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

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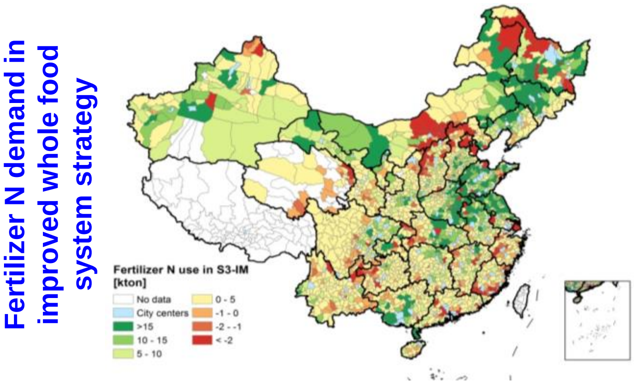
Abstract:

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 of >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 with in 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 re-location of livestock farms to areas with sufficient cropland.

Graphic abstract:



Introduction

Human pressures on the Earth-system have increased to unprecedented levels, with many of these pressures having severe impacts on the stability of 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 the boundaries. The biogeochemical flows of nitrogen (N) and phosphorus (P) have been considered to even reach a high-risk zone.[1-3] 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.[2,4-7] It has been estimated that the total N and P fertilizer input to agriculture need 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.[3,8] Most of the environmental effects of N and P become visible on the local to regional range,[5] 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 last decade,[9] and it faces serious water and air pollution due to low N and P use efficiencies.[10-11] The central government has set a 'zero increase target' for N and P fertilizer use between 2016 and 2020 to alleviate the environmental pollution.[12] Though 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,[13-15] improved livestock manure management, improved linking of crop production and livestock production, [16-17] diet manipulations and reduced food wastages [18-19]. 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.[20-21] 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.[22-23] If synthetic fertilizer is replaced by manure without consideration of the bioavailability of the manure, 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.[24] 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. [25-28] 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 county level, and thereby the potentials to reduce N and P fertilizer use in China. The updated NUtrient flows in Food chains, Environment

Re-sources use (NUFER) county model was used, which contains data and information of more than 2300 counties.[29-30] 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.

Material and methods

NUFER model

The modified NUFER-county model was used to quantify the N and P flows in the whole food chain.[29-30] 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,[19-20] 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 sub-module (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.[31]

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.[29-30] The NUFER-county model was further improved by including crop yield dependent biological N fixation for legume crops.[32]

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

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 above-ground production, and BGN is a multiplicative factor taking into account the contribution to total N_2 fixation of below-ground fixation associated with roots and nodules production as well as to rhizodeposition via exudates and decaying root cells and hyphae (dimensionless).[32]

The N and P losses via surface runoff, erosion and leaching were estimated as function of land use, precipitation, soil depth, soil type, temperature and soil texture and soil organic matter content at the county scale. The detailed method has been described in Zhao et al.[33] The data and parameters were derived from the Data Center for Resources and Environmental Sciences, Chinese Academy of Sciences (RESDC),[34] or estimated via the spatial interpolation methods applied by RESDC.

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 towards 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 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 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.

Description of strategies

Strategy S1: Balanced N and P fertilization in crop production (Fig 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 assure 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. The required synthetic N (or P) fertilizer input was estimated as follows:

$$\mathrm{Ic_{fertilizer}} = \sum_{i=1}^{n}\left[\left(\mathrm{Oc_{Main_{product,i}}} + \mathrm{Oc_{Straw,i}}\right) \times \mathrm{UF_{crop,i}}\right] + \mathrm{Oc_{Managed\ grass}} \times \mathrm{UF_{Managed\ grass}} - \mathrm{Ic_{Soilmineralization}} \quad (2)$$

Where $Ic_{\mathrm{fertilizer}}$ is the total input of synthetic N (or P) fertilizer, in kg N (or P); $\mathrm{Oc_{Main_{product,i}}}$ and $\mathrm{Oc_{Straw,i}}$ are the amounts of N (or P) in the main crop product and straw per county, respectively, in kg N (or P); $\mathrm{Oc_{Managed\ grass}}$ is the amount of N (or P) in harvested grass from managed grassland per county, in kg N (or P); $\mathrm{UF_{crop,i}}$ and $\mathrm{UF_{Managed\ grass}}$ are the uptake factors for crop species and grass, respectively (dimensionless) (Table S1); $Ic_{Soilmineralization}$ 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.[35] 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. [36] Further, we assumed that balanced fertilization reduced ammonia emission, runoff, erosion and leaching factors by 40% relative to the reference situation.[37-38] 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 still may occur.

