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Article

Keywords: livestock manager, environment, ecosystem, climate, human health

Posted Date: June 2nd, 2021

DOI: https://doi.org/10.21203/rs.3.rs-523982/v1

Version of Record: A version of this preprint was published at Nature Food on February 24th, 2022. See the published version at https://doi.org/10.1038/s43016-022-00462-6.

High-resolution based sustainable livestock management in China

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China is the largest producer of livestock globally, and associated nutrient losses and greenhouse gases emission have substantially affected the environment and human health. Here we present a first detailed assessment of livestock systems using over 480,000 feedlot surveys over the period from 2007 to 2017 across China. Results indicate that China produced more livestock protein with fewer animal numbers and total pollution impacts through better breeding and feeding in large-scale feedlots over this period. Hotspots of production can be observed across the North China Plain, Northeastern China and Sichuan Basin. The Clean Water Act reduced manure nutrient losses to water by one third, while slightly increasing methane emissions over the study period. Integrated abatement measures could reduce more than half of livestock pollution in 2050, with benefits of 30 billion US dollar (USD) due to avoided human health, ecosystem and climate costs with only 6 billion USD abatement cost.

An increasing share of global grain production is used for livestock feed rather than directly serving as human food, with a share of up to 85% in developed countries, which can pose substantial pressure on food security [1]. Agricultural nitrogen (N) and phosphorus (P) emissions are considered a dominant source of air and water pollution in many global regions [2]. Nutrient losses to the environment can change ecosystem structure and threaten biodiversity [3, 4]. Agriculture is also an important source of greenhouse gas (GHG) emissions [5, 6]. Livestock production typically contributes to over half of all agricultural emissions, especially in developing countries [7, 8]. If taking the environmental impact of feed production into account, livestock production dominates agricultural pollution on both global and regional scales [1].

Despite the important role it plays for both food security and environmental protection, livestock production is generally not well understood due to a lack of detailed spatial distribution data [9]. In contrast to the spatial distribution of croplands that can be derived from remote sensing [10], the distribution of livestock production can only be robustly based on surveys of feedlots 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 the secondary formation of secondary inorganic aerosols, which contribute to fine particle matter ($PM_{2.5}$) pollution [11]. Previous studies mainly estimated the distribution of livestock production through proxy variables such as rural human population [12]. However, this is only viable when livestock production is dominated by backyard farming. With the increase of large-scale feedlots [13], it is essential to build accurate feedlot maps for the assessment of livestock systems.

China is the largest livestock producer globally [14]. Livestock production in China not only affects food security and environmental pollution within China, but also cascades impacts through international trade and global atmospheric circulation beyond China's land area [15, 16]. Although many previous studies have estimated the distribution of livestock production in China, the majority of them did not capture spatial patterns in sufficiently high spatial resolution [12, 17]. Without such a detailed map, the environmental impacts of livestock cannot be adequately investigated, e.g. through the simulation of air and water pollution, and results on a national scale will be subject to large uncertainties [18, 19]. Abatement measures and policies based on such results will not be spatially explicit enough to scientifically underpin targeted, local mitigation measures [9]. Fortunately, two agricultural pollution source censuses were conducted in 2007 and 2017 that covered all feedlots including on both smallholder and large-scale farms with precise locations (Extended Data Fig. 1). Based on these two censuses, we first (1) generate 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 and discussion

Distribution maps. The overall spatial patterns of livestock production (pig units) were similar in 2007 and 2017, with several hotspots observed across the North China Plain, 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 Monglia and Xinjiang. Beef cattle and sheep/goats are primarily observed in Shandong, Henna, Yunnan and Sichuan (Extended Data Fig. 3 & 4). Generally, more forage and straw supplies available in North China explain the preference for ruminant production there. Moreover, in addition to feedlots, ruminants are also grazed on grasslands in North China, e.g. Inner Mongolia, Xinjiang and Tibet.

Compared to ruminants, monogastric animals are found in both North and South China, and 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 s spatially associated with the distribution of croplands in China, especially for pig in 2017, due to grain feeds mainly being derived from crop production and the comparatively low transport costs due to proximity [20]. 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 [13].

These distribution maps are substantially different from previous global and China-specific studies [7, 12], 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 [21]. While a few scattered feedlots are present in Southeast China, our assessment did not find a continuous distribution of feedlots across the whole of South China present in previous studies [7]. Hilly and mountainous areas are commonly found in this region, which is not suitable for livestock production, and a lack of grain production from local crops would also result in prohibitively high feed transportation costs in these regions [20]. Spatial misrepresentation of livestock maps can add substantial uncertainties to policy decision-making as it may lead to inefficient targeting of livestock pollution control measures, while the Clean Air Act and Clean Water Act both identify livestock production as an important pollution source [22, 23].

From 2007 to 2017, a substantial decrease of the numbers of hotspots could be found, especially in South China (Fig. 1c and 1f). To control water pollution, a large number of pig and chicken farms in South China were closed and relocated to North China [22]. This reduced the overall spatial concentration of feedlots with a decrease in former hotspot regions and an increase in regions which previously did not have substantial livestock production activities. The resulting distribution of feedlots in addition better matches the locations of animals with croplands and thus may potentially contribute to the reduction of livestock pollution by minimizing transport emissions and having more land for manure spreading and reducing mineral fertilizer application [24]. Meanwhile, red meat and milk consumption are increasingly satisfied by import, which contributed to a reduction in domestic production [14]. Furthermore, although a general reduction of livestock numbers was observed, the relative production efficiency per animal increased, which compensated for the negative impact of animal number decline on total livestock production.

Better performance. Although livestock numbers decreased by 14% between 2007 an 2017, total livestock output production increased by 3% (Extended Data Fig. 6), suggesting that production per animal increased (Figs. 2 and 3). The proportion of large-scale farming increased from 31% to 45% between 2007 an 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 [24]. 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. 3 & 4). This led to reductions in both feed consumption and N excretion (Fig. S6-S9), 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 feedlots 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 [22]. To reduce water pollution, manure in feedlots was mainly subjected to dry cleaning with limited water use by 2017, and a requirement introduced for manure from large-scale feedlots to be treated before discharge (Extended Data Fig. 7). The way to store manure has also changed over the study period from air-dry on the ground to liquid form in open storage lagoons. These changes reduced pollutant discharge to water bodies by one-third (e.g., total N and P, Fig. S2-S3) as a consequence of the Clean Water Act entering into force in 2008 [13]. The national government invested over 5 billion Chinese Yuan to subsidize setting up over 5,000 large-scale feedlots with better facilities to clean manure from surfaces and storageein open lagoons, while the solid manure storage area was covered and thus protected from rain and leakage [25].

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 increased losses to air through NH_3 as well as generating additional methane (CH_4) emissions (Fig. S4-S5). Due to the increase of NUE, total manure N was reduced, however, which led to a 5% reduction of NH_3 emissions overall (Extended Data Fig. 9). However, management options aimed at controlling water pollution resulted in small changes to the the loss pathway of 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 liquid manure storage in open lagoons.

To increase the reuse of manure, the national government implemented policies to redistribute feedlots national, based on where sufficient cropland areas were available to use locally produced manure produced [25]. 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 [22, 24]. This change in spatial distribution improved the balance between cropland area and livestock density between regions, and as a result led to an increase in the recycling of manure to croplands at the local scale [20]. 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). This inconsistency can be mainly explained by N losses through gaseous NH_3 emissions (Fig. 2 and 3). Despite the solid part of manure being recycled, the open design of manure storage did not prevent nutrient lossess to air 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 to the atmosphere at every step of the N cascade [26].

Environmental and climate impacts. NH_3 and GHG emissions (including CH_4 and N_2O) and 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 all animals at county scale, with the exception of 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 major precursor of $PM_{2.5}$ pollution in China, especially in winter when NH_3 emissions from croplands are limited [11]. $PM_{2.5}$

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 a 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 formatin, as well as increasing carbon sequestration via nutrient N deposition, amounting to an estimated benefit of 6 billion USD overall (Fig. S1c).

GHG emissions including both N_2O and CH_4 can also damage human and ecosystem health indirectly and bring climate impact directly [27], 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 emission bring about 1.7 billion USD damages to climate, referring to ozone depletion and global warming.

Nitrate loss to water bodies from livestock production also damages human and ecosystem health [28]. Nitrate concentrations in drinking water is associated with cancer risks of the digestive system, and it is also contributing to eutrophication and harmful algae blooms in freshwater and coastal ecosystems [29]. Overall damage costs realted to water pollution were estimated at 14 billion USD damages, with ecosystem damages contributing over 85% of this value. Nitrate pollution does not contribute directly to damage costs related to climate change, and hence is not estimated in this study.

