
Zhu, Zhiping; Zhang, Xiuming; Dong, Hongmin; Wang, Sitong; Reis, Stefan; Li, Yue; Gu, Baojing. 2022. **Integrated livestock sector nitrogen pollution abatement measures could generate net benefits for human and ecosystem health in China.** *Nature Food*, 3 (2). 161-168. https://doi.org/10.1038/s43016-022-00462-6

Integrated livestock sector nitrogen pollution abatement measures could generate net benefits for human and ecosystem health in China

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Nearly one quarter of global meat production occurs in China, but a lack of detailed spatial livestock production data hinders ongoing pollution mitigation strategies. Here, we generate high-resolution maps of livestock systems in China using over 480,000 farm surveys from 2007 to 2017, finding that China produced more livestock protein with fewer animals and less total pollution impact through better breeding, feeding and manure management in large-scale livestock farms. Hotspots of production can be observed across the North China Plain, Northeastern China and the Sichuan Basin. The Clean Water Act reduced manure nutrient losses to water by one third, but with limited changes to methane and ammonia emissions. Integrated production and consumption abatement measures costing approximate US$ 6 billion could further reduce livestock pollution by 2050 – realizing benefits of up to US$ 30 billion due to avoided human health and ecosystem costs.

China is the largest livestock producer globally, accounting for 22% of global meat production [1]. Despite the important role for both food security and environmental impacts, the spatial distribution of livestock production is generally not well understood due to a lack of detailed spatial production data in China [2]. In contrast to the spatial distribution of croplands that can be derived from remote sensing [3], the distribution of livestock production can only be robustly based on surveys of livestock farms that are rare and costly. Without such survey data, it is difficult to determine the spatial patterns of pollutant emissions, such as ammonia (NH_3), which is crucial to the simulation of air pollution [4, 5]. Previous studies mainly estimated the distribution of livestock production through proxy variables such as rural human population in China [6]. However, this is only viable when livestock production is dominated by small-scale farms. With the increase of large-scale farms [7], it is essential to build accurate farm maps for the assessment of geospatial-related impacts from livestock production.

Livestock production not only affects food security and environmental pollution within China, but also exports impacts through international trade and global atmospheric circulation beyond China's territory [8, 9]. The development and implementation of effective abatement measures and policies would benefit from detailed, highly spatially resolved, maps e.g. on the implementation of local mitigation measures [2]. Fortunately, two agricultural pollution source censuses were conducted in 2007 and 2017 that covered all livestock farms including both smallholder and large-scale farms with precise locations (Extended Data Fig. 1). Based on these two censuses, we (1) generate high resolution livestock maps for China with 1 km × 1 km spatial resolution; (2) assess the performance of livestock production and the underlying driving forces over the period from 2007 to 2017; (3) quantify the contribution of livestock production to environmental pollution and identify mitigation potential.

Results

Distribution maps. The overall spatial patterns of livestock production (pig units, the definition can be found in the Methods section) were similar in 2007 and 2017, with several hotspots observed across the North China Plain (NCP), the middle of Northeastern China, Gansu province and the Sichuan Basin (Fig. 1). Ruminants are mainly reared in Northern China (Fig. 1a and 1d), especially dairy cattle, and are concentrated in a few small regions, mainly Hebei, Shanxi, Heilongjiang, Inner Mongolia and Xinjiang. Beef cattle and sheep/goats are primarily observed in Shandong, Henna, Yunnan and Sichuan (Extended Data Fig. 2 & 3). Generally, more forage and straw supplies available in North China explain the preference for ruminant production there. To contrast stable-based livestock farms, grazing animals are more commonly found in the North and Southwest China, e.g. Inner Mongolia, Xinjiang and Tibet.

Compared to ruminants, monogastric animals are found in both North and South China and are less concentrated in certain regions (Fig. 1b and 1e). North China Plain, Middle and Lower Yangtze River Plain (MLYRP), and Sichuan Basin are the three most important hotspots of monogastric livestock production in China. This is spatially associated with the distribution of croplands in China, especially for pigs in 2017, due to grain feeds mainly being derived from crop production and the comparatively low transport costs due to proximity [10]. Layer and broiler farms are more concentrated across the North China Plain, while pig farms are distributed more widely as they are typically substantially smaller than poultry units [7].

From 2007 to 2017, a substantial decrease in the number of livestock production hotspots could be found, especially in South China (Fig. 1c and 1f). To control water pollution, many pig and chicken farms in the region were closed and relocated to North China [11]. This reduced the overall spatial concentration of livestock farms with a decrease in former hotspot regions and an increase in regions that previously did not have substantial livestock production activities. Red meat and milk consumption are increasingly satisfied by imports, which contributed to a reduction in domestic production [1]. While a general reduction of livestock numbers was observed, the relative production efficiency per animal increased (Fig. 2 and 3), which offset the negative impact of animal number decline on total livestock production.

Better performance. Although livestock numbers decreased by 14% between 2007 and 2017, total livestock output production increased by 3% (Extended Data Fig. 4), suggesting that production per animal increased (Figs. 2 and 3). The proportion of large-scale farming increased from 31% to 45% between 2007 and 2017, and more efficient animal breeds and feed formulas are more commonly found in large-scale farming, both contributing to the better performance of livestock production before excretion [12]. Meanwhile, the decrease in the numbers of ruminants can also increase the overall performance of livestock production given their relatively low efficiency compared to monogastric animals (Extended Data Fig. 2 & 3). This led to reductions in both feed consumption and nitrogen (N) excretion, while resulting in an 8% increase in N use efficiency (NUE).

Once generated, different manure treatment methods lead to different fates of these livestock excretions over the study period (Extended Data Fig. 5). Manure in livestock farms was mainly cleaned through rinsing, producing a large amount of wastewater that was mostly discharged to surface water bodies directly, leading to substantial water pollution in 2007 [11]. To reduce water pollution, manure in livestock farms was mainly subjected to dry cleaning with limited water use by 2017, and a requirement was introduced for manure from large-scale livestock farms to be treated (Extended Data Fig. 6). Manure storage methods have also changed over the study period from air drying on the ground to liquid slurry form in open storage lagoons. These changes reduced pollutant discharge to water bodies by one-third as a consequence of the Clean Water Act entering into force in 2008 [7]. The national government invested over 770 million USD to subsidize setting up over 5,000 large-scale livestock farms with better facilities to collect wastewater from surfaces and improved storage in open lagoons or treated, while solid manure storage and treatment areas were covered and thus protected from rain and leakage [13].