Strategy S2: Balanced fertilization and improved nutrient accounting in the crop-livestock production sector (Fig 1b). A number of recent studies emphasized the need to recouple crop and livestock production. This would allow to increase nutrient recycling, and hence reduce the external new nutrient input in the agricultural system.[15,25,26] 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:

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

Where, $Ic_{deposition}$ 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 1c). The required N (or P) fertilizer input was estimated as follows:

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

Where $Ic_{food_{byproduct}}$ and Ic_{human_manure} are the N (or P) input via recycled food waste and human excreta, $AF_{food_{byproduct}}$ and AF_{human_manure} are the mineral fertilizer values of the treated (composted) food waste and human excreta (Table S2).

Strategy S1-IM: As S1, but with improved soil management and crop husbandry, including soil fertility management, erosion control, crop rotation, green manuring (Fig 1d). We assumed that these practices will lead to a considerable improvement of soil fertility.[36] As a result, net soil N and P mineralization will increase.

Strategy S2-IM: As S2, but now with improved soil management and emission mitigation in livestock production (Fig 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,[39] through a combination of measures, including acidification of slurry, covering slurry storages, and closed manure composting technologies.[40-42] 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 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 1f).[11] A new system will be built to collect human excretions instead go to 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.

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.

Results and Discussion

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 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 2).

This will lead to strong reduction of N losses, especially from the crop production (Fig 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 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.[43]

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 reduces 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 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.[16] In addition, there were accountable inputs via the return of crop straw and residues from other crops,[44] and atmospheric N deposition.[10]

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 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 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.

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 following enhanced nutrient management strategies (Fig 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 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.[45-46] 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.[22-23]

Improved nutrient management greatly reduces the losses of N and P from the food chain to the environment (Fig 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 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.

Improved nutrient management increases the N and P use efficiency in crop production, crop-livestock production and in the whole food chain (Fig 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 larges 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.[47] 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.[9] 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.[48-49] 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.[21, 50] Though some of these wastes are being used as animal feeds, most of the food wastes ends up in garbage burning installations or landfill sites.[51] 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.[16]

Spatial disconnection of nutrient supply and demand

There is a big 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. Though 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. [52] 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. [52] The provincial level results are showed in Fig S3, and are not in-depth described here.

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 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 5b). Further differences were introduced by excluding Xinjiang, Gansu and Tibet from the calculations, for which county level data was 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.[31]

For the S3-IM strategy, the mean manure N and P loadings per county are presented in Figure 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). This indicates that these amounts of manure N and P cannot be used effectively as a substitutes 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 6a). Situations were even worse for P (Fig 6b).

Similar but less extreme situations have been found at country level in a global study. Lassaletta et al.[53] 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.[27] 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.[54-55] An exception is perhaps the Netherlands, where the surplus manure P produced (about 25% of total P excretion) has to be exported, [56] 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.

Required synthetic N and P fertilizer input at the county level

The required synthetic fertilizer input per county and strategy is presented in Fig 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 S6-7. 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.[57-61] Largest inputs are required in the Northeast Plain, North China Plain, and the middle- and down- stream of the Yangtze River (Fig 7, S4). These are major grain, vegetable and fruits producing areas.[31] 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.[31]

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 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 re-locate livestock farms to other regions. There are several technologies for manure treatment, but economic costs are often high, such as the produce the struvite, incineration and closed continuous composting technologies.[45-46, 62]

The main uncertainty originated from the mineral fertilizer value of livestock manures and organic wastes, which were estimated to range from 0.10-1.0 (Table 1). The manures and wastes provide huge amounts of N and P compared with the N and P withdrawal with harvested crop (Fig 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. [22-23,35] 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 [63] and partly from European studies. [22-23]

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 2). The planetary boundaries for biogeochemical N and P flows at the global level have been estimated a 62 Tg year^{-1} for N and 6.2 Tg year^{-1} for P.[3] The total global inputs in 2012 were 150 Tg for N and 14 Tg for P.[9] If all the required reduction would have to come from synthetic fertilizers, 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.