On a spatial scale, human and ecosystem health damages are generally correlated with livestock production, and hotspots can be found in the middle of Northeastern China, the North China Plain, the Middle and Lower Yangtze River Basin, and the Pearl River Delta (Fig. 5). 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, chicken, 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. These impacts on the environment and climate lead to economic costs to the whole society. To mitigate these impacts, measures based on currently available technology can reduce about 50% of total N losses through increasing NUE and recycling ratio, and optimizing human and livestock diets (Fig. 6). 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 Combined (C) that integrated these three scenarios. Detailed information of these abatement scenarios could be found in Methd section (Table S3). The Combined scenario can achieve about 30 billion USD benefit per year in 2030, which would double by 2050, while 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 socioeconomic 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 [30]. It suggests that incentive to farmers is crucial for the implementation of pollutiaon control measures since the benefits are for the whole society.

However, with a project 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 is 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 [31]. The situation for N runoff reduction is similar. The Clean Water Act contributed to the reduction of N losses to water bodies from livestock feedlots and was influenced by the Tai Lake algal bloom event in 2007 [23]. Further reduction of N losses to water bodies beyond what has already been achived 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 to reducing livestock pollution in over 600 counties in China [25]. 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 [30].

Policy implications. Livestock pollution control is crucial for water and air quality, and it also contributes to achieving carbon neutrality by 2060 in China. Despite central and local governments having implemented multiple measures to reduce livestock pollution, future actions are still required given the continued increase of livestock production in China [25]. Given the substantial reduction of N losses to water bodies achieved in the period from 2007 to 2017, more attention should be paid to the reduction of NH_3 and GHG emissions [23]. NH_3 emissions from livestock production in winter is known to be a major precursor to $PM_{2.5}$ pollution. In the context of substantial reductions of sulfur dioxides (SO_2) and N oxide (NO_x) emissions already achieved from industrial and transport sources, a reduction of NH_3 emissions from agricultural sources will be more cost-effective than additional reduction of SO_2/NO_x from other sources [11].

Meanwhile, to meet the target of the Clean Air Act (2nd stage), more sophisticated measures have been developed and demonstrated in some typical regions where both intensive livestock production and substantial air pollution challenges occur, such as closed systems for manure storage and liquid manue injection [31]. However, these NH_3 mitigation measures generally do not typically consider synergies and co-benefits of GHG emissions reductions. In contrast, some measures reducing NH_3 emissions can even increase GHG emissions [23]. Therefore, more research on 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.

A large part of the reduction of N losses to water bodies has been achieved through delimitation of restricted areas for livestock production and moving pigs from South China to North China [22, 25]. Manure recycling is considered the most efficient way to reduce livestock pollution, since croplands are the best destination for manure [20]. This paper provides, for the first time, high-resolution maps of livestock feedlots that can be used for the analysis of recoupling of livestock and croplands on an unprecedented scale to reduce storage and transport cost of manure.

Methods

Data sources

Data on livestock numbers in both large-scale and small-scale feedlots 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 feedlots are surveyed in 2007 and 2017, respectively (Extended Data Fig. 1). For small-scale feedlots, statistical surveys are conducted at a county scale. Their spatial distribution is highly correlated with that of the rural population density distribution in each county. The threshold numbers defining the category of “large-scale” feedlots 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).

Emission calculation

To determine excretion generated per animal, approximately 200 feedlots were selected for monitoring across China based on the distribution of feedlots of different livestock species including pig, layer, broiler, beef and dairy cattle in both census years (Extended Data Fig. 2). 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.

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 (Fig. S8):

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

QF (kg/head) is the total amount of feces produced in the *i* feeding stage of a certain animal; QF_i (kg/head/day) is the number of feces produced per day in the *i* 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 (Fig. S9):

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

QU (L/head) is the total amount of urine produced in the *i* feeding stage of a certain animal; QU_i (L/head/day) is the amount of urine produced per day in the *i* 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* 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* 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/goad 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 is 365 weighted based on age, farm calf: young cattle: lactating cow = 15:30:55. Feeding days of beef cattle is 365 weighted based on age 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:

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

$FD_{i,j}$ (g/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 feeding stage of this animal; $CF_{i,j}$ (g/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 feeding stage of this animal; $CU_{i,j}$ (g/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:

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

QFD_j (kg/head) is the total emission of the j^{th} pollutant of a certain animal; $FD_{i,j}$ (g/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 is applied to calculate the system N balance.

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

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

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

N_{input} is the total N input to the livestock system, including N fertilizer (N_{fer}), 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.

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)

$$\Delta E_{i,m} = \sum_j A_{i,j} * \left[EF_{i,j,m} \times \eta_{i,j,k} \times X_{i,j,k}\right] \qquad (10)$$

Where i represents the region; j represents the animal type; m represents the form of N losses (NH_3, NO_x, N_2O, N leaching and runoff) from livestock production; k represents the specific mitigation options; $A_{i,j}$ is the livestock population; $EF_{i,j,m}$ is the corresponding uncontrolled emission factor; $\eta_{i,j,k}$is the specific abatement efficacy; $X_{i,j,k}$ 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 [32, 33]. The annual implementation cost ($IC_{i,m}$) in China is calculated as:

$$IC_{i,m} = \Delta E_{i,m} * UC_{i,m} \qquad (11)$$

where $UC_{i,m}$ 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 ($SOC_{i,j}$) of mitigating N pollution from livestock production (Table S2) is defined as the sum of avoided damage cost for human health ($HH_{i,j}$), ecosystem health ($EH_{i,j}$), GHG reduction ($GHG_{i,j}$, e.g., CH_4 reduction) and climate effect ($Climate_{i,j}$, e.g., climate warming due to reduction of aerosol) as shown in Eq. (6):

$$SOC_i = HH_i + EH_i + GHG_i - Climate_i \tag{12}$$

Ecosystem benefits. A number of US and EU studies have examined the damage cost of N_r effects on the ecosystems [28, 34-38], 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. (7)

$$EH_i = \sum_m \Delta E_{i,m} * \partial_{EU,m} * \frac{WTP_i}{WTP_{EU}} * \frac{PPP_{China}}{PPP_{EU}} \tag{13}$$

where ∂_{EU} is the estimated unit ecosystem damage cost of Nr emission in Europe based on the European N Assessment (Sutton et al., 2011; Jones et al., 2014); WTP_i and WTP_{EU} are the values of the willingness to pay (WTP) for ecosystem service in region *i* 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 [23, 39]. 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 [40]. A detailed description of the health damages attributed to air pollution ($PM_{2.5}$) and water pollution due to N_r emissiona 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):

$$HH_i = \sum_m \Delta E_{i,m} * HCost_{i,m} \tag{14}$$

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

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.

$$GHG_i = \Delta E_{GHG,i} * GCost_i \tag{15}$$

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

Climate impacts. NH_3 emission is reported to have a cooling effect on the climate [41].The climate impact of improved N management is assessed as showed in Eq. (10):

$$Climate_i = \sum_m \Delta E_{i,m} * CCost_{i,m} \qquad (16)$$

Where $\Delta E_{i,j,m}$ represents the reduction of Nr loss. $CCost_{i,j,m}$ 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 [42]. Based on current policy, action and programs for livestock production and future social-economic development prediction, this study first 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 [23]. 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.

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Acknowledgments

This study was supported by the National Natural Science Foundation of China

(41822701, 42061124001, 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 and all authors contributed to the discussion and revision of the paper.

Data availability

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

Competing interests

The authors declare no competing interests.