The decrease of N losses to water bodies also led to a 36% reduction of nitrous oxide (N_2O) emissions due to nitrification and denitrification processes with less water and total excretion N (Fig. 3). But while manure treatment reduced N losses to water bodies, it slightly changed losses to air through NH_3 as well as generating additional methane (CH_4) emissions. Due to the increase of NUE, total manure N was reduced, however, which led to an 8% reduction of NH_3 emissions overall (Extended Data Fig. 7). However, management options aimed at controlling water pollution resulted in small changes to the loss pathway via NH_3 emission to air after manure was generated. Furthermore, it increases the CH_4 emission from 210 to 217 Tg carbon dioxide equivalent (CO_{2eq}) due to the increase of liquid manure storage in open lagoons in large scale farm.

To increase the reuse of manure, the national government implemented policies to redistribute livestock farms nationally, based on where sufficient cropland areas were available to use locally produced manure [13]. North China is home to a larger proportion of croplands and fewer water bodies, leading to a redistribution of pig production from South to North China [11, 12]. The manure recycling ratio grew from less than 50% in 2007 to over 70% in 2017 (Fig. 4). However, the total N recycling ratio was only around 40% in 2017, although it increased from around 30% in 2007 (Extended Data Fig. 8). The value is much lower than that estimated in previous studies, which estimated the manure N recycling rate at higher than 60% [14]. This inconsistency can be mainly explained by N losses through gaseous NH_3 emissions and leaching to groundwater (Fig. 2 and 3).

Despite the solid part of manure being recycled, the open design of manure storage did not prevent nutrient losses to air and leaching during manure storage, before application to fields. This highlights that for effective control of N losses at all stages, it is vital to fully account for losses at every step of the N cascade [15].

Environmental and climate impacts. NH_3 and greenhouse gas (GHG) emissions (including CH_4 and N_2O (Extended Data Fig. 9)), as well as N losses to water bodies from livestock production have substantial impacts on human and ecosystem health and contribute to global climate change (Fig. 5). To estimate the environmental and climate impacts of livestock production in China, we included data on all animal categories at the county scale, except for the six main animal categories included in the census. Damages of N losses and GHG emissions from livestock production in China were estimated for the year 2017 (Table S1). Total damage costs were estimated to be about 60 billion USD, with three-quarters attributable to NH_3 emissions, followed by 22% from N losses to water bodies through runoff and leaching, and the remainder related to GHG emissions.

NH_3 emissions from livestock production are a major precursor of fine particles ($PM_{2.5}$) pollution in China, especially in winter when NH_3 emissions from croplands are limited [4]. $PM_{2.5}$ pollution can lead to respiratory and cardio-pulmonary health effects, with total health damage costs estimated at 14 billion USD attributable to NH_3 emissions from livestock production (Fig. S1a). Furthermore, air pollutants can deposit to terrestrial ecosystems, resulting in such as soil acidification, eutrophication. These changes reduce ecosystem services with total estimated damage in China of 37 billion USD (Fig. S1b). Other than human health and ecosystem services, NH_3 emissions can also contribute to cooling the climate through aerosol formation, as well as increasing carbon sequestration via nutrient N deposition, amounting to an estimated benefit of 6 billion USD overall (Fig. S1c).

GHG emissions can also damage human and ecosystem health indirectly and bring climate impact directly [16], with total damage estimated at 2 billion USD (Fig. S1d-f). Human health and ecosystem damage due to GHG emissions is less than 0.2 billion USD given their small emission amounts and the weak effect on human health and ecosystem functions. GHG emissions bring about 1.7 billion USD damages to climate, referring to ozone depletion and global warming. Nitrate concentrations in drinking water are associated with cancer risks of the digestive system, and it is also contributing to eutrophication and harmful algae bloom in freshwater and coastal ecosystems [17]. Overall damage costs related to water pollution were estimated at 14 billion USD, with ecosystem damages constituting over 85% of this value.

Pig production is the largest source of overall damage, amounting to 23 billion USD, followed by sheep/goat production estimated at 14 billion USD, and other major animal categories (cattle, layers, broiler, dairy cows), which contribute about 3-8 billion USD to overall damages. Other than these major animals included in the agricultural census, other animals, such as ducks and horses contribute an estimated 6 billion USD damages in total.

Cost and benefit to abate livestock pollution. Reduction of N loss and GHG emission would lead to societal benefits under the three major abatement scenarios: Diet (D), NUE (N) and Recycle (R), and the combined scenario Combo (C) that integrated these three scenarios (Fig. 6). Detailed information on these abatement scenarios could be found in the Methods section (Table S3). The Combo scenario can achieve about 30 billion USD benefit per year in 2030, which would double by 2050, while the implementation cost of all measures included in the scenario amounting to only around one fifth of these values in the respective years. It suggests that from a socio-economic viewpoint, abatement of livestock pollution would yield a substantial net benefit (Table S2). However, the benefits are likely gained by other parts of the society than those carrying the costs of implementation normally farmers or governments [18]. It suggests that incentive to farmers is crucial for the implementation of pollution control measures since the benefits are for the whole society.

However, with a projected future increase in livestock production, while these measures can reduce GHG emissions compared to the baseline scenario (Business As Usual - BAU), total GHG emissions by 2050 are at the same level as in 2017. This suggests that the focus of current abatement measures is primarily on NH_3 abatement and does not adequately take into account GHG emission reduction. The Clean Air Act explicitly identifies NH_3 emission reduction as an important target to achieve [19]. The situation for N runoff reduction is similar. The Clean Water Act contributed to the reduction of N losses to water bodies from livestock farms and was influenced by the Tai Lake algal bloom event in 2007 [20]. Further reduction of N losses to water bodies beyond what has already been achieved by 2017 will require additional efforts. In recent years, the central government has invested over 3 billion USD to increase manure recycling with the aim of reducing livestock pollution in over 600 counties in China [13]. These governmental campaigns highlighted the feasibility of livestock pollution controls and encouraged more investment in future pollution control for livestock production. However, these pollution controls are only achieved by government subsidies to farmers who bear the costs while the rest of the society primarily reaps the benefits [18].

Discussion

The distribution maps developed in this study are substantially different from previous global and China-specific studies [6, 21], which had identified hotspots of livestock production mainly in South China, especially Southwestern China. In contrast, our study indicates that apart from the Sichuan Basin, livestock production is rarely found in Southwestern China, with the dominant land use being forest [22]. While a few scattered livestock farms are present in Southeast China, our assessment did not find evidence for a widespread distribution of livestock farms across the whole of South China in contrast to previous studies [21]. Hilly and mountainous areas are commonly found in this region, which are typically not suitable for livestock production. A lack of grain production from local crops would also result in prohibitively high feed transportation costs in these regions [10]. Spatial misrepresentation of livestock maps may lead to low efficiency on high-level policy making, while the Clean Air Act and Clean Water Act both identify livestock production as an important pollution source [11, 20].

