nature food

ARTICLES

https://doi.org/10.1038/s43016-021-00430-6

Dietary shifts can reduce premature deaths related to particulate matter pollution in China

Xueying Liu[1,9], Amos P. K. Tai[1,2,3] ✉, Youfan Chen[4], Lin Zhang[4], Gavin Shaddick[5,6], Xiaoyu Yan[7] and Hon-Ming Lam[2,3,8]

Shifting towards more meat-intensive diets may have indirect health consequences through environmental degradation. Here we examine how trends in dietary patterns in China over 1980–2010 have worsened fine particulate matter ($PM_{2.5}$) pollution, thereby inducing indirect health impacts. We show that changes in dietary composition alone, mainly by driving the rising demands for meat and animal feed, have enhanced ammonia (NH_3) emissions from Chinese agriculture by 63% and increased annual $PM_{2.5}$ by up to ~10 μg m^{-3} (~20% of total $PM_{2.5}$ increase) over the period. Such effects are more than double that driven by increased food production solely due to population growth. Shifting the current diet towards a less meat-intensive recommended diet can decrease NH_3 emission by ~17% and $PM_{2.5}$ by 2–6 μg m^{-3}, and avoid ~75,000 Chinese annual premature deaths related to $PM_{2.5}$.

Food production to feed an ever-growing population and worldwide changes in dietary patterns have had substantial impacts on human and planetary health. A more nutritionally enriched diet nurtures human health and increases life expectancy, but the health benefits of higher nutrition in many countries are now offset by the shift towards poor dietary habits in the form of meat-intensive diets and overnutrition. In China, per capita meat consumption has been increasing rapidly since the 1980s, rendering China the number-one meat consumer in the world now, while per capita refined cereal consumption has been declining over 1997–2011[1]. The rapidly increasing demand for animal feeds poses another challenge to China's food supplies[2]. Much in synchrony with the worldwide transition towards diets high in calories and rich in heavily processed and animal-derived foods, the dietary changes in China and the resulting rise in body mass index have imposed increasing risks of cardiovascular diseases, cancers and type 2 diabetes to its population over recent decades[3,4]. Indeed, dietary risks are the number-one leading cause of mortality in China, estimated to contribute to 3.1 million premature deaths nationwide in 2017[5].

Meanwhile, environmental degradation incurred by increasing and changing patterns of food production may add further burdens to human health. Globally, food production is a major contributor to greenhouse gas emissions[6–9], land degradation[6–9], freshwater depletion[1,7,9] and biodiversity loss[10]; it also contributes to ~60% (30–36 TgN yr^{-1}) and ~8% (4 TgN yr^{-1}) of the global emission of ammonia (NH_3)[11,12] and nitrogen oxides (NO_x)[11], respectively, which play a major role in the formation of fine particulate matter 2.5 μm or less in diameter ($PM_{2.5}$)[13,14]. Agricultural intensification together with rapid industrialization has rendered China among the most polluted countries by the early 2010s, when the annual mean population-weighted exposure to $PM_{2.5}$ is around 55–60 μg m^{-3} for China as a whole and above 100 μg m^{-3} in some regions of eastern China, that is, two to seven times the exposure levels observed in developed countries[15–17]. As a public health threat, ambient $PM_{2.5}$ is responsible for 1.1–2.0 million deaths in China in 2017, ranked as the number-four leading cause of mortality[5,18].

Dietary changes, along with population growth, are key drivers of changes in food production, and thus pose a threat to public health not only directly via malnutrition but also indirectly via degrading air quality and respiratory health (Supplementary Fig. 1). Yet, such an 'indirect' health cost of dietary changes is previously unquantified but potentially important in major food-producing and food-consuming regions such as China. In this Article, we examine the impacts of rapid dietary changes in China on agriculturally derived $PM_{2.5}$ pollution and human health over the period of 1980–2010, controlling for the effects of population growth and diagnosing the contribution from different food types. We then compare the current diet with a less meat-intensive reference diet as recommended by the 2016 Chinese Dietary Guideline (CDG)[19] to explore the potential air quality and indirect health benefits if the Chinese population adopts a healthier diet.