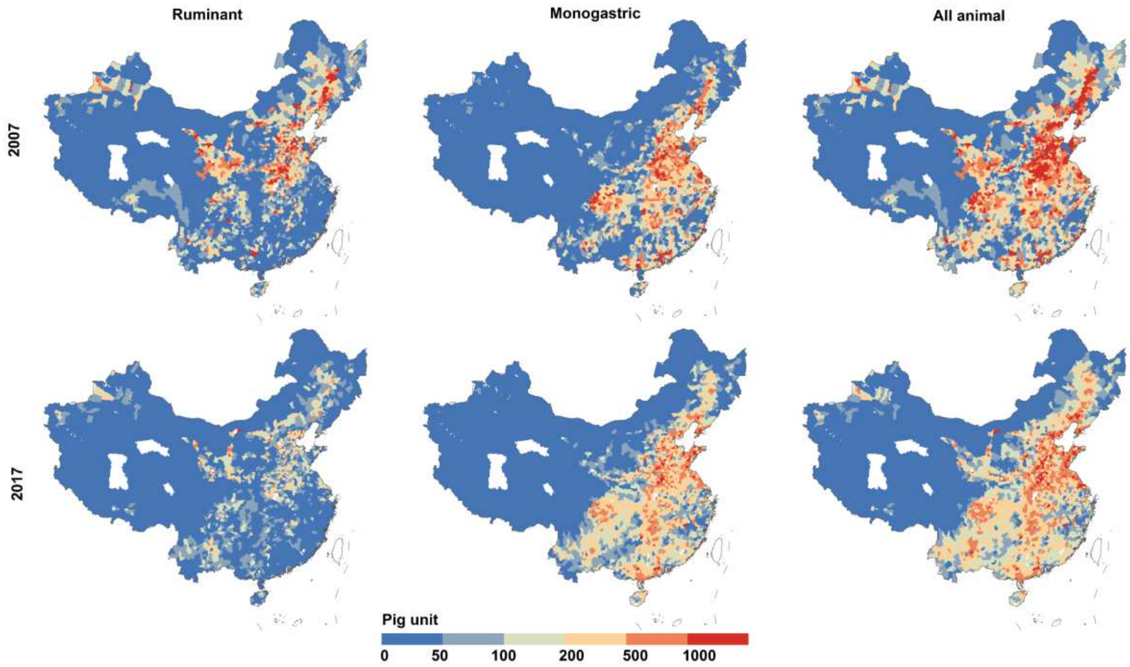


Fig. 1 | Distribution of livestock production in China on county scale. All numbers are converted to pig units. 1 dairy cattle = 10 pigs; 1 beef cattle = 5 pigs; 3 sheep/goat = 1 pig; 15 layer chickens = 1 pig; 60 broiler chickens = 1 pig. Ruminant includes cattle and sheep/goats, and monogastric animals 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 feedlots (Extended Data Fig. 1). Base map is applied without endorsement from GADM data (https://gadm.org/).

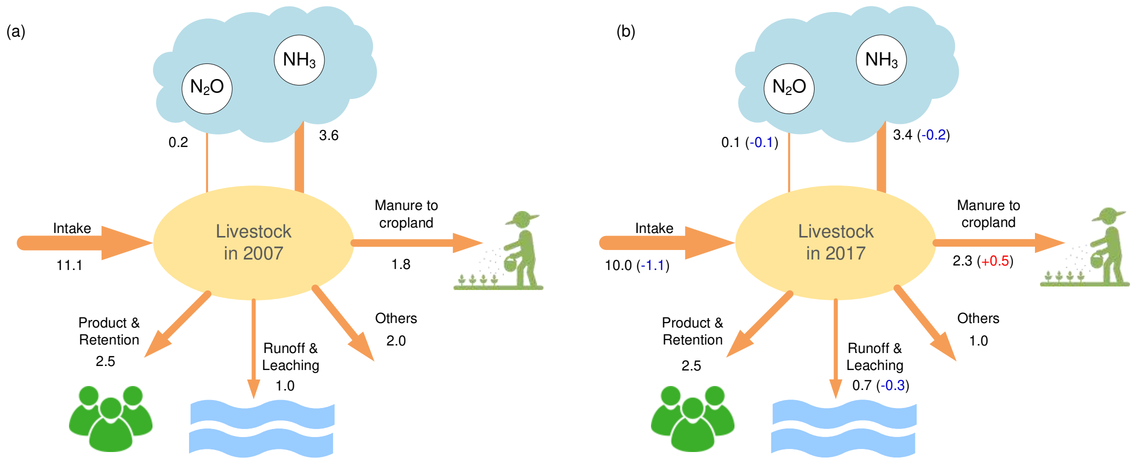


Fig. 2 | Changes of N balance of livestock system from 2017 to 2017 in China. Due to data limitation, livestock species only includes cattle, sheep/goat, pig and chickens, which account for about 90% of total livestock protein produced. Changes (+/- or red for negative, blue for positive) in figure (b) show what has changed from 2007 to 2017. Others refer to unknows N losses such as N_2 emission through denitrification.

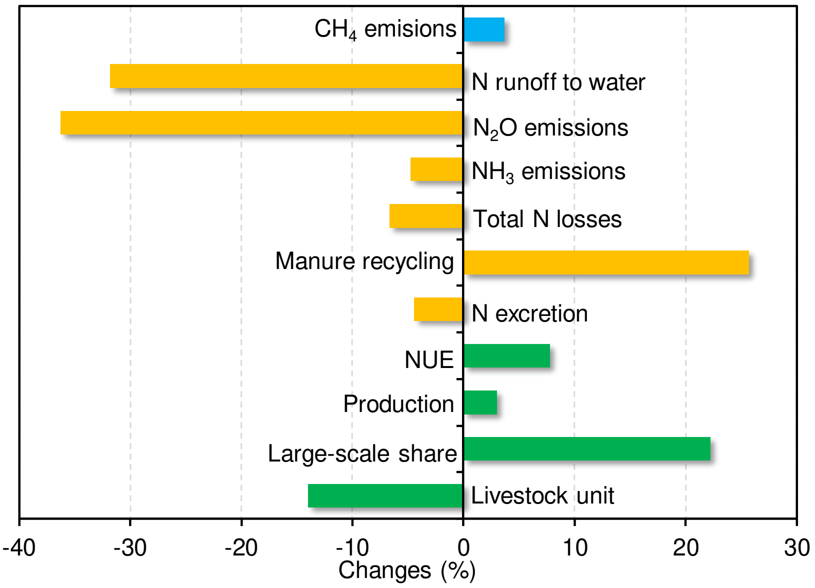


Fig. 3 | Changes of livestock system performance from 2007 to 2017. Production refer to livestock products such as meat and milk. Large-scale share refers to the ratio of animals raised in large-scale feedlots. Livestock unit refers to total animal numbers counted in pig units.

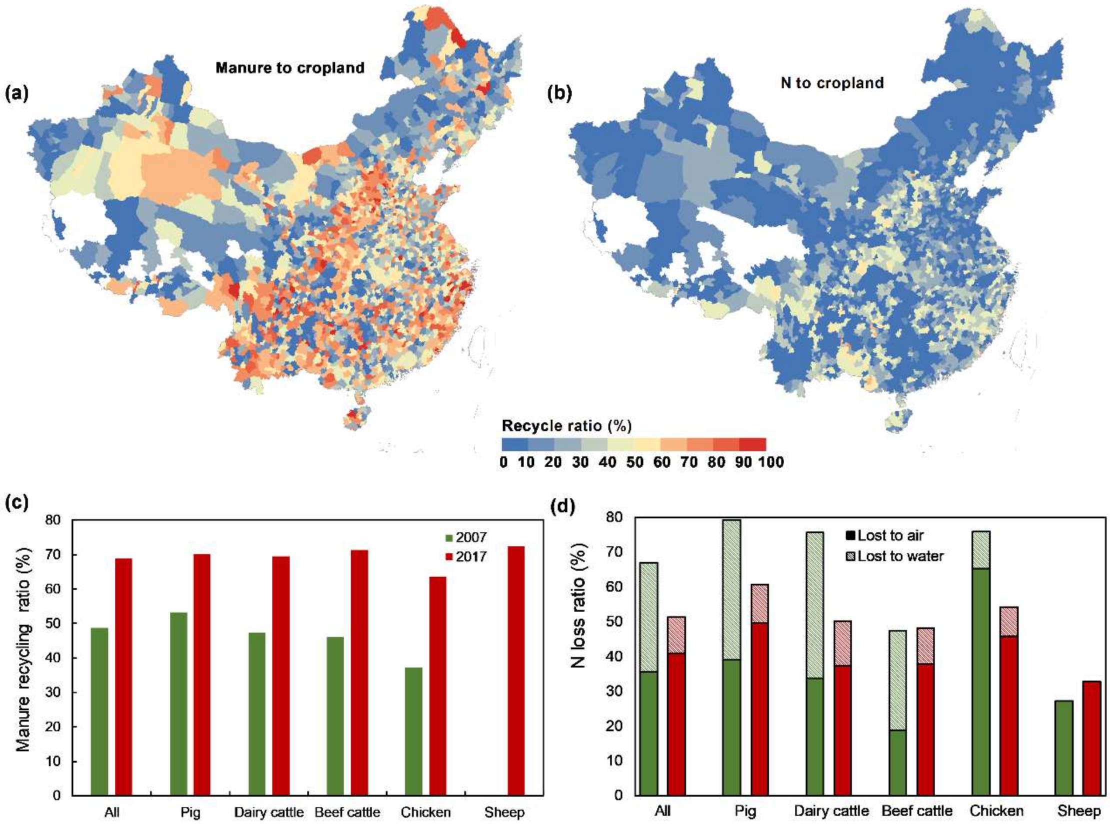


Fig. 4 | Manure recycling to croplands. (a) The ratio of manure recycling to cropland in 2017; (b) Ratio of total N derived from excretion recycling to cropland in 2017; (c) Comparison of manure recycle ratio in 2007 and 2017; (d) Comparison of N loss to air and water. Shaded vs. full colour area in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