Based on 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 as suggest 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 farmers and their advisors. A series of technologies and polices are needed to efficiently recycle manure.[16,25,61] Recently, demonstration programs have been established in 100 counties to boost manure recycling, and there are plans for another 200 counties.[62] In addition, zoonotic diseases 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.[11] 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 re-location of pig farms from south to north, to solve water pollution problems in the south, which has been not without side-effects.[64] 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.[65] 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 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-years 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.[65] 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.

Supporting Information:

The supplementary information (13 pages) contains the brief description about the definition of nutrient use efficiency, key parameters and reference list. Moreover, the SI includes graphs which illustrate the N and P flow of food chain in 2012, the distribution of sewage treatment plant, and required N and P input at the county level.

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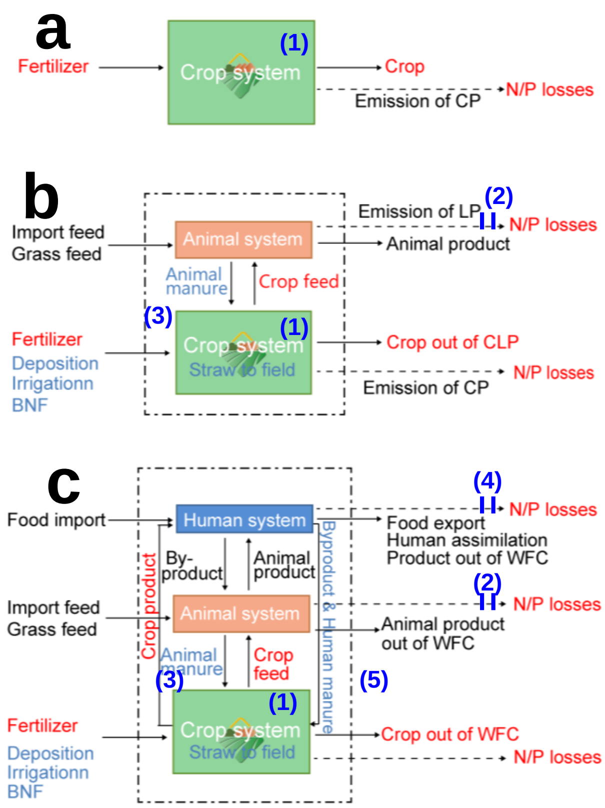

Fig 1. The system boundaries for the different strategies considered in this study: S1 and S1-IM (a), S2 and S2-IM (b) and S3 and S3-IM (c).

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 are representing 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; (5) New system to recycle human excretion and food waste.

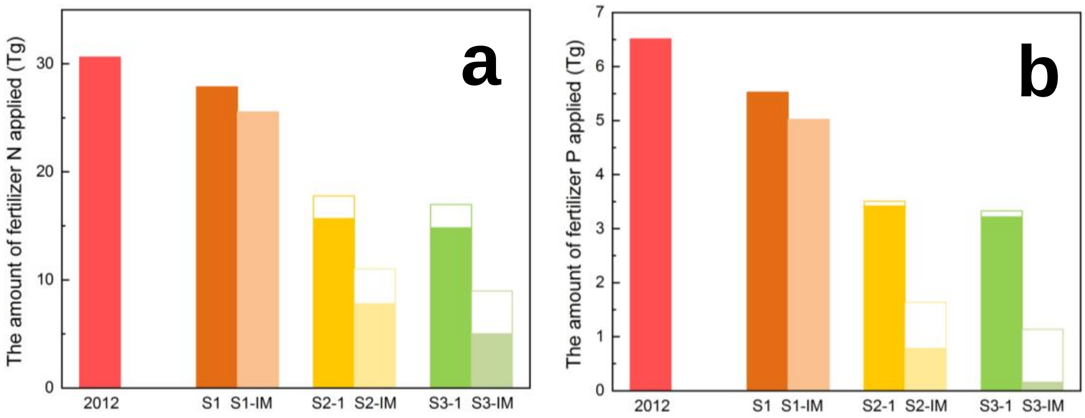

Fig 2. Inputs of synthetic nitrogen (N) fertilizer (a) and phosphorus (P) fertilizer (b) to Chinese agriculture in 2012, and the required inputs of synthetic N and P fertilizers for various strategies. The solid (filled) bars represent the required synthetic N and P fertilizer inputs, following assumptions and estimations at the national level. The blank top-up bars represent the estimated required inputs following assumptions and estimations at county level.

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.