Manure recycling is considered the most efficient way to both reduce livestock pollution and promote crop production with less synthetic fertilizer use [10]. Reducing the numbers of livestock farms in hotspots regions where manure production has exceeded the carrying capacity of croplands while increasing numbers of livestock farms in the non-hotspots region to promote the recycling of manure to croplands. This paper provides the high-resolution maps of livestock farms that can be used for the recoupling of livestock and croplands at an unprecedented scale to reduce the storage and transport cost of manure. Newly-built livestock farms must consider their spatial co-location with the distribution of croplands to increase the potential of manure recycling. Relocation of large farms is undertaken considering strict environmental standards on livestock pollution control. This is a cost-effective way for a long-term run, which could reduce the transportation cost of manure. For smallholder farms, manure management is more challenging due to the more common occurrence of decoupling of livestock and croplands in areas with smallholders in rural China [12]. Enlarging cropland farm size to promote the recoupling of livestock and croplands is an important way, which could be implemented on village scale [23].

Better breeding and feeding are always beneficial to reduce the pollution from livestock production. Given the high costs for storage and application of manure, there still is room for manure recycling improvements. Compared to chemical fertilizers, manure has a lower nutrient content per unit weight, and thus more effort is required for its application, either in solid or liquid form. Meanwhile, antibiotics and heavy metals are commonly found in livestock manure through addition from animal feed and medical treatment. Without additional measures, long-term application of animal manure may lead to cropland contamination. Therefore, legislation on the safe standard of the use of antibiotics and heavy metals in livestock feed must be set up since it is difficult to remove these pollutants, especially for heavy metals, once they are released into the environment.

Advanced technologies and facilities to improve storage and application of manure should be development priorities, such as closed systems for manure storage demonstrated in the NCP where both intensive livestock production and substantial air pollution challenges occur [19]. These manure storage, treatment and application approaches should be designed for both small- and large-scale of livestock farms. A key Research and Development (R&D) program of the Ministry of Science and Technology of China has supported the development of new technologies to reduce NH_3 emissions from livestock production through better storage, treatment and application of manure to cropland during the 13th Five Year Plan. The outcomes of the R&D program have been successfully tested in several demonstration sites within livestock production hotspots and would help to promote manure recycling in the following years.

Previously, management has mainly focused on the production side with the sole aim to produce more food with less pollution, while little attention was dedicated to the consumption side. Food waste and overconsumption are also important drivers of livestock pollution. Measures optimizing human diet based on nutrient recommendations would reduce livestock pollution fundamentally. However, neither production or consumption side measures are solely sufficient to control livestock pollution. Integrated measures combining both production and consumption aspects are crucial. In addition, previous livestock pollution mitigation measures in China did not typically consider synergies and co-benefits of GHG emissions reductions, with some measures introduced with the aim of reducing NH_3 emissions having the potential to increase GHG emissions [20]. Therefore, co-benefits and unintended consequences of measures designed for the reduction of NH_3 and GHG emissions will facilitate the implementation of net-beneficial, integrated abatement strategies in China.

Methods

Data sources

Data on livestock numbers in both large-scale and small-scale farms and the pollution they generate per animal were collected in agricultural pollution source censuses across China in 2007 and 2017. In total, approximately 100,000 and 380,000 large-scale farms were surveyed in 2007 and 2017, respectively (Extended Data Fig. 1). The geographical coordinates of each large-scale farm were recorded in the censuses and used to generate high resolution distribution maps of large-scale farms. For small-scale farms, statistical surveys were conducted at a county scale to record the total number of each animal type. The spatial distribution of small-scale farms is highly correlated with that of the rural population density distribution in each county, hence the rural population distribution is used to allocate the total number of animals from small-scale farms to each 1 km × 1km grid cell in each county. The threshold numbers defining the category of "large-scale" farms are larger than 50, 100, 500, 500, 10,000, and 2,000 for beef cattle (slaughtered), dairy cattle (stock), pig (slaughtered), sheep/goat (slaughtered), broiler (slaughtered) and laying hen (stock). All numbers are converted to pig units when comparing animal numbers. 1 dairy cattle = 10 pigs; 1 beef cattle = 5 pigs; 3 sheep/goat = 1 pig; 15 layer chickens = 1 pig; 60 broiler chickens = 1 pig. No statistical method was used to predetermine sample size. No data were excluded from the analyses; The experiments were not randomized; The Investigators were not blinded to allocation during experiments and outcome assessment.

Emission calculation

To determine excretion generated per animal, approximately 200 farms were selected for monitoring across China based on the distribution of farms of different livestock species including pig, layer, broiler, beef and dairy cattle in both census years (Extended Data Fig. 10). Given the general stable rate of excretion generated by sheep and goats, they are not included in the monitoring systems and recommended values from the Ministry of Agriculture and Rural Affairs of China were applied. To quantify excretion production at different feeding stages, feces and urine from each animal were collected across all four seasons, covering five to seven days in each season. At each feeding stage, five animals (25 animals for chickens, respectively) with similar body weight and age were selected for detailed analysis and fed in separate enclosures. All feces and urine generated were collected 24 hours a day, then weighed and analyzed for nutrient contents. To monitor the efficiency of manure treatment measures, the emissions from excreted manure before and after the treatment were monitored. The results of different emission factors can be found in supplementary data.

Based on the information collected from the monitoring systems described above, emission factors and activity rates were determined for each animal type in different regions as follows. Amount of feces produced during the life cycle of an animal:

$$\mathrm{QF} = \sum QF_i \times T_i \qquad (1)$$

QF (kg/head) is the total amount of feces produced during all feeding stages of a certain animal; QF_i (kg/head/day) is the amount of feces produced per day in the i^{th} feeding stage of this animal; T_i (day) is the number of feeding days in the i^{th} feeding stage of this animal.

Amount of urine produced during the life cycle of an animal:

$$\mathrm{QU} = \sum QU_i \times T_i \qquad (2)$$

QU (L/head) is the total amount of urine produced during all feeding stages of a certain animal; QU_i (L/head/day) is the amount of urine produced per day in the i^{th} feeding stage of this animal; T_i (day) is the number of feeding days in the i^{th} feeding stage of this animal.

Amount of pollutant in excretion during a certain stage in a day:

$$FP_{i,j} = QF_i \times CF_{i,j} + QU_i \times CU_{i,j} \qquad (3)$$

$FP_{i,j}$ (mg/head/day) is the daily production amount of the j^{th} pollutant in the feces and urine of a certain animal in the i^{th} feeding stage; QF_i (kg/head/day) is the amount of feces produced per day in the i^{th} feeding stage of this animal; $CF_{i,j}$ (mg/kg) is the concentration of the j^{th} pollutant in the feces of this animal in the i^{th} feeding stage; QU_i (L/head/day) is the amount of urine produced per day in the i^{th} feeding stage of this animal; $CU_{i,j}$ (mg/L) is the concentration of the j^{th} pollutant in the urine of this animal in the i^{th} feeding stage.