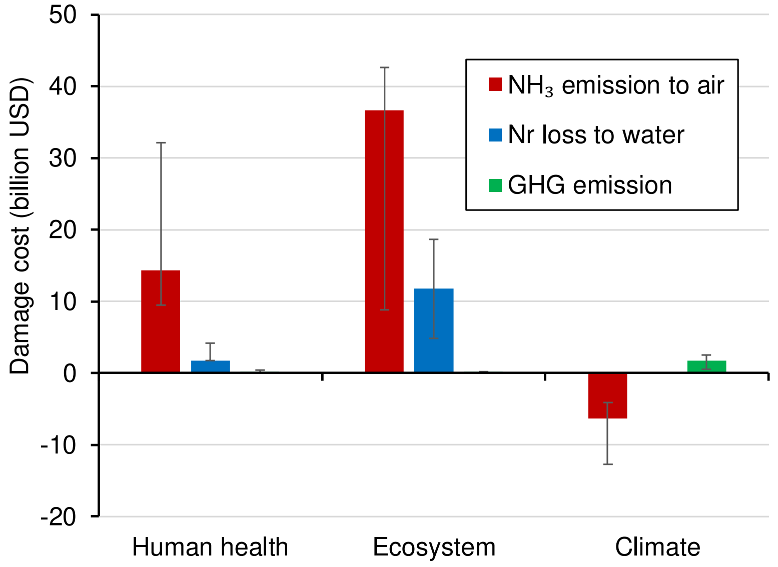


Fig. 5 | Health, ecosystem and climate effects of livestock pollution in 2017 in China. The error bars represent the lower and higher limits of the damage costs due to livestock pollution. Negative bumbers represent benefits of NH_3 emission. Detailed distribution of these costs could be found in Fig. S1.

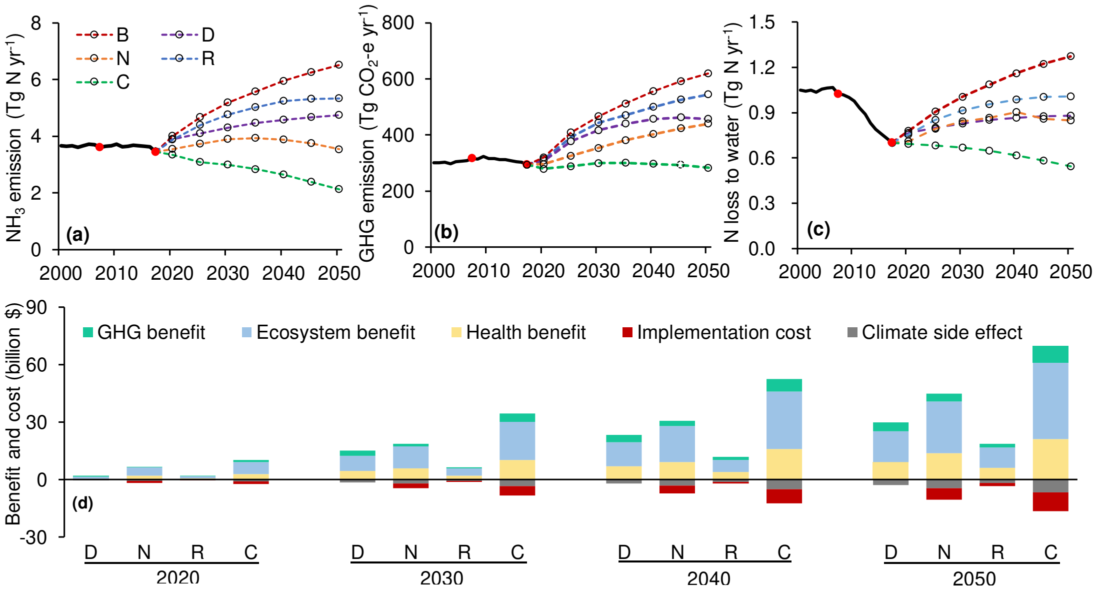
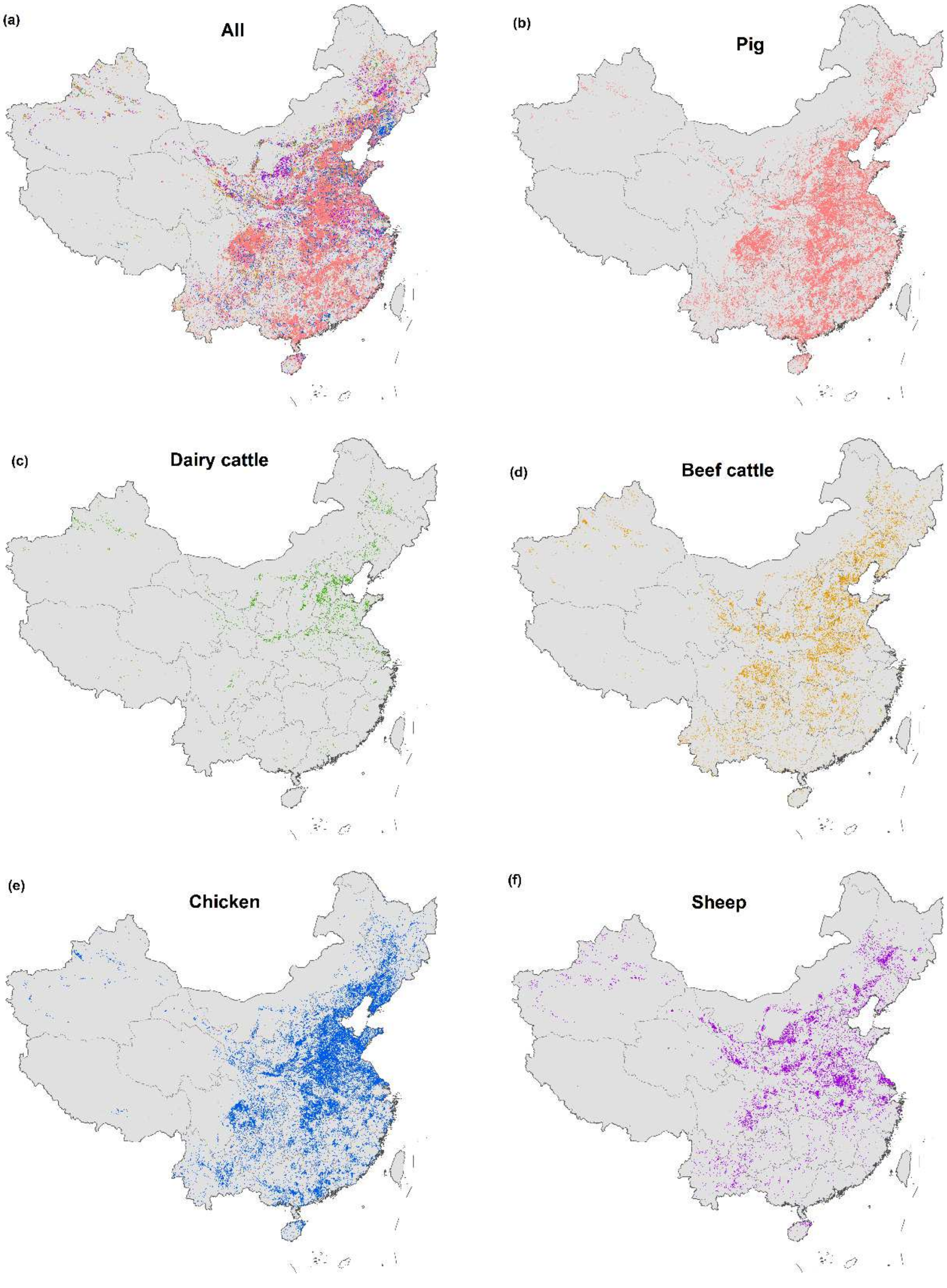
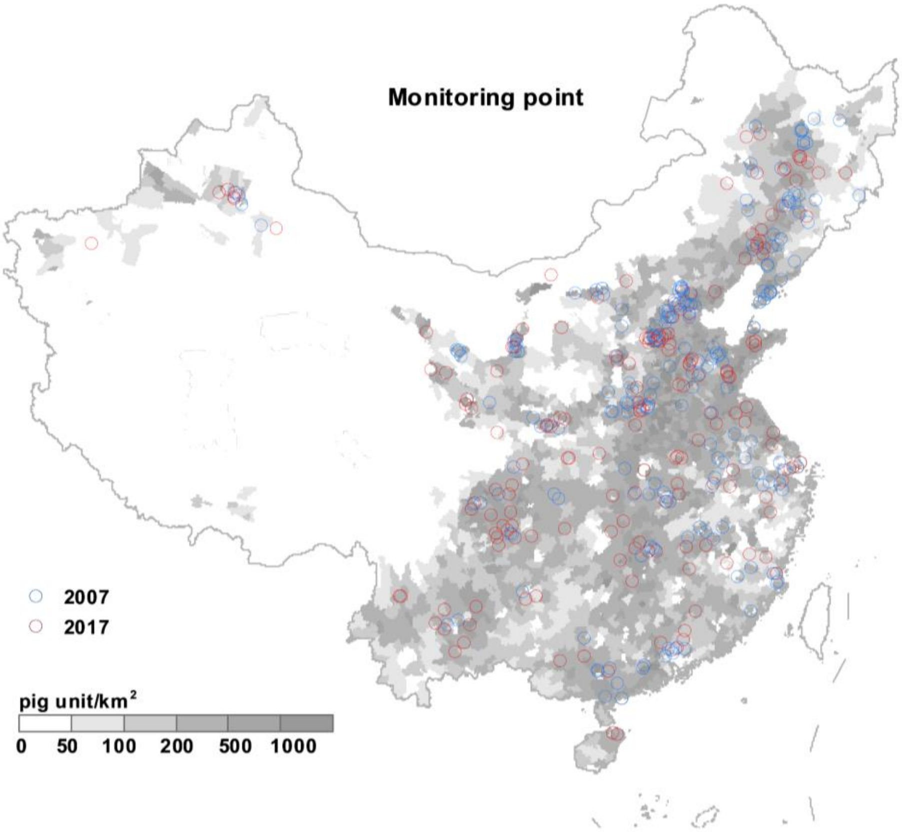