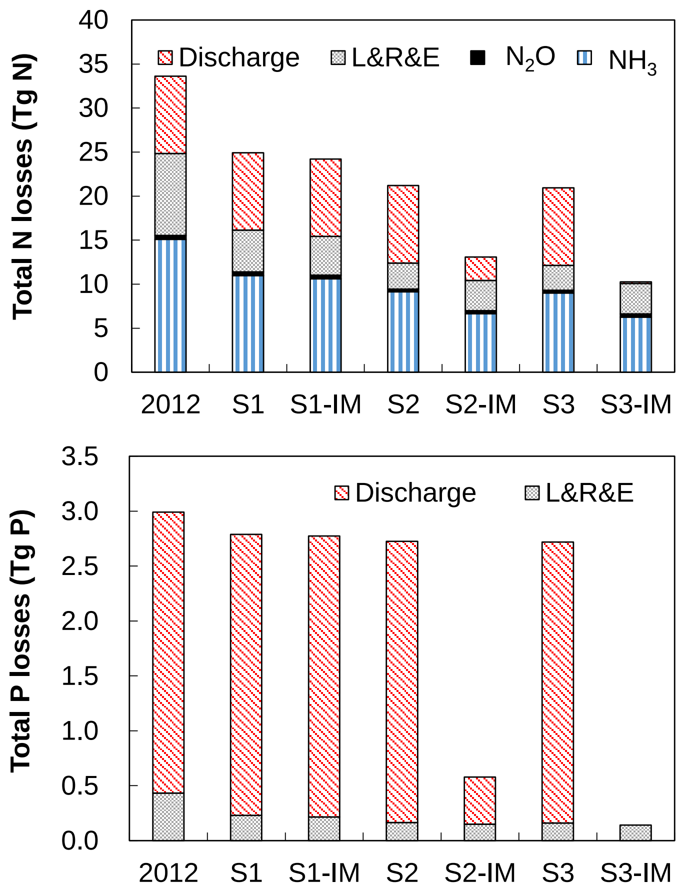

Fig 3. Total nitrogen (N) and phosphorus (P) losses from the whole food chain of different strategies at the national level in 2012.

Note: L&R&E is the leaching, runoff and erosion losses.

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.

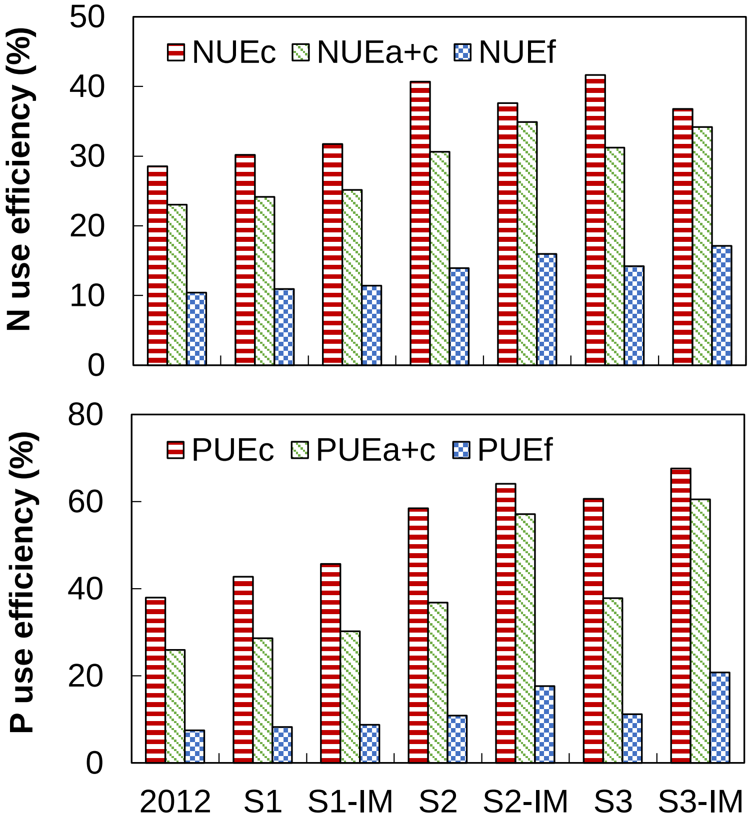

Fig 4. Nitrogen (N) and phosphorus (P) use efficiency in crop production (NUEc and PUEc, respectively), in crop-livestock production (NUEa and PUEa, respectively), and in the food chain (NUEf and PUEf, respectively) in 2012 and in 2050 for different strategies.

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.