Chinese food production and consumption patterns

We first analyse the changing patterns of food production and consumption in China using data from multiple sources, including the Food and Agriculture Organization (FAO)[20], the National Bureau of Statistics of China (NBSC)[21] and the China Health and Nutrient Survey (CHNS)[22]. The Chinese population is now consuming more food and thus more calories, protein and fat on a daily basis (Supplementary Fig. 2). The daily dietary energy supply per capita has increased from 2,100–2,400 kcal in the early 1980s to 3,000–3,100 kcal in the early 2010s, putting China at the top of developing countries approaching the levels of high-income countries. The proportion of nutrients derived from animal products has

[1]Earth System Science Programme and Graduate Division of Earth and Atmospheric Sciences, Faculty of Science, The Chinese University of Hong Kong, Hong Kong SAR, China. [2]Institute of Environment, Energy and Sustainability, The Chinese University of Hong Kong, Hong Kong SAR, China. [3]Centre for Soybean Research, State Key Laboratory of Agrobiotechnology, The Chinese University of Hong Kong, Hong Kong SAR, China. [4]Laboratory for Climate and Ocean–Atmosphere Studies, Department of Atmospheric and Oceanic Sciences, School of Physics, Peking University, Beijing, China. [5]Department of Mathematics, University of Exeter, Exeter, UK. [6]Joint Centre for Excellence in Environmental Intelligence, Met Office, University of Exeter, Exeter, UK. [7]Environment and Sustainability Institute, University of Exeter, Penryn, UK. [8]School of Life Sciences, Faculty of Science, The Chinese University of Hong Kong, Hong Kong SAR, China. [9]Present address: Department of Earth and Atmospheric Sciences, University of Houston, Houston, TX, USA.
✉e-mail: amostai@cuhk.edu.hk