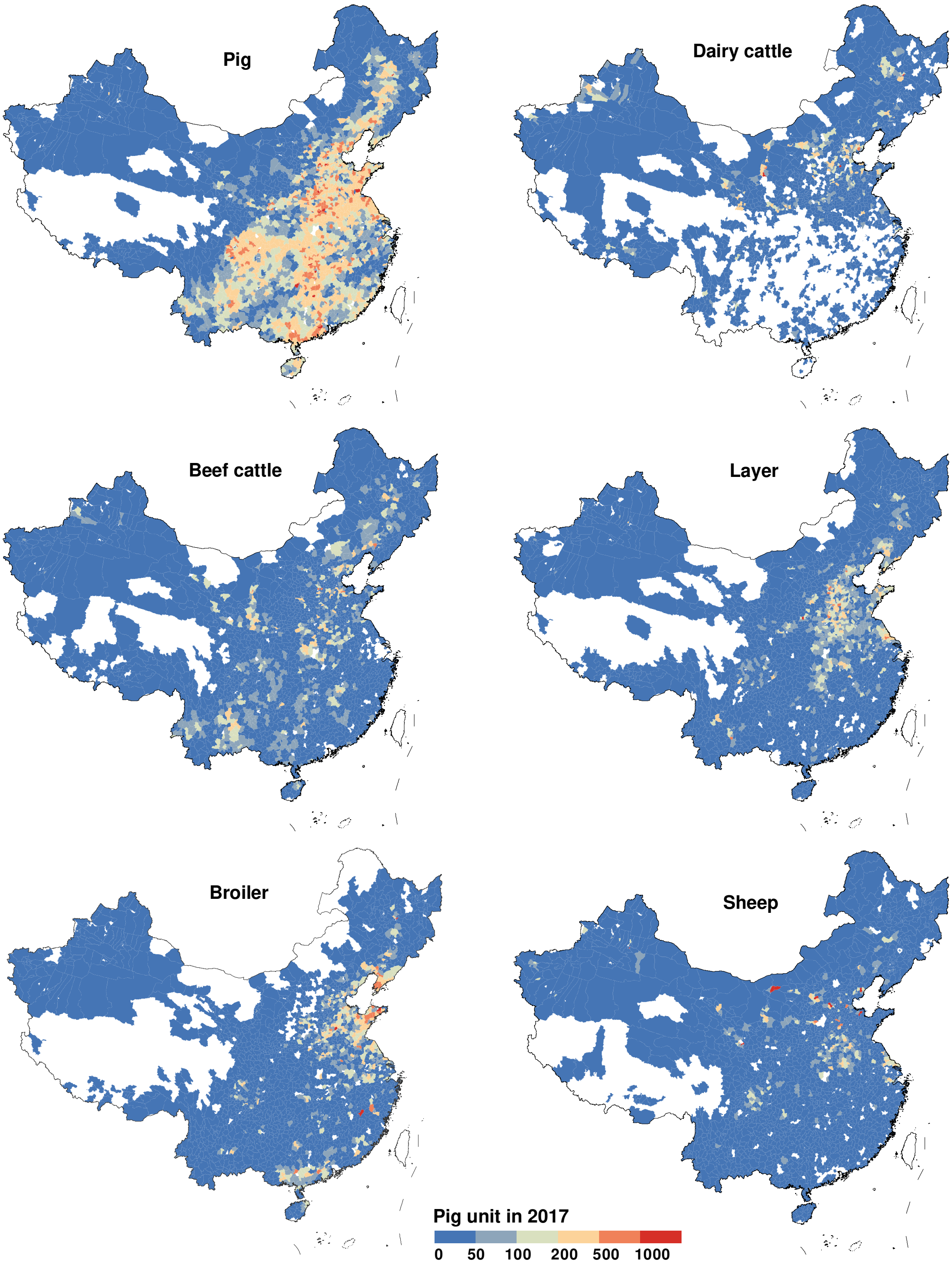
Fig. 6 | Future scenario of livestock pollution in China. (a) NH_3 emission; (b) GHG emission; (c) N loss to water; (d) cost and benefit to abate livestock pollution. B, Business as usual; D, diet; N, N use efficiency; R, Manure recycling; C, D+N+R.



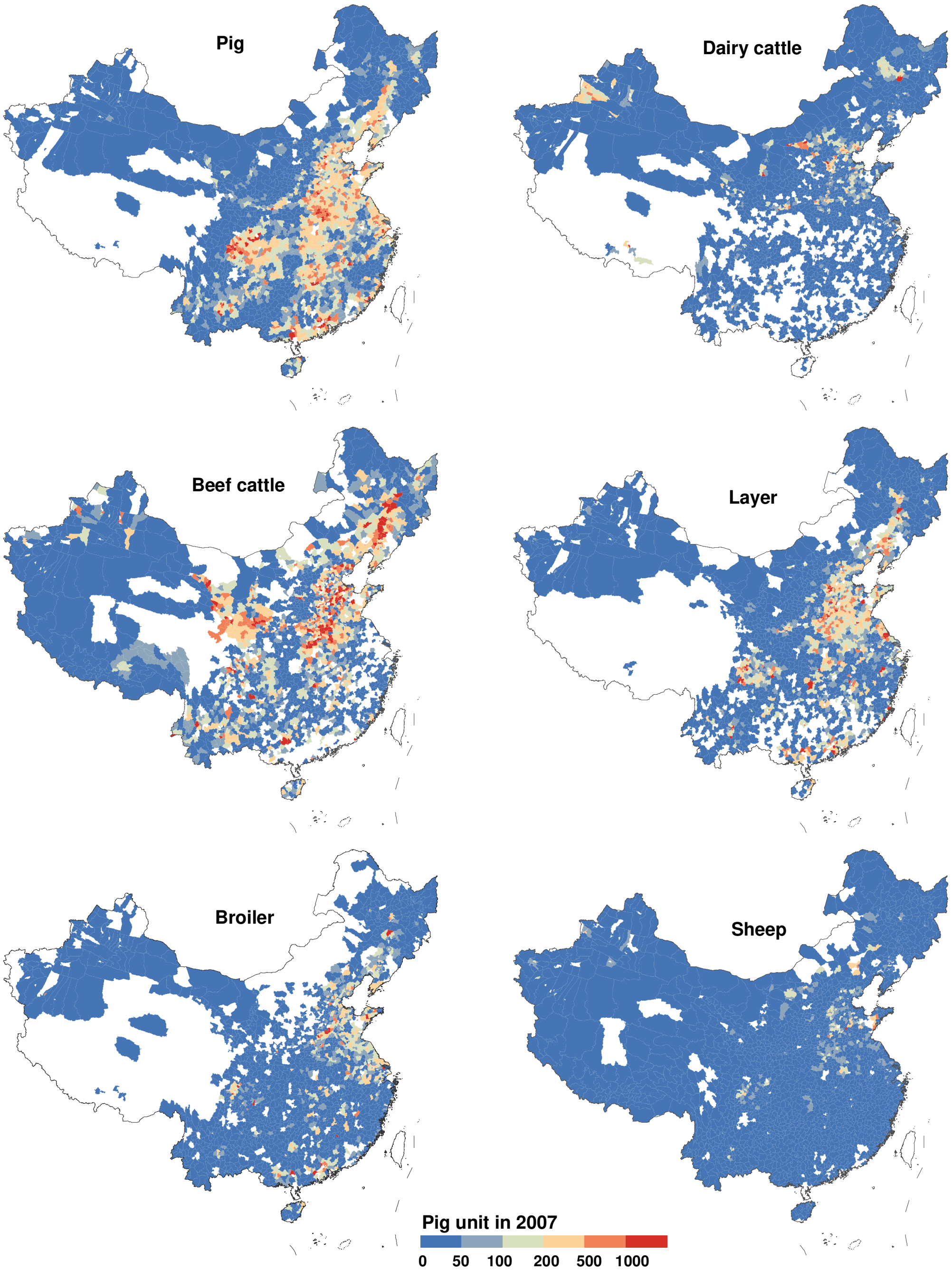
Extended Data Fig. 1 | Locations of the feedlots in agricultural pollution source censuses in 2017. (a) All animals; (b) Pig; (c) Dairy cattle; (d) Beef cattle; (e) Chicken; (f) Sheep. Base map is applied without endorsement from GADM data (https://gadm.org/).