Amount of pollutant produced during the life cycle of an animal:

$$QFP_j = \sum_i FP_{i,j} \times T_i \qquad (4)$$

QFP_j (mg/head) is the total production amount of the j^{th} pollutant in the feces and urine of a certain animal; $FP_{i,j}$ (mg/head/day) is the daily amount of the j^{th} pollutant in the feces and urine of this animal in the i^{th} feeding stage; T_i (day) is the number of feeding days in the i^{th} feeding stage of this animal. Feeding days of pig and sheep/goat are calculated according to the slaughtered period with a life cycle of 165 days including 45 days of nursery and 120 days of fattening. Feeding days of dairy cattle are 365 weighted based on age, farm calf: young cattle: lactating cow = 15:30:55. Feeding days of beef cattle is 365 weighted based on farm calf: fattening cattle: cow = 20:40:40. Feeding days of laying hens are 365 weighted based on age, chick: laying hens = 20:80. Feeding days of broilers are 60 days.

Amount of daily pollutant emission of an animal:

$$FD_{i,j} = \left\{ \left[QF_i \times CF_{i,j} \times \left(1 - \eta_F\right) + QU_i \times CU_{i,j}\right] \times \left(\frac{100 - \sum T_k}{100}\right) \times \prod_t \frac{\left(100 - R_{t,j}\right)}{100} + QF_i \times CF_{i,j} \times \eta_F \times \left(1 - \eta_U\right) \right\} \qquad (5)$$

$FD_{i,j}$ (mg/head/day) is the daily emission of the j^{th} pollutant in the feces and urine of a certain animal in the i^{th} feeding stage; QF_i (kg/head/day) is the amount of feces produced per day in the i^{th} feeding stage of this animal; $CF_{i,j}$ (mg/kg) is the concentration of the j^{th} pollutant in the feces of this animal in the i^{th} feeding stage; η_F (%) is the collection ratio of feces; QU_i (L/head/day) is the amount of urine produced per day in the i^{th} feeding stage of this animal; $CU_{i,j}$ (mg/L) is the concentration of the j^{th} pollutant in the urine of this animal in the i^{th} feeding stage; T_k (%) is the k^{th} reuse ratio of excretion; $R_{t,j}$ (%) is the removal ratio of the j^{th} pollutant with the t^{th} treatment measure; η_U (%) is the total resource use efficiency of feces.

Amount of total pollutant emission of an animal within a whole life cycle:

$$QFD_j = \sum_i FD_{i,j} \times T_i / 1000 \qquad (6)$$

QFD_j (g/head) is the total emission of the j^{th} pollutant of a certain animal; $FD_{i,j}$ (mg/head/day) is the total amount of the j^{th} pollutant of this animal in the i^{th} feeding stage; T_i (day) is the number of feeding days in the i^{th} feeding stage of this animal.

Nitrogen balance calculation

Based on the emission monitoring, the Coupled Human And Natural Systems (CHANS) model[20, 24, 25] is applied to calculate the system N balance.

$$N_{input} = N_{fer} + N_{feed} + N_{forage} \quad (7)$$

$$N_{output} = N_{human} + N_{manure} + N_{gas} + N_{water} \quad (8)$$

$$NUE = N_{human}/N_{input} \quad (9)$$

N_{input} is the total N input to the livestock system, including N fertilizer (N_{fer} used for straw ammonization for livestock system), grain and straw feed (N_{feed}) and forage (N_{forage}). N_{output} is the total N output from the livestock system, including livestock products for human consumption (N_{human}), manure recycle to croplands and grassland (N_{manure}), NH_3 and N_2O emission (N_{gas}) and N losses to water bodies through runoff and leaching (N_{water}). *NUE* is N use efficiency. More details of the CHANS model can be found in Table S4, Figure S2 and Gu et al[24, 25] and Zhang et al[20].

Potential to reduce N losses to air and water

Adoption of appropriate mitigation measures will reduce N losses from livestock production to the environment. The mitigation potential of N losses is estimated based on the mitigation efficiency of selected mitigation options for different animal type and region and the livestock N mass balance integrated with the CHANS model, as showed in Eq. (10)

$$\Delta E_{r,n} = \sum_m A_{r,m} \times \left[EF_{r,m,n} \times \eta_{r,m,o} \times X_{r,m,o}\right] \quad (10)$$

Where *r* represents the region; *m* represents the animal type; *n* represents the form of N losses (NH_3, NO_x, N_2O, N leaching and runoff) from livestock production; *o* represents the specific mitigation options; $\Delta E_{r,n}$ represents the reduction of Nr loss in region *r;* $A_{r,m}$ is the livestock population; $EF_{r,m,n}$ is the corresponding uncontrolled emission factor; $\eta_{r,m,o}$is the specific abatement efficacy; $X_{r,m,o}$ is the implementation rate of the abatement technique or options.

Cost and benefit analysis

Implementation costs. The implementation cost of reducing N losses by improved management for livestock production is defined as the social expenditure (the sum of investment costs and operation costs) for implementation of the best-fitted measures to reduce N losses from livestock production. Here we mainly refer to the database and methodology of cost-effectiveness assessments from the online Greenhouse Gas and Air Pollution Interactions and Synergies (GAINS) model (https://gains.iiasa.ac.at/models/index.html) to calculate national-level abatement costs. China-specific livestock conditions and farming practices have been considered in GAINS by taking into account Chinese labor costs, energy prices, farm size and costs of by-products, etc. All cost data from the model calculations are adjusted by the purchasing power parity (PPP) index and measured in constant 2017 US$ for this study. A detailed description of the GAINS model and cost calculation could be found in Klimont et al [26, 27]. The annual implementation cost (IC$_{r,n}$) in China is calculated as:

$$IC_{r,n} = \Delta E_{r,n} \times UC_{r,n} \quad (11)$$

where $UC_{i,n}$ represents the unit abatement cost of the best-fitted mitigation option to reduce livestock N loss in China, which is derived from the online GAINS model database and adjusted according to region-specific farming practices.

Societal benefits. The societal benefits (SOC_r) of mitigating N pollution from livestock production (Table S2) is defined as the sum of avoided damage cost for human health

(HH_r), ecosystem health (EH_r), GHG reduction (GHG_r, e.g., CH_4 reduction) and climate effect ($Climate_r$, e.g., climate warming due to reduction of aerosol) as shown in Eq. (12):

$$SOC_r = HH_r + EH_r + GHG_r - Climate_r \qquad (12)$$

Ecosystem benefits. A number of US and EU studies have examined the damage cost of N_r effects on the ecosystems [28-33], currently we do not have costs and benefits data established for other nations of the world. For this reason, we assume the unit N_r damage costs (Table S1) to the ecosystem in the EU and USA are also applicable to other countries after correction for differences in the willingness to pay (WTP) for ecosystem services and Purchasing Power Parity (PPP) to assess the benefits and trade-offs associated with N-related management actions for different regions, as shown in Eq. (13)

$$EH_r = \sum_n \Delta E_{r,n} \times \partial_{EU,n} \times \frac{WTP_r}{WTP_{EU}} \times \frac{PPP_{China}}{PPP_{EU}} \qquad (13)$$

where $\partial_{EU,n}$ is the estimated unit ecosystem damage cost of Nr emission in Europe based on the European N Assessment [30, 34]; WTP_r and WTP_{EU} are the values of the willingness to pay (WTP) for ecosystem service in region *r* and Europe; PPP_{China} and PPP_{EU} stand for the PPP of China and the EU.