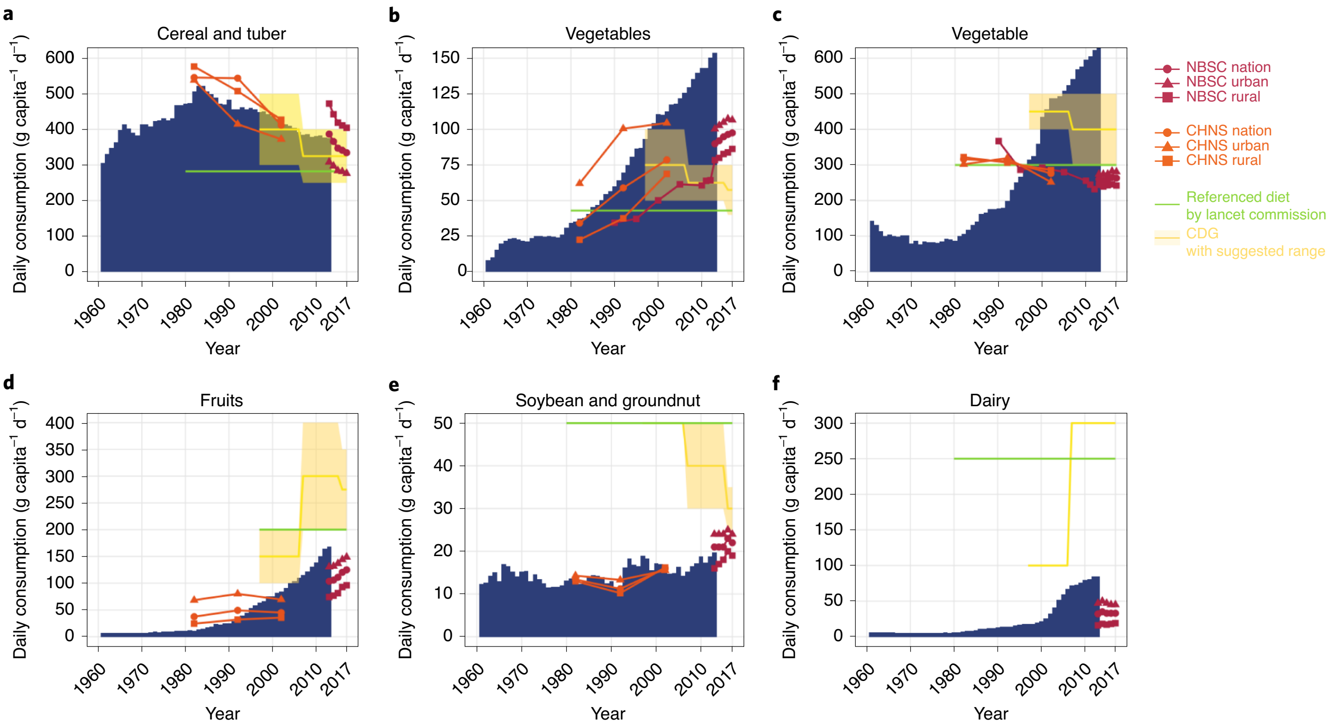

Fig. 1 | Trends in daily per capita food consumption. a–f, Daily per capita food consumption is shown for cereal and tuber (**a**), meat (**b**), vegetables (**c**), fruits (**d**), soybean and groundnut (**e**) and dairy products (**f**). Data from the FAO are shown as dark-blue bars. Data from NBSC and CHNS are shown in magenta and orange lines, respectively. Lines with solid circles represent nationwide values, while lines with triangles and boxes represent those of urban and rural households, respectively. Healthy reference diets from the CDGs (1997, 2007, 2016 versions) are shown in light-yellow lines, and shading indicates the recommended means and ranges. Healthy reference diets from the EAT–Lancet Commission (Lancet) are shown in light-green lines. The Lancet diet used 2,500 kcal d^{-1} as a basis to construct the reference diet scenario, and thus shows a relatively low level of suggested food intake compared with CDGs. Reference cereal and tuber intake from Lancet is the sum of intakes of whole grains and tubers or starchy vegetables.

also increased: calories from 8% to 23%, protein from 13% to 39% and fat from 43% to 61% (Supplementary Fig. 2). Meanwhile, people have reduced the proportion of staples (for example, cereal and tuber) in their daily diet, in favour of a more diverse diet including more livestock products, fruits and vegetables. As shown in Fig. 1, the daily per capita intake of staples has declined from ~500 g to ~380 g, while that of meat (from 30–40 g to 90–140 g), dairy products (from ~10 g to 30–90 g) and other non-starchy plant food (from 30–50 g to 120–180 g) has generally increased. Overall, the Chinese diet has not only increased in quantity but also shifted from a plant-based diet towards a meat-intensive one over 1980–2010.

National production of different foods has risen but to different extents depending on the amplifying or counteracting effects from changes in per capita consumption, dietary composition, population growth, changes in feed production associated with the demand for meat, food losses and other minor factors (Fig. 2). Meat production has risen substantially from 15 Mt to 80 Mt (a 433% increase) over 1980–2010. By scaling up the national average diet in the 1980s from the corresponding population of 1.0 billion to the higher population of 1.4 billion in the 2010s and comparing with actual consumption, we differentiate between the rising demand for meat driven by population growth given the same 1980s diet and that driven by dietary changes. We estimate that rising population alone has increased the demand for meat by 5 Mt (33% increase from 15 Mt in the early 1980s), and the changing diet has increased it by 60 Mt (400% increase). As shown in Fig. 2a, staple production has increased relatively steadily from 300 Mt to 480 Mt (60% increase) because of the rising demand for animal feed (+105 Mt), population growth (+73 Mt) but reduced per capita consumption (−18 Mt); rising feed demand is found to be the leading factor. In 2010, on average, 34% of the national cereal production is used as feed, up from 26% in 1980 (Supplementary Fig. 3). The number can be even higher for specific crops, for example, 67% of maize and 49% of sweet potato produced are used as feed in 2010. The increasing feed production in response to the rising demand for meat can have impacts comparable to those of meat production, but few previous studies on diet–environment relationships have examined its implications. Here we include both meat and animal feed production to more comprehensively quantify the true environmental costs of increased meat consumption.

