to a production "belt" or to exclude production i from urban or protected areas, etc. Thresholds b_k and r_{ik} may indicate strictly prohibited levels. The procedure may also allow for the thresholds to be exceeded while imposing taxes or requiring a premium to be paid for the mitigation of certain risks. Coefficients $a_{ik} \geq 0$ describe heterogeneities among products in different locations. Equations (1)-(3) are well established in the literature and belong to the type of generalized transportation problems. They may describe also dynamic problems of allocations by using i to enumerate demands for different products at different time intervals, e.g., indices $i = 1,...,m$ may indicate demand for products at time $t = 1$; $i = m+1,...,2m$ indicate demand for products at time $t = 2$, and so on. In general problems, equation (3) has the structure $\sum_i \beta_{ik} y_{ik} \leq b_k$, $i = \overline{1,m}$, $k = \overline{1,K}$, where β_{ik} may reflect different pollution outputs from i in location k. With deterministic β_{ik} this type of equations can be reduced to equation (3) by introducing new decision variables $x_{ik} = \beta_{ik} y_{ik}$.

Apart from equations (1)-(3) there always exist additional information regarding uncertain activity levels y_{ik}, i.e., behavioral uncertainties. These information is used to derive a prior probability q_{ik} reflecting the assumption that a unit of demand d_i for product i should be supplied by activity/location k. For instance, it is reasonable to allocate more livestock to areas with a higher demand increase, higher productivity or better access to animal feeds, transportation services. This preference structure is expressed in prior q_{ik}, $\sum_k q_{ik} = 1$ for all i. The use of priors is consistent with spatial economic theory (see, e.g., in Fujita 1999; Karlqvist et al. 1978). The likelihood q_{ik} can be modeled as inversely proportional to production costs, distances, risks and ambient targets b_k and r_{ik}. In the case a_{1k}, i.e. standard transportation model, an initial estimate of production i allocated to k can be derived as $q_{ik} d_i$. This could, however, result in a violation of applicable restrictions (3). Sequential rebalancing (Fischer et al. 2006) proceeds as follows. For the simplicity of illustration, we assume that $a_{ik} = 1$. The *expected* initial allocation of d_i to k is $y_{ik}^0 = q_{ik} d_i$, $i = \overline{1,m}$. As this allocation may not comply with constraint $\sum_i y_{ik}^0 \leq b_k$, $j = \overline{1,n}$, the relative imbalances $\beta_k^0 = b_k / \sum_i y_{ik}^0$ are derived and updated $z_{ik}^0 = y_{ik}^0 \beta_k^0$, $i = \overline{1,m}$. Now constraint $\sum_i y_{ik} \leq b_k$ is met, $k = 1,2,...$, but the estimate z_{ik}^0 may cause an imbalance for relation (1), i.e., $\sum_k z_{ik}^0 \neq d_i$. Continue calculating $\alpha_i^0 = d_i / \sum_k z_{ik}^0$, $i = \overline{1,m}$ and updating the imbalances $y_{ik}^1 = z_{ik}^0 \alpha_i^0$, etc. The estimate y_{ik}^s can be represented as $y_{ik}^s = q_{ik}^k d_i$, $q_{ik}^s = \left(q_{ik}\beta_k^{s-1}\right) / \left(\sum_j q_{ik}\beta_k^{s-1}\right)$, $i = \overline{1,m}$, $k = 1,2,...$. Assume $y^s = \left\{y_{ik}^s\right\}$ has been calculated. Find $\beta_k^s = \bar{b}_k / \sum_i y_{ik}^s$ and $q_{ik}^{s+1} = \left(q_{ik}\beta_j^s / \sum_i q_{ik}\beta_j^s\right)$, $i = \overline{1,m}$, $k = 1,2,...$, etc.

In this form the procedure can be considered as a redistribution of required supply d_i among producers $k = 1,2,...$ by applying the sequential adjustment q_{ik}^{s+1} , i.e., by using a Bayesian type of rule to update the prior distribution: $q_{ik}^{s+1} = q_{ik}\beta_k^s / \sum_i q_{ik}\beta_k^s$, $q_{ik}^0 = q_{ik}$.