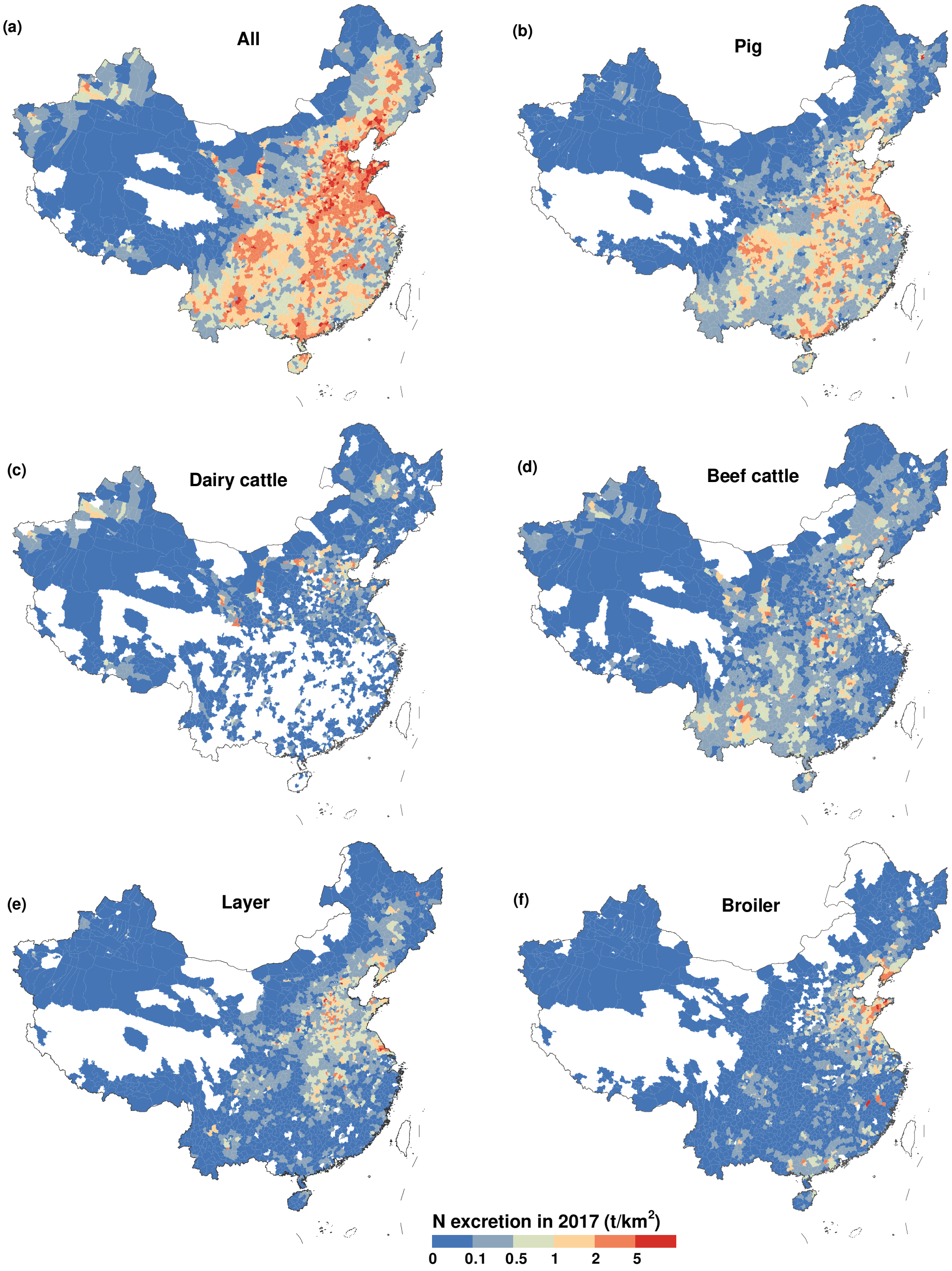
Extended Data Fig. 2 | Locations of the monitoring points for per animal feed use, excretion generated, pollutant emission and manure treatment in 2007 and 2017. Base map is applied without endorsement from GADM data (https://gadm.org/).



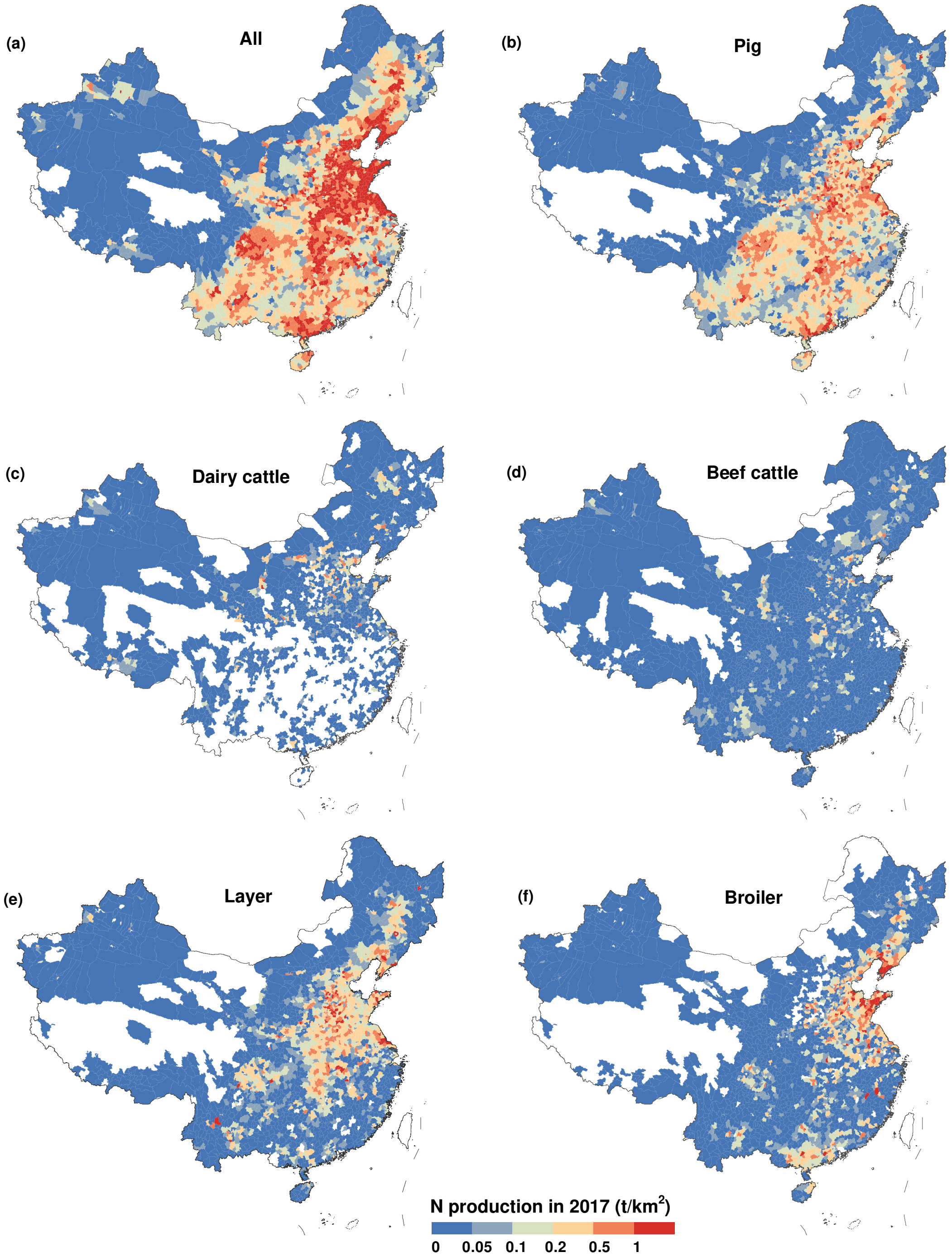
Extended Data Fig. 3 | Distribution of livestock numbers in China on county scale in 2017. All numbers are converted to pig units. 1 dairy cattle = 10 pigs; 1 beef cattle = 5 pigs; 3 sheep/goat = 1 pig; 15 layer chickens = 1 pig; 60 broiler chickens = 1 pig. Base map is applied without endorsement from GADM data (https://gadm.org/).