Diet-driven worsening of $PM_{2.5}$ pollution

We then investigate how much of the historical changes in NH_3 emission and $PM_{2.5}$ pollution in China could be attributable to dietary changes (including changes in both per capita consumption and dietary composition) on top of the rising total food consumption due to population growth from the 1980s to 2010s. On the basis of the model of Zhang et al.[23] accounting for crop- and livestock-specific NH_3 emissions via volatilization from nitrogen-based fertilizer and livestock manure from 21 crops and 6 types of livestock, we find that agricultural NH_3 emission has almost doubled from a total of 6.4 $TgNH_3$ to 12.1 $TgNH_3$ in China (Supplementary Table 4 and Supplementary Fig. 4). Through factorial model experiments, we estimate that dietary changes alone (excluding population growth) could increase agricultural NH_3 emission by 63% (equivalent to 4.0 $TgNH_3$ nationally), and population growth alone (in a counterfactual case without dietary changes) could increase it by 27% (1.7 $TgNH_3$). Among different aspects of dietary changes, the rising demand for meat (encapsulating the associated demand for animal feed crops) is the major driver of diet-related increase

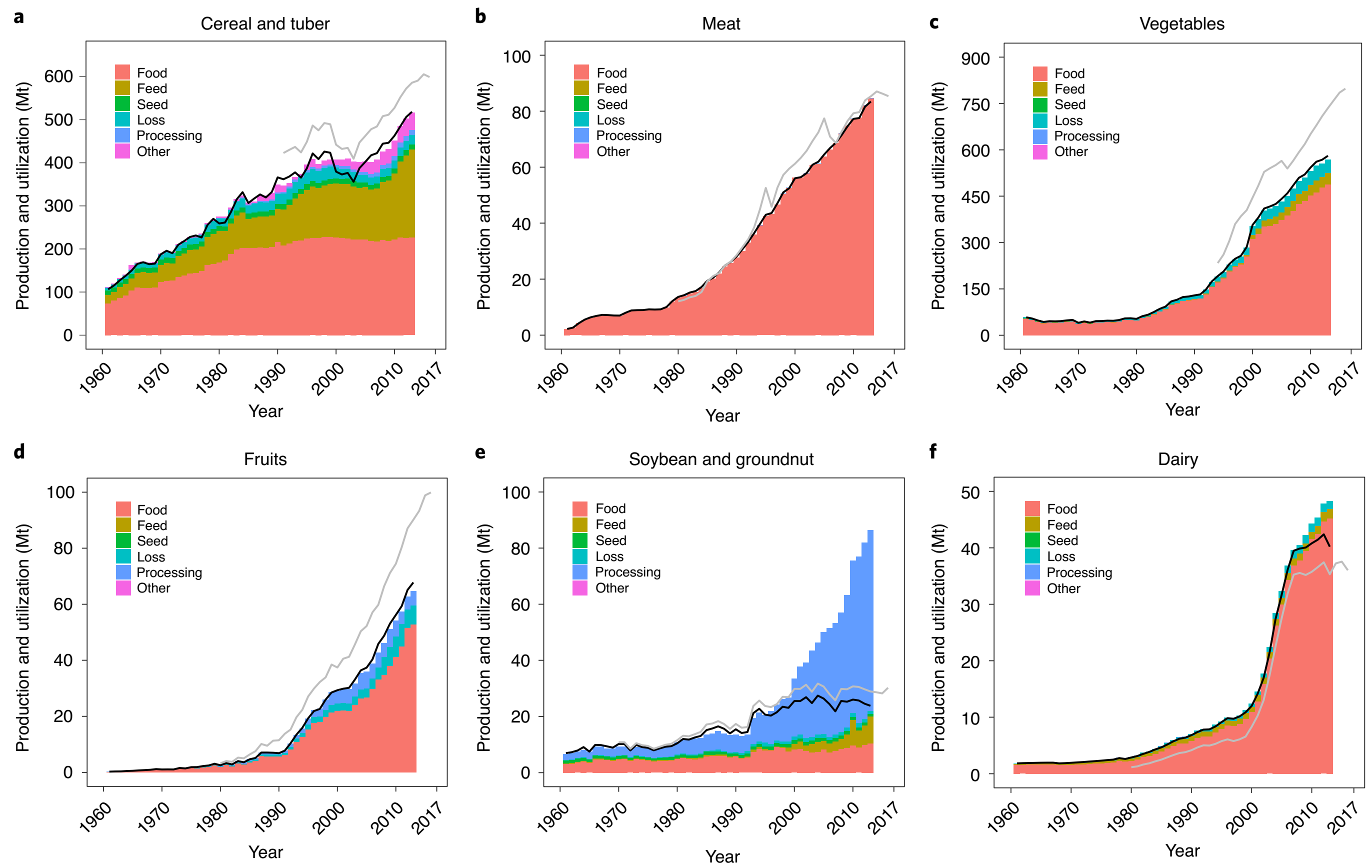

Fig. 2 | Trends in national food production and utilization. a–f, National production (shown as lines) and utilization (shown as stacked bars) are shown next for cereal and tuber (**a**), meat (**b**), vegetables (**c**), fruits (**d**), soybean and groundnut (**e**) and dairy products (**f**). Production data are from both the FAO (black line) and the NBSC (grey line). Utilization data are from the FAO, and distinctions are made between the quantities available for human consumption ('food'), fed to livestock ('feed'), used for seed ('seed'), lost during storage and transportation ('loss'), put to manufacture for food use and non-food uses ('processing'), and the rest ('other'). In **e**, a fraction of 'processing' soybean can be first pressed for oil, and then the remaining bean protein and fibre can be fed to animals, but as the exact fraction for China over the whole period was not fully documented, it was not included as animal feed in our analysis.