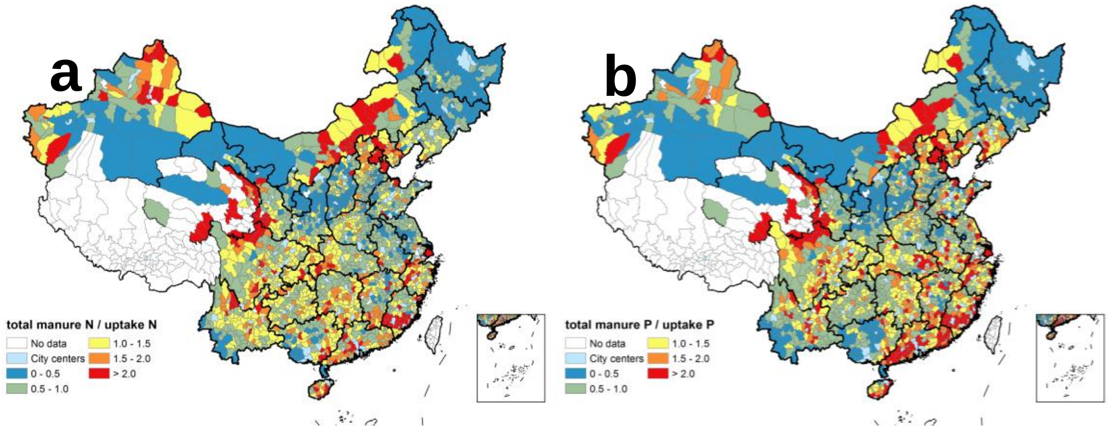

Fig 5. Map of the distribution of the manure N loading (a) and manure P loading (b) at county level in 2012. The manure N (or P) loading is defined as the ratio of the total excretions of N (or P) by livestock and humans and the N (or P) withdrawal with harvested crops.

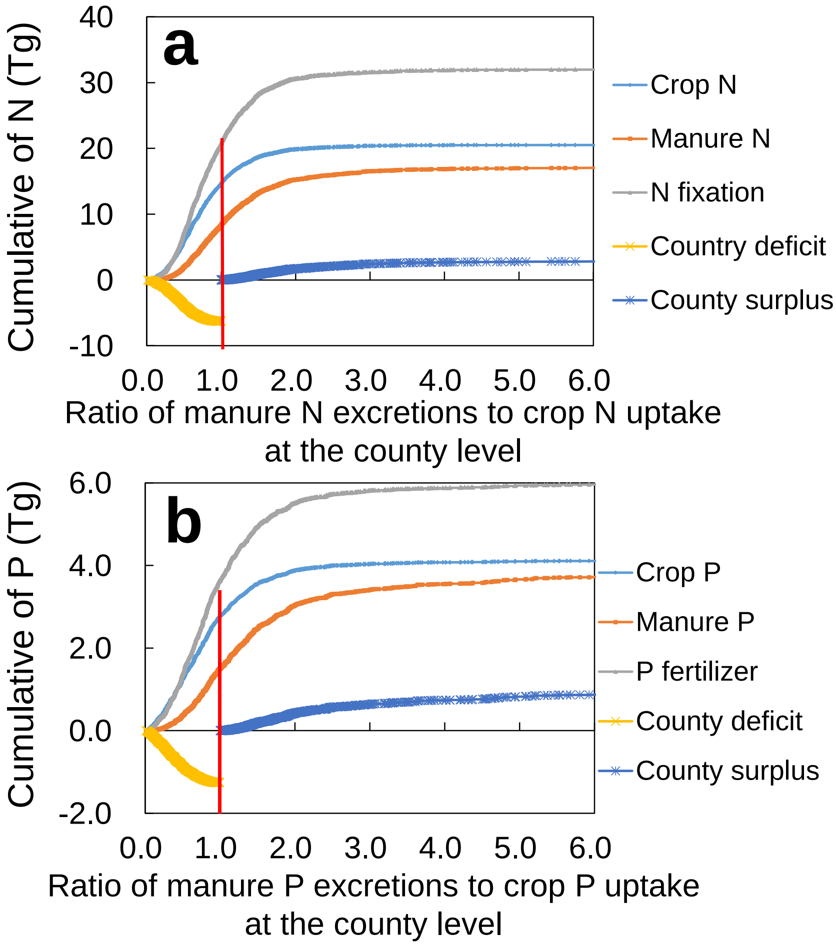

Fig 6. Cumulative distribution curves of N (or P) withdrawal in harvested crops, livestock N (or P) excreta, use of N (or P) fertilizer, and the surplus (or deficit) livestock N (or P) relative to the N (or P) withdrawal in harvest crops of counties in 2012.