The iterative update of q_{ik} is based on an 'observation' of imbalances of the basic constraints rather than calculated for observations of random variables. A simple rebalancing procedure, similar to the one mentioned above for standard transportation constraints (1)-(3) was proposed by G.V. Sheleikovskii (for more details and references, see Bregman 1967) for the estimation of passenger flows between old and projected new regions. Similar procedure for general problem (1)-(3) may be used for analysing of interregional migration, agricultural export-import flows, etc. Verification of its convergence to the optimal solution maximizing the cross-entropy function

$$\sum_{i,k} y_{ik} \ln \frac{y_{ik}}{q_{ik}} \qquad (4)$$

is provided in Fischer et al. (2006) for general forms of constraints. The alternative scenarios introduced in Section 4 correspond to different production allocation priors q_{ik} , $i = \overline{1,m}, k = \overline{1,K}$. The behavioral uncertainty can also be treated in a stochastic manner as a random allocation of demand d_i among points $k = \overline{1:K}$ with respect to the prior probability q_{ik}, which is a topic of a separate paper.

ANNEX 2:

The approach presented in ANNEX 1 guides production expansion relying on individual behavioral principles set by priors. The risks are characterized by imposing certain standards as additional ambient or "safety" constraints. In general, these constraints may depend on some scenarios of potential future shocks to the system. Let us consider now a more general multi-producer (location) model in a stochastic environment. We may assume that there is a coordinating agency (a principle agent). The agency acts as a social planner and is responsible for maximizing the overall performance of the production chain to stabilize the aggregate production under minimal costs. Suppose that the agency has to determine levels y_{ik} of product i in locations k in order to meet stochastic demand $d_i(\omega)$, where $\omega = (\omega_1, \omega_2,...)$ is a vector of all contingencies affecting demand and production. It is naturally to assume that the decision on production expansion has to be made before the information on contingencies arrives. In this case, the total ex-ante production may not exactly correspond to the real demand, i.e., we may face both oversupplies and shortfalls. In other words, the amount of production y_{ik} , $k = 1,...,K$, which is planned ex-ante to satisfy the demand $d_i(\omega)$, $y_i(\omega) = \sum a_{ik}(\omega) y_{ik}$ may underestimate ($y_i(\omega) < d_i(\omega)$) or overestimate ($y_i(\omega) > d_i(\omega)$)

the real demand $d_i(\omega)$ under revealed contingencies ω and the safety constraints imposed by strict thresholds b_k in (3). The constraint (3) necessitates, in general, additional supply of ex-ante production $z_i \geq 0$ from external sources (say, through international trade). It may also require the ex-post redistribution of the production from internal producers, $k = \overline{1,K}$, to eliminate arising shortfalls and oversupplies in locations. For now, let us ignore these ex-post redistributional aspects assuming that the most significant impacts are associated with ex-ante decisions y_{ik} and z_i. In fact, the presented further model can be easily extended to represent the ex-post adjustments of decisions y_{ik}, z_i, as well as more detailed temporal aspects of production planning.

Let c_{ik} be the unit production cost. In more general model formulation, c_{ik} also include the unit transportation cost for satisfying location-specific demand. Then the model of production planning among the facilities under ambient and other constraints can be formulated as the minimization of the total cost function:

$$f(y,z) = \sum_{i,k} c_{ik} y_{ik} + \sum_{i=1}^{m} e_i z_i \,,$$

subject to constraints (2), (3), and the following additional safety constraints

$$P\left[\sum_{k=1}^{K} a_{ik}(\omega) y_{ik} + z_i \geq d_i(\omega)\right] \geq p_i \,,\ z_i \geq 0 \,,\ i = \overline{1,m} \,, \tag{5}$$

where $e_i > 0$, $i = \overline{1,m}$, denotes the unit import cost. A safety level p_i, $0 < p_i < 1$, defines the food security constraint (regulates the supply-demand relations) for all possible scenarios (contingencies) ω. The introduction of constraints (5) is a standard approach for characterizing stability in case of insurance business, security of nuclear power plants and other risky activities. Safety constraints of type (5) are usually used in cases where impacts of random interruptions can not be easily evaluated. In this case, the value p_i is selected such that an expected shortfall occurs only, say, once in 100 month, i.e., $1 - p_i = 1/100$.