in agricultural NH_3 emissions, responsible for an increase of 2.7 $TgNH_3$. In contrast, the rising demand for food crops for direct human consumption causes an increase of 0.9 $TgNH_3$. Moreover, feed crop production is found to account for ~23% (1.2 Tg) of NH_3 emission from croplands in 2010, a substantial quantity often overlooked in previous studies that evaluated the nitrogen footprint of meat consumption.

NH_3 reacts with sulfuric and nitric acids (from SO_2 and NO_x oxidation) to form sulfate–nitrate–ammonium (SNA) aerosols that constitute an important fraction of $PM_{2.5}$ (refs. [24,25]). Being the dominant driver of agricultural NH_3 emission, dietary changes could thus greatly affect ambient $PM_{2.5}$ concentration. We input the above NH_3 emission changes driven by different factors into a high-resolution chemical transport model to simulate the corresponding changes in $PM_{2.5}$, and estimate that the increase in annual $PM_{2.5}$ concentration due to dietary changes alone can be up to ~10 $\mu g\,m^{-3}$ in the North China Plain and other major agriculture-intensive regions (Fig. 3b). This increase accounts for ~70% of the agriculturally driven $PM_{2.5}$ increase (3–15 $\mu g\,m^{-3}$, Fig. 3f) and ~20% of the total anthropogenic $PM_{2.5}$ increase driven by all sources over 1980–2010 (10–40 $\mu g\,m^{-3}$, Supplementary Fig. 5). Among the $PM_{2.5}$ increase, up to 3–8 $\mu g\,m^{-3}$ can be attributable to the demand for meat, compared with ~2–4 $\mu g\,m^{-3}$ attributable to the demand for food crops (Fig. 3c,d).

The emissions of SO_2 and NO_x in China have been reduced since 2012 by stringent control measures[18]. In our study, we estimated that SO_2 and NO_x emissions have reduced by 58% (from 24 Tg to 10 Tg) and 21% (from 24 Tg to 19 Tg) over 2010–2017, respectively (Supplementary Fig. 11). These reductions would alleviate the negative impacts of 1980–2010 dietary changes on $PM_{2.5}$ increase from +10–20 $\mu g\,m^{-3}$ to +5–10 $\mu g\,m^{-3}$, suggesting that recent SO_2 and NO_x emission reductions partly alleviate the diet-driven, NH_3-mediated enhancements in total $PM_{2.5}$.