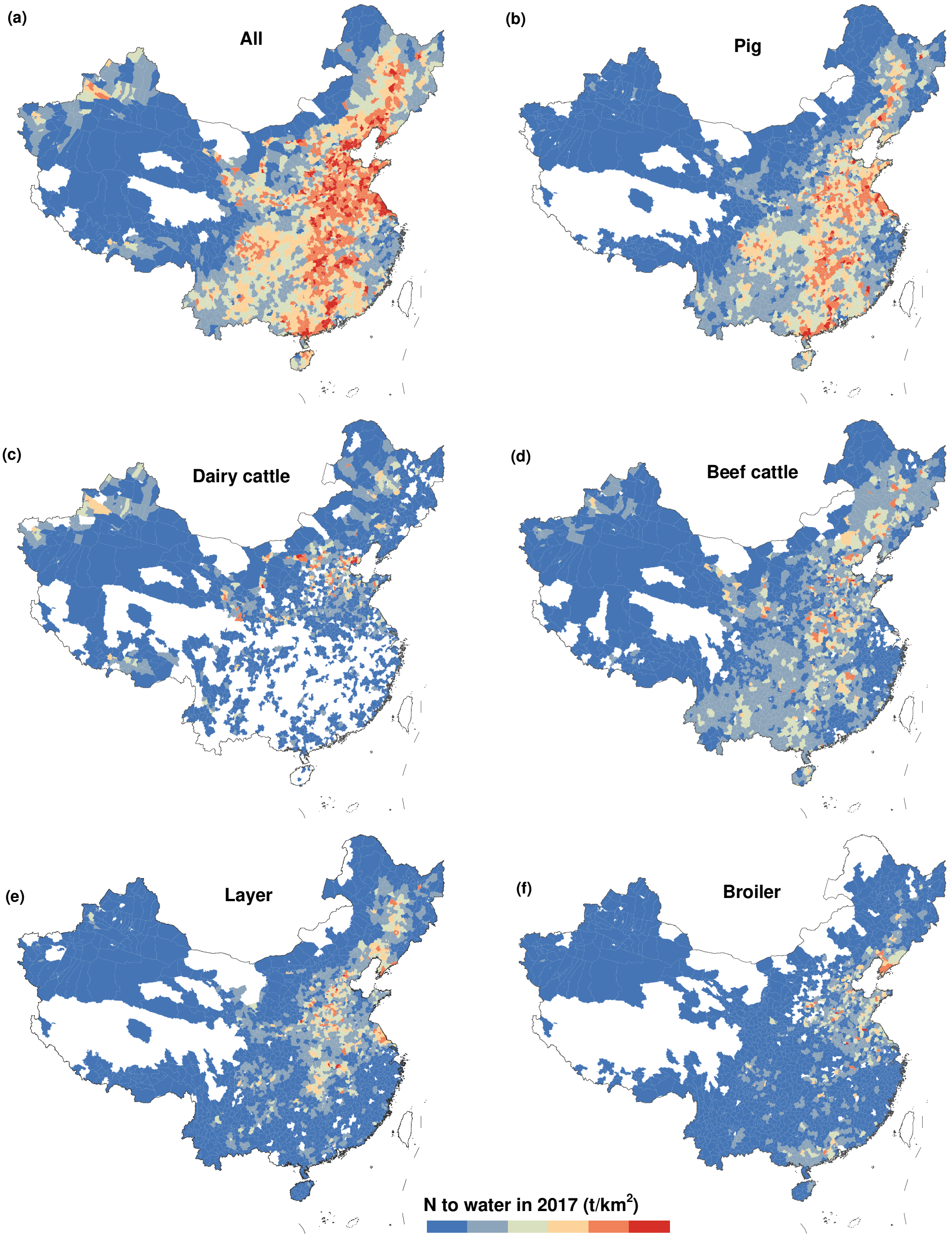
Extended Data Fig. 4 | Distribution of livestock numbers in China on county scale in 2007. All numbers are converted to pig units. 1 dairy cattle = 10 pigs; 1 beef cattle = 5 pigs; 3 sheep/goat = 1 pig; 15 layer chickens = 1 pig; 60 broiler chickens = 1 pig. Base map is applied without endorsement from GADM data (https://gadm.org/).



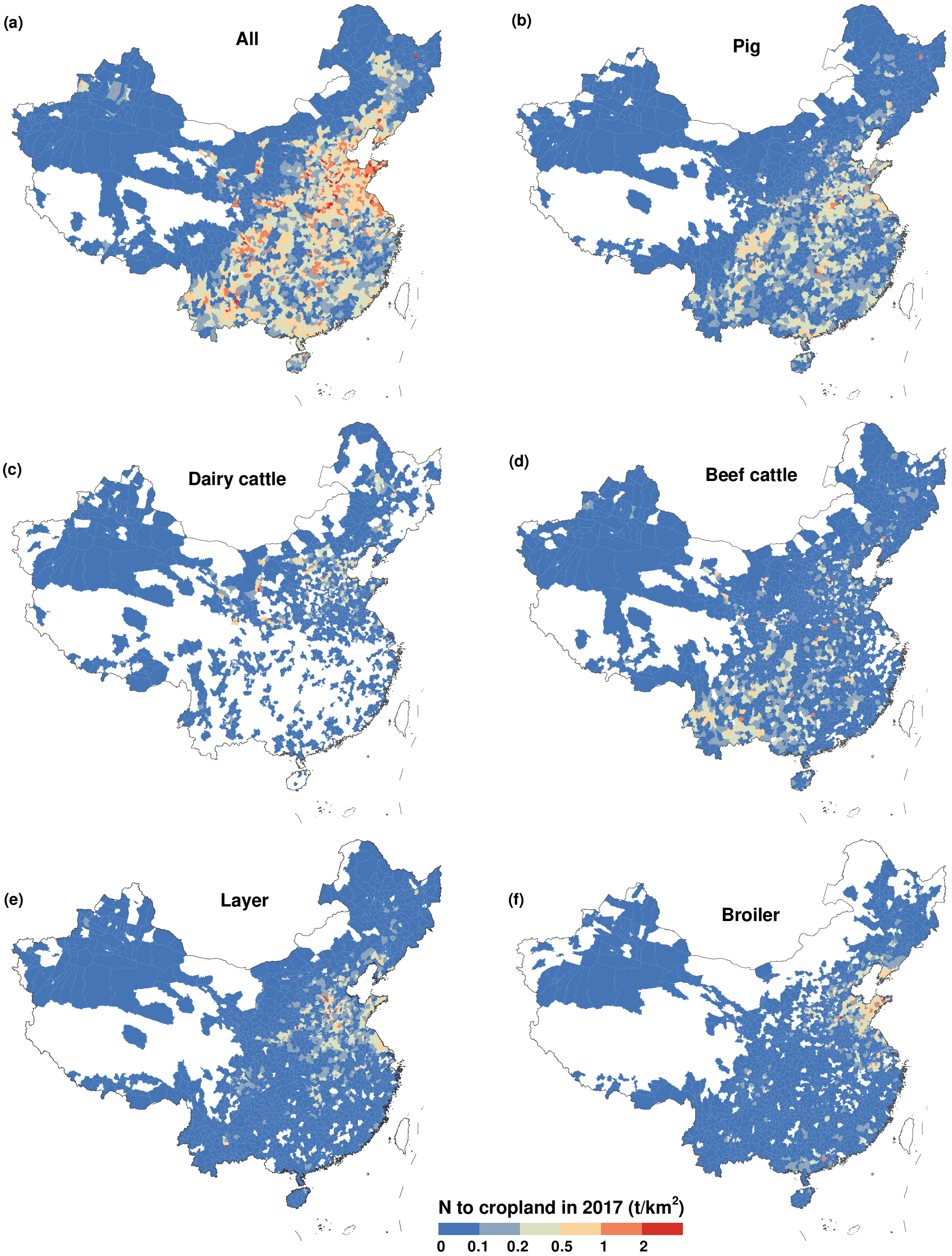
Extended Data Fig. 5 | Distribution of livestock N excretion in China on county scale in 2017. Base map is applied without endorsement from GADM data (https://gadm.org/).