Health benefits. For health benefits, we derived unit health damage cost of N_r emissions in China based on the cause-specific integrated exposure-response (IER) functions elaborated in previous studies [20, 35]. The IER functions are derived with the help of epidemiological data that estimate the relative mortality risk from exposure to $PM_{2.5}$ across different world regions [36]. A detailed description of the health damages attributed to air pollution ($PM_{2.5}$) and water pollution due to N_r emission could be found in the World Bank report and the GBD website (http://ghdx.healthdata.org/). The calculation of health benefits from livestock N management is shown in Eq. (14):

$$HH_r = \sum_n \Delta E_{r,n} \times HCost_{r,n} \qquad (14)$$

Where $HCost_{r,n}$ is the unit health cost of Nr losses in region *r*.

GHG benefits. The GHG benefit refers to the benefits of GHG (N_2O and CH_4) reductions due to the implementation of improved N management.

$$GHG_r = \Delta E_{GHG,r} \times GCost_r \qquad (15)$$

Where $\Delta E_{GHG,r}$ is the reduction of GHG emissions in Carbon dioxide equivalent (CO_2-eq) due to the improved livestock management in region *r*, which include the N_2O and CH_4 reduction; $GCost_r$ is the unit mitigation cost of GHG emissions in carbon price in region *r*.

Climate impacts. NH_3 emission is reported to have a cooling effect on the climate [37].The climate impact of improved N management is assessed as showed in Eq. (16):

$$Climate_r = \sum_n \Delta E_{r,n} \times CCost_{r,n} \qquad (16)$$

Where $\Delta E_{r,n}$ represents the reduction of Nr loss in region *r*. $CCost_{r,n}$ represents the unit damage cost of Nr reduction to the climate in US $ per kg N (Table S2).

Scenario analyses

To explore the mitigation strategy and pathways of livestock pollution in the future, the CHANS model was employed to conduct systematic and comprehensive analyses of livestock N emissions, fluxes and environmental fates [24]. Based on current policy, action and programs for livestock production and future social-economic development prediction, this study generated a comprehensive business-as-usual (BAU) scenario as a base case to evaluate the potential N_r losses and their environmental effects. Against this base case, four different abatement scenarios (DIET, NUE, REC and COMBINED) with corresponding packages of mitigation measures (detailed description in Table S3) were integrated into the CHANS model to quantify resulting livestock N budgets and identify the reduction potential for N losses in China. Human population numbers and per capita gross domestic product (PGDP) are assumed to remain constant in all five scenarios while other parameters, such as human diet structure, livestock NUE, animal populations, and feed production will vary among scenarios. Details on the data sources, prediction methods and parameters can be found in Table S3 and Zhang et al [20]. It should be noted that optimizing human diet structure as a non-technical measure was also included in the scenario analysis to obtain a more comprehensive assessment of the mitigation potential and pathways.

Data availability

Data supporting the findings of this study are available within the article, source data file and its supplementary information files, or are available from the corresponding author upon reasonable request.

Code availability

No Code is used in this research. The spatial analysis is run in ArcGIS v.10.6.

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Acknowledgments

This study was supported by the National Natural Science Foundation of China (42061124001, 41822701, and 41773068) and International Science & Technology Innovation Program of Chinese Academy of Agricultural Sciences (CAASTIP).

Author contributions

Z.Z., H.D. and B.G. designed the study. B.G. performed the research. X.Z. and S.W. analyzed economic related data and prepared the distribution maps. H.D. and Z.Z. provided the census data and help to interpret the results. B.G. wrote the paper, S.R. revised the paper and all other authors contributed to the discussion of the paper.

Competing interests

The authors declare no competing interests.

Figure legends

Fig. 1 | Distribution of livestock production in China on county scale. All numbers are converted to pig units. 1 dairy cattle = 10 pigs; 1 beef cattle = 5 pigs; 3 sheep/goat = 1 pig; 15 layer chickens = 1 pig; 60 broiler chickens = 1 pig. Ruminant includes cattle and sheep/goats, and monogastric animals include pigs and chickens here. Other animals are not included in the maps due to data limitations. A distribution map with 1 km×1 km resolution can be found in SI, derived from the first (2007) and second (2017) agricultural pollution source census with over 480,000 livestock farms (Extended Data Fig. 1). Base map is applied without endorsement from GADM data (https://gadm.org/).

Fig. 2 | Changes of N balance of livestock system from 2007 to 2017 in China. Due to data limitation, livestock species only includes cattle, sheep/goat, pig and chickens, which account for about 90% of total livestock protein produced. Others refer to unknows N losses such as N_2 emission through denitrification. Unit, Tg.

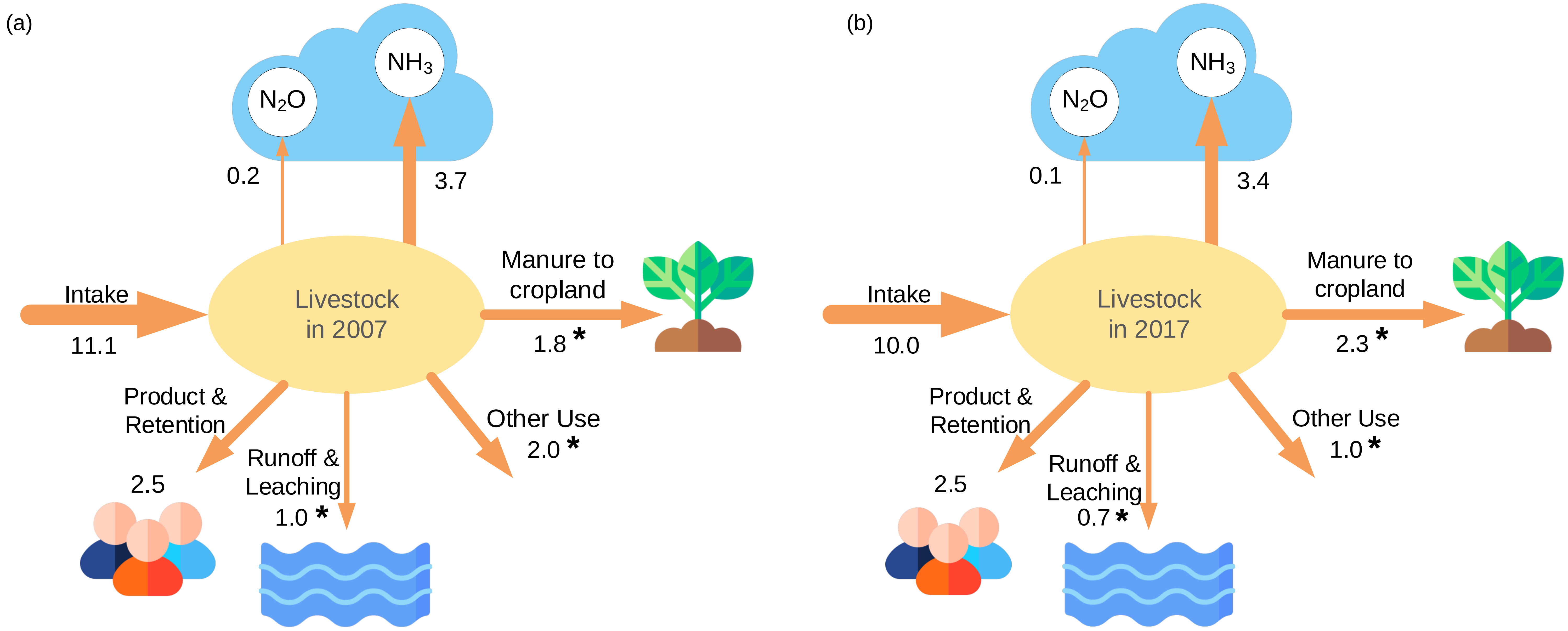
Fig. 3 | Changes of livestock system performance from 2007 to 2017. Production refer to livestock products such as meat and milk. Large-scale share refers to the ratio of animals raised in large-scale farms. Livestock unit refers to total animal numbers counted in pig units.