Indirect versus direct health impacts of dietary changes

Changes in the average Chinese diet increasingly threaten both human and environmental health. Here we evaluate the 'indirect' health cost of dietary changes via air quality degradation based on the Global Exposure Mortality Model (GEMM)[26]. We do so by estimating the $PM_{2.5}$-related premature mortalities that could be attributable to the more meat-intensive 2010 diet as opposed to the 1980 diet. We estimate that 90,866 (95% CI 84,834–96,479), or ~5% of the 1.83 million (95% CI 1.68–1.97) $PM_{2.5}$-related Chinese premature deaths in China in 2010, could be attributable to dietary changes over 1980–2010, representing a non-negligible 'indirect' heath cost (Fig. 4). Among these, 66,117 (95% confidence interval (CI) 61,710–70,220) deaths are attributable to rising demand for meat, and 26,442 (95% CI 24,695–28,066) deaths to the increased emission from vegetable and fruit production and the reduced emission from cereal production. About 75% of the total indirect health cost is incurred by production of meat and associated animal feed.

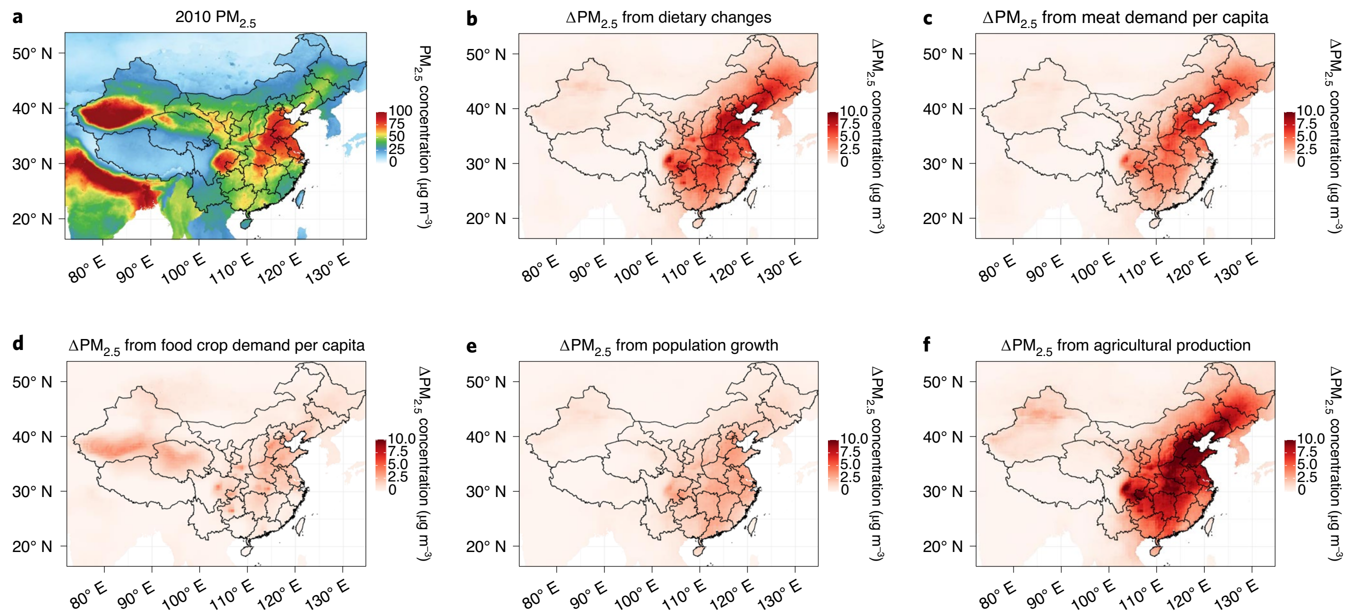

Fig. 3 | Worsening of $PM_{2.5}$ air quality due to dietary changes. a, Annual mean $PM_{2.5}$ concentrations from all sources in 2010. **b–f**, The 1980–2010 changes in $PM_{2.5}$ due to dietary changes (including demands for both meat and all crops) (**b**), rising demand for meat (including animal feed crops) (**c**), rising demand for food crops for direct human consumption (**d**), rising total food consumption due to population growth (**e**) and rising agricultural production overall (including both dietary changes and population growth) (**f**). These plots correspond to [2010] (**a**), [2010–POP] (**b**), [MEAT–POP] (**c**), [CROP–POP] (**d**), [POP–1980] (**e**) and [2010–1980] (**f**) in Methods.