Note: 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 showed 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.

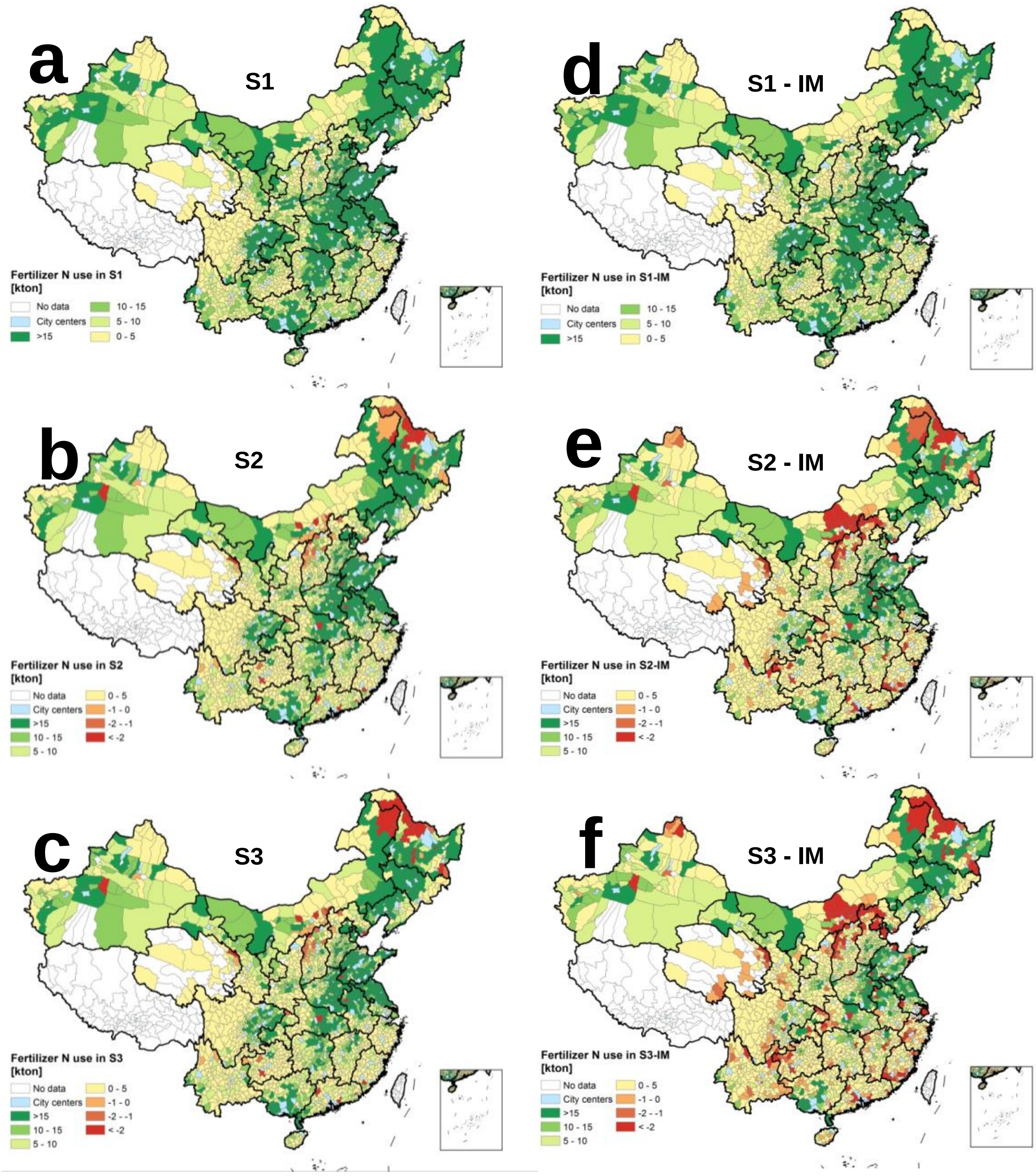

Fig 7. Mineral fertilizer (N) demand at the county level under the respective strategies (see Fig. 1 for definitions). Blue shades (negative numbers) designate areas where availability manure N already exceeds plant requirements.

Note: 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.

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.