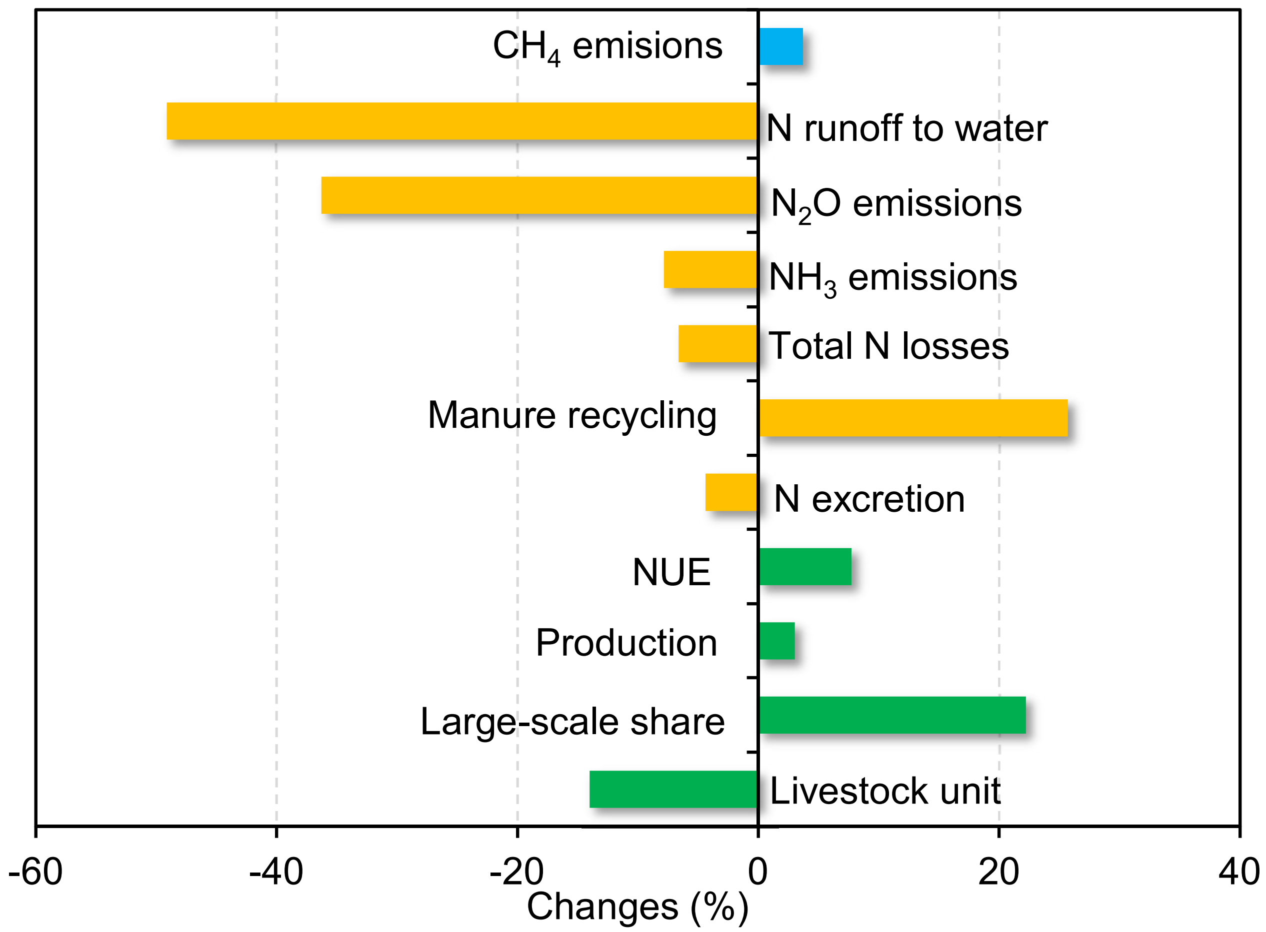
Fig. 4 | Manure recycling to croplands. (a) The ratio of manure recycling to cropland in 2017; (b) Ratio of total N derived from excretion recycling to cropland in 2017; (c) Comparison of manure recycle ratio in 2007 and 2017; (d) Comparison of N loss to air and water in 2007 and 2017. The error bars represent the standard error of estimates. The base map is applied without endorsement from GADM data (https://gadm.org/).

Fig. 5 | Health, ecosystem and climate effects of livestock pollution in 2017 in China. Uncertainty level of the health and environmental impact assessment by pollution type is indicated by the error bars, which are estimated by the Monte Carlo simulation (1000 runs). The negative value of climate damage cost represents the benefit of NH_3 emission. Detailed spatial distribution of the health and environmental impact by animal type could be found in Fig. S1.