The main methodological challenge is concerned with the lack of convexity of constraints (5). Yet, the remarkable fact is that the model defined by (2)-(3), (5) can be effectively solved by linear programming methods due to the following convex reformulation of this model. Let us consider the minimization of the expectation function

$$F(y,z) = f(y,z) + \sum_{i=1}^{m} \alpha_i E \max\left\{0, d_i(\omega) - \sum_{k=1}^{K} a_{ik}(\omega) y_{ik} - z_i\right\}, \tag{6}$$

subject to constraints (2), (3), and $z_i \geq 0$, $i = \overline{1,.m}$. The minimization of function $F(y,z)$ is a rather specific case of stochastic minimax models analyzed (both optimality conditions and solution procedures) in Ermoliev and Wets, 1988. In particular, if $F(y,z)$ has continuous derivatives with respect to z_i, e.g., the probability distribution function of ω has continuous density function, then

$$\frac{\partial F}{\partial z_i} = e_i - \alpha_i EI(d_i(\omega) - \sum_{k=1}^{K} a_{ik}(\omega) y_{ik} - z_i \geq 0) \,,$$

where $I(\xi \geq 0)$ is the indicator function: $I(\xi \geq 0)=1$, if $\xi \geq 0$, and $I(\xi \geq 0)=0$ otherwise. Therefore, we can rewrite $\frac{\partial F}{\partial z_i}$ as

$$\frac{\partial F}{\partial z_i} = e_i - \alpha_i P\left[d_i(\omega) - \sum_{k=1}^{K} a_{ik}(\omega) y_{ik} - z_i \geq 0\right], \tag{7}$$

which allows to establish connections between the original model defined by (2), (3), (5) and the minimization of convex function $F(y,z)$ defined by (6).

Assume (y^*, z^*) minimizes $F(y,z)$ subject to constraints (2), (3), and $z_i \geq 0$, $i=\overline{1,.m}$. Assume also that $e_i < \alpha_i$, $i=\overline{1,.m}$. Then from (7) it follows that for all i with positive components $z_i^* > 0$, i.e., when $\frac{\partial F}{\partial z_i}=0$, the optimal solution (y^*, z^*) satisfies the following safety constraints

$$P\left[d_i(\omega) - \sum_{k=1}^{K} a_{ik}(\omega) y_{ik} - z_i \geq 0\right] = e_i / \alpha_i . \tag{8}$$

Moreover, for all i with $z_i^* = 0$, i.e., when $\frac{\partial F(y^*, z^*)}{\partial z_i} \geq 0$, the optimal (y^*, z^*) satisfies the following safety constraint

$$P\left[d_i(\omega) - \sum_{k=1}^{K} a_{ik}(\omega) y_{ik} \geq 0\right] \leq e_i / \alpha_i . \tag{9}$$

If we choose α_i as $e_i / \alpha_i = 1 - p_i$, i.e., $\alpha_i = e_i/(1-p_i)$, then (8)-(9) become equivalent to the safety constraint (5) of the original model (2), (3), (5). In other words, the minimization of convex function $F(y,z)$ defined by (6) subject to (2), (3), and $z_i \geq 0$, $i=\overline{1,m}$, yields the desirable solution of the original model (2), (3), (5). Efficient computational procedures for solving stochastic minimax problems with objective functions defined as in (6) can be found in Ermoliev and Wets, 1988, Rockafellar and Uryasev, 2000. In particular, Rockafellar and Uryasev, 2000 discuss the applicability of linear programming methods in cases where the original model defined by a general probability distributions of ω can be sufficiently approximated by models with discrete probability distributions. This paper establishes also important connections between the minimization of (6)-type functions and Conditional-Value-at-Risk risk measure.

The minimization of function (6) can also be solved by a stochastic quasi-gradient method (Ermoliev and Wets, 1988). In applying this method to minimization of (6), the differentiability of $F(y)$ and any assumption on probability distribution of ω is not required. Also, the probability distribution of ω may only be given implicitly. For instance, only observations of random $d_i(\omega)$ and $a_{ik}(\omega)$ may be available or only a Monte Carlo procedure (“pseudo-sampling” simulation model as described in Section 2) is used to simulate supply and demand. In Section 2 and 3 we illustrate application of the rebalancing algorithm described in ANNEX 1 while the outlined stochastic allocation algorithm has been elaborated, e.g., in Fisher et al., 2007; Borodina et al., forthcoming. Impressive application of stochastic quasi-gradient methods in a form of adaptive Monte Carlo optimization can be found in Wang, 2010.