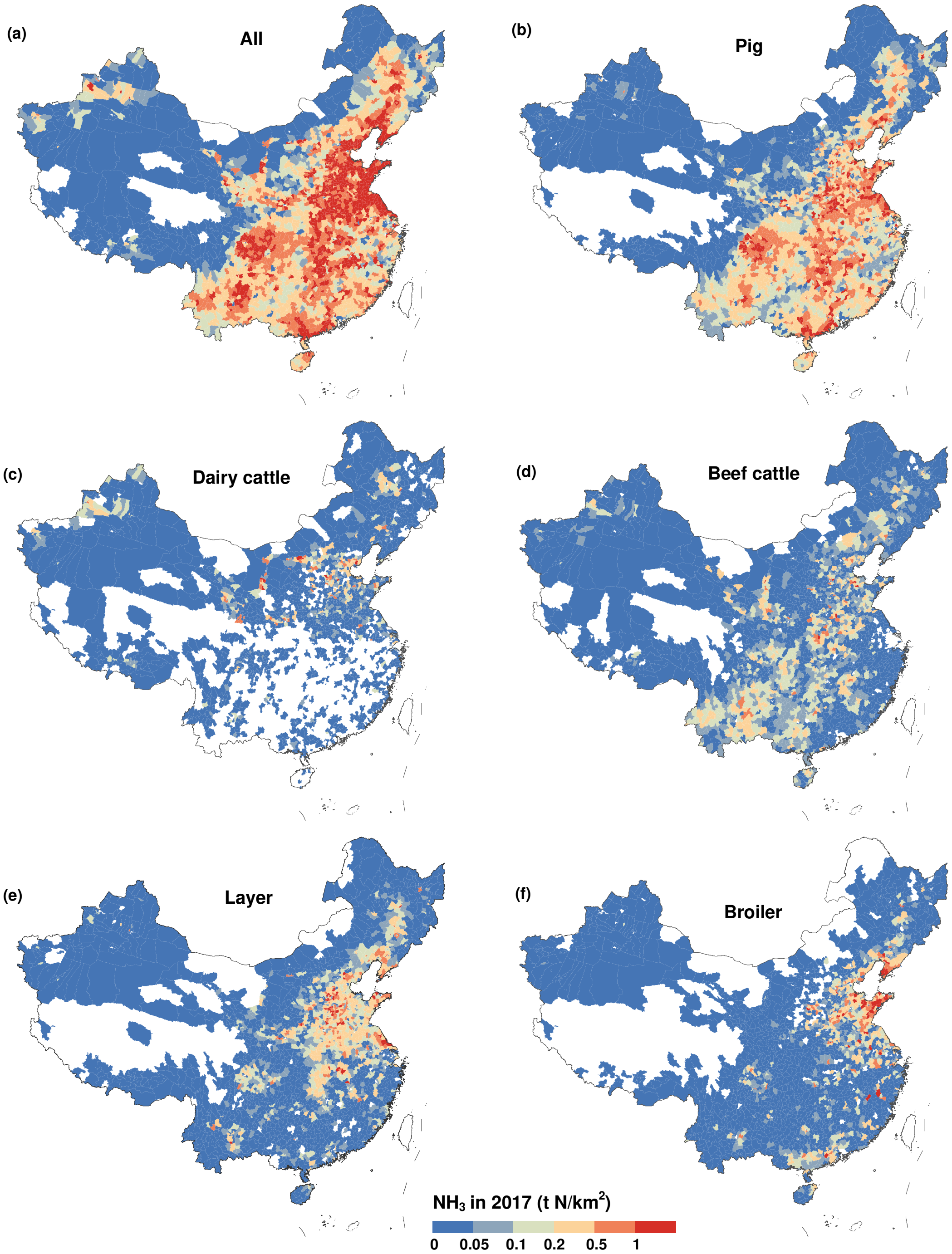
Extended Data Fig. 6 | Distribution of livestock N production in China on county scale in 2017. Base map is applied without endorsement from GADM data (https://gadm.org/).



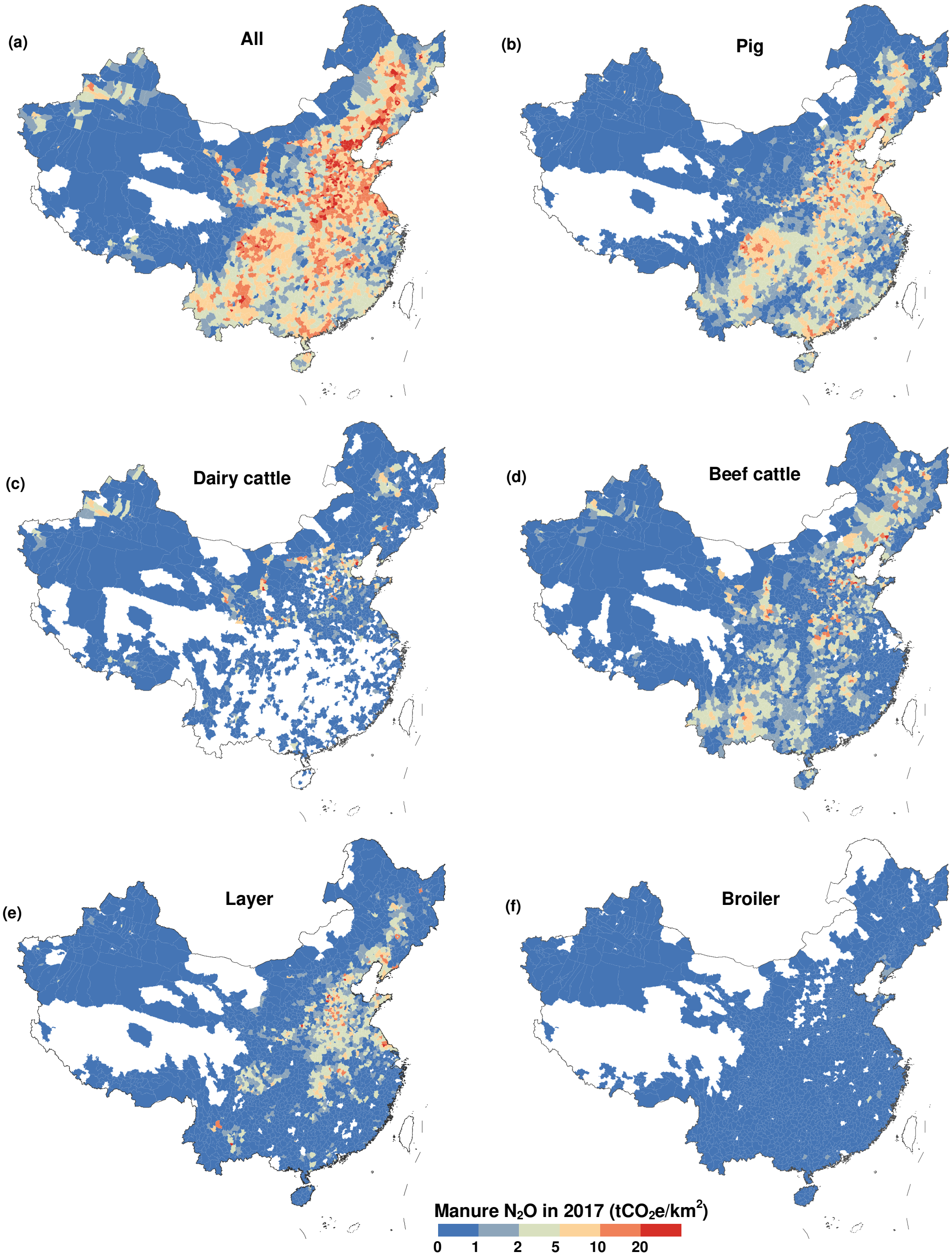
Extended Data Fig. 7 | Distribution of livestock N loss to water in China on county scale in 2017. Base map is applied without endorsement from GADM data (https://gadm.org/).



Extended Data Fig. 8 | Distribution of manure recycle to croplands in China on county scale in 2017. Base map is applied without endorsement from GADM data (https://gadm.org/).



Extended Data Fig. 9 | Distribution of NH_3 emission from livestock production in China on county scale in 2017. Base map is applied without endorsement from GADM data (https://gadm.org/).



Extended Data Fig. 10 | Distribution of manure N_2O emission in China on county scale in 2017. Base map is applied without endorsement from GADM data (https://gadm.org/).

Supplementary Files

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