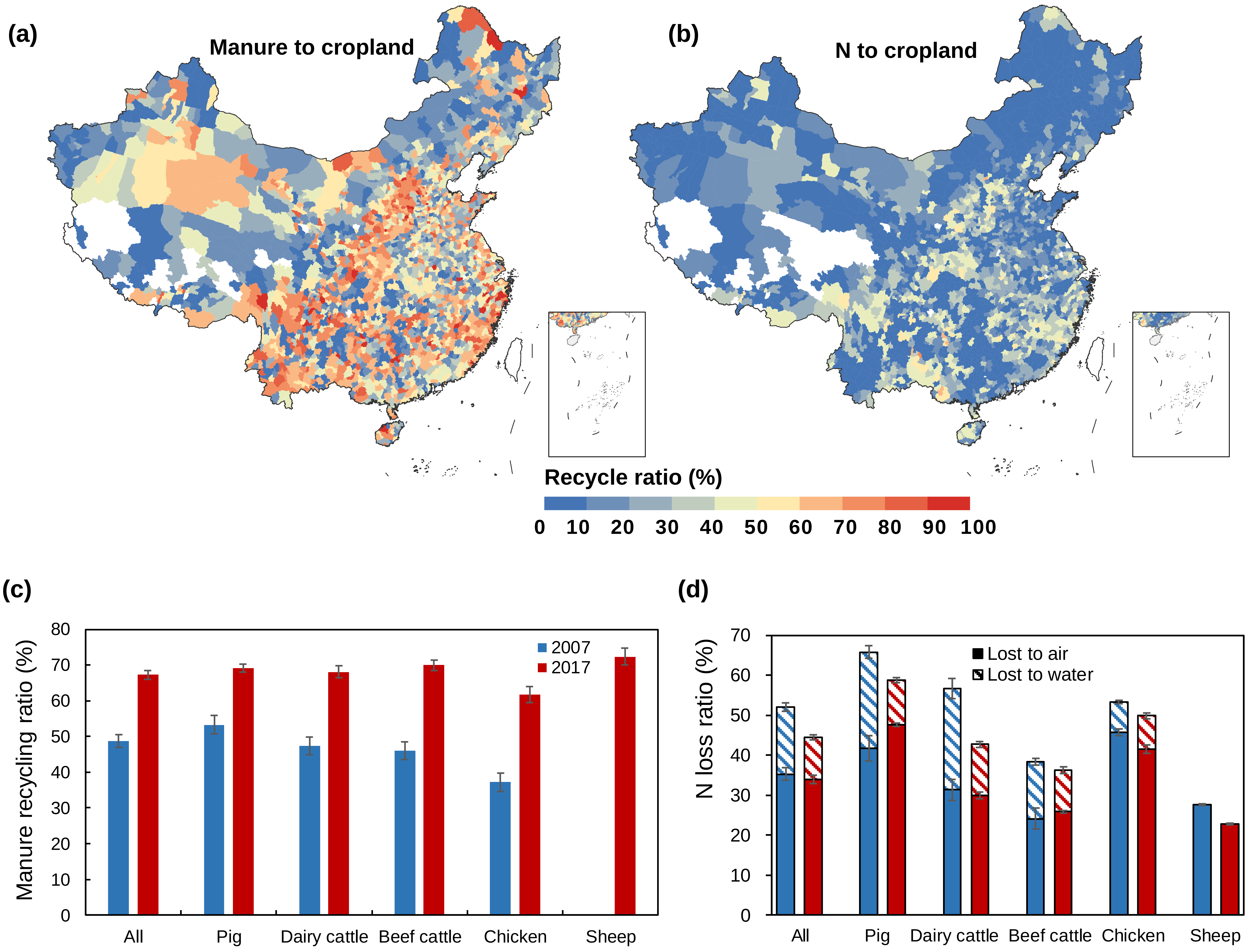
Fig. 6 | Future scenario of livestock pollution in China. (a) NH_3 emission; (b) GHG emission; (c) N loss to water; (d) cost and benefit to abate livestock pollution. B, Business as usual; D, diet; N, N use efficiency; R, Manure recycling; C, D+N+R. Shaded areas in (a)-(c) and error bars in (d) indicate the uncertainty level (with 95% confidence limits) of mitigation cost and benefits. Monte Carlo simulation (n=1000) is performed based on the data derived from the Second National Census of agricultural Pollution Sources (involving 2,981 counties/districts and 378,800 animal farm surveys).

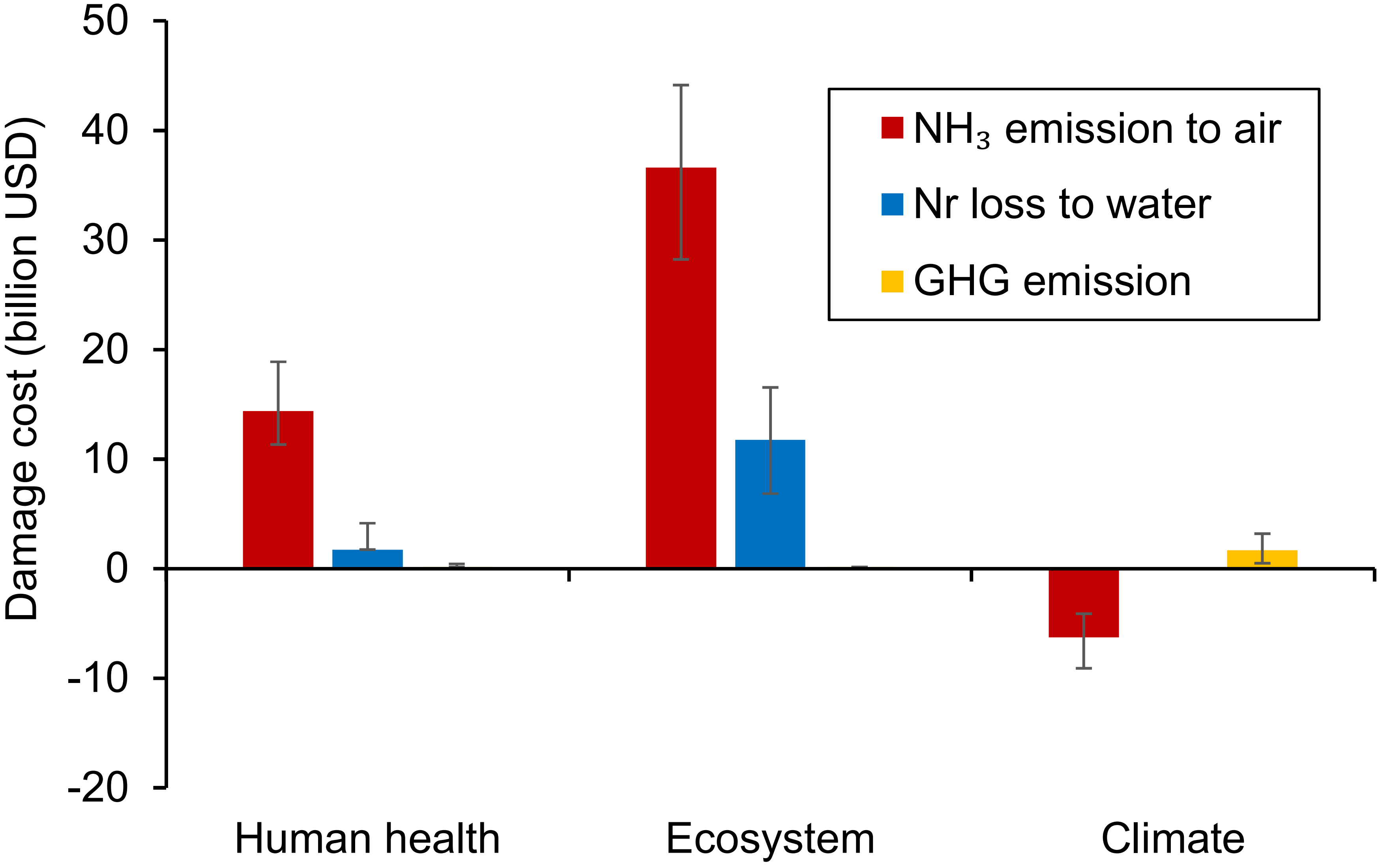

Ruminant
Monogastric
All animal
2007
2017
Pig unit
0
50
100
200
500
1000

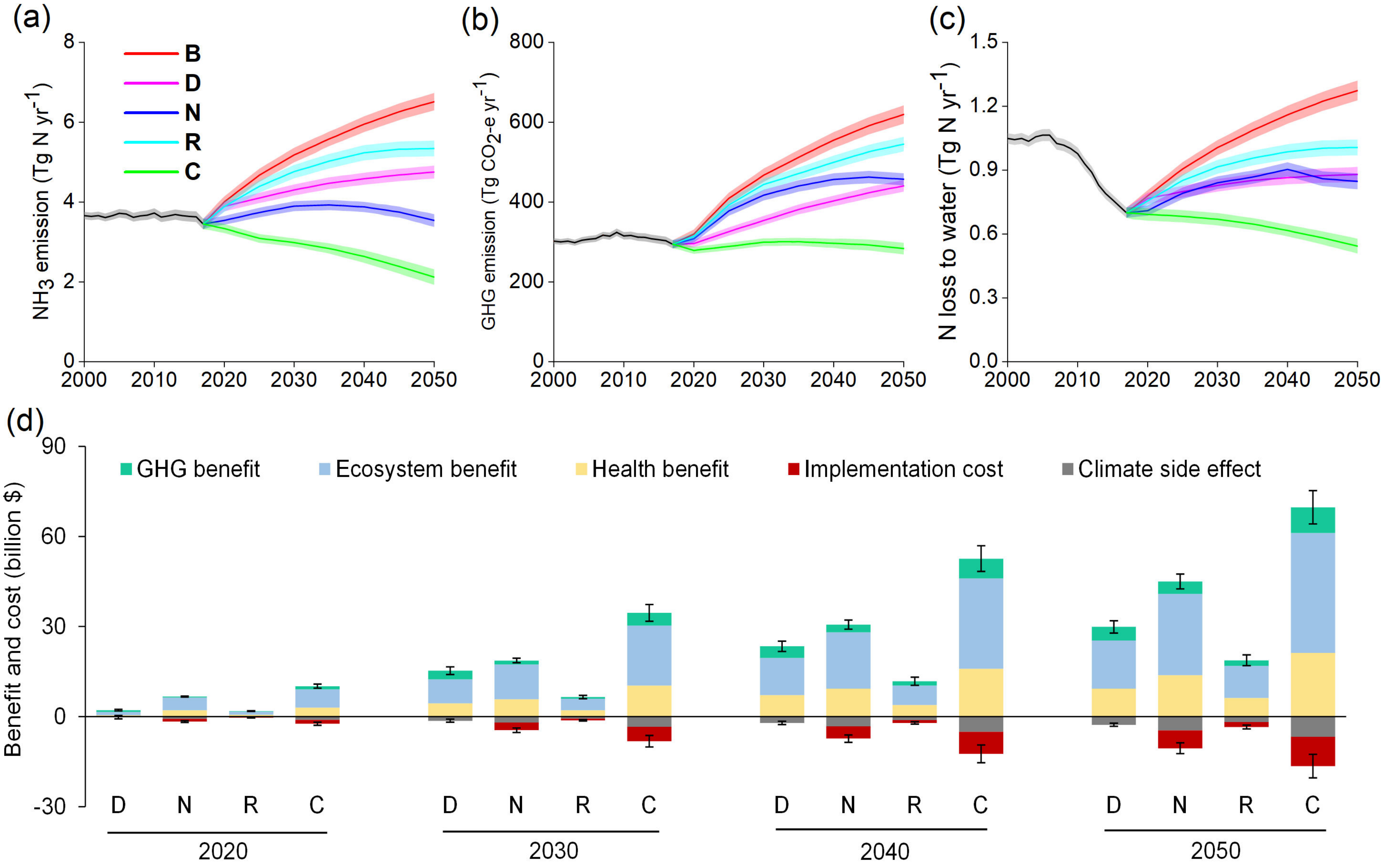

(a)
N_2O
NH_3
0.2
3.7
Intake
11.1
Livestock in 2007
Manure to cropland
1.8 *
Product & Retention
2.5
Runoff & Leaching
1.0 *
Other Use
2.0 *
(b)
N_2O
NH_3
0.1
3.4
Intake
10.0
Livestock in 2017
Manure to cropland
2.3 *
Product & Retention
2.5
Runoff & Leaching
0.7 *
Other Use
1.0 *



CH4 emisions
N runoff to water
N2O emissions
NH3 emissions
Total N losses
Manure recycling
N excretion
NUE
Production
Large-scale share
Livestock unit
-60
-40
-20
0
20
40
Changes (%)


(a)
Manure to cropland
(b)
N to cropland
Recycle ratio (%)
0
10
20
30
40
50
60
70
80
90
100
(c)
Manure recycling ratio (%)
0
10
20
30
40
50
60
70
80
2007
2017
All
Pig
Dairy cattle
Beef cattle
Chicken
Sheep
(d)
N loss ratio (%)
0
10
20
30
40
50
60
70
Lost to air
Lost to water
All
Pig
Dairy cattle
Beef cattle
Chicken
Sheep


Damage cost (billion USD)
50
40
30
20
10
0
-10
-20
NH3 emission to air
Nr loss to water
GHG emission
Human health
Ecosystem
Climate


(a)
(b)
(c)
(d)
B
D
N
R
C
NH3 emission (Tg N yr-1)
GHG emission (Tg CO2-e yr-1)
N loss to water (Tg N yr-1)
0
2
4
6
8
200
400
600
800
0.0
0.3
0.6
0.9
1.2
1.5
2000
2010
2020
2030
2040
2050
GHG benefit
Ecosystem benefit
Health benefit
Implementation cost
Climate side effect
Benefit and cost (billion $)
-30
0
30
60
90
D
N
R
C
2020
2030
2040
2050