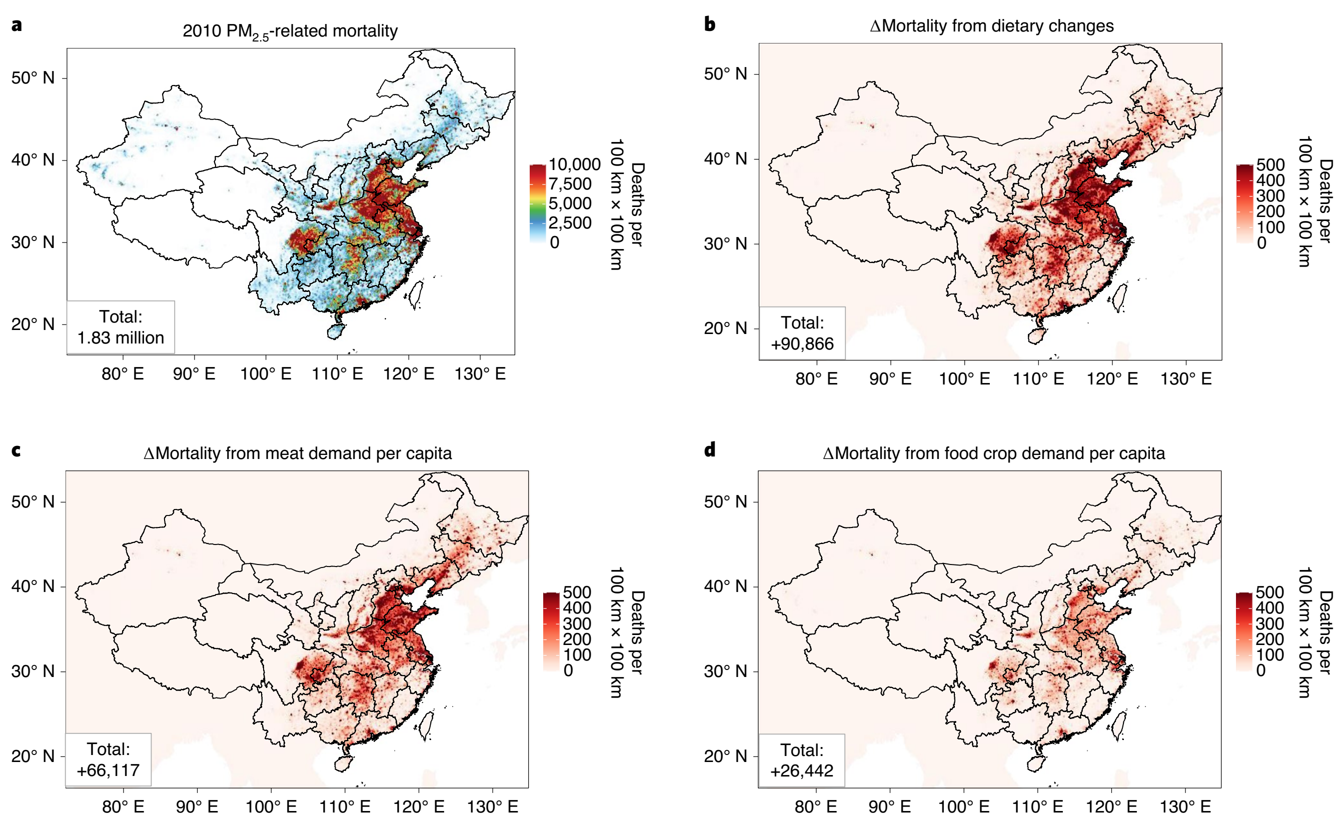

Fig. 4 | Indirect health cost of dietary changes related to $PM_{2.5}$ pollution. a, Chinese $PM_{2.5}$-related premature mortalities in 2010. **b–d**, The portions of those mortalities that could be attributable to 1980–2010 changes in dietary pattern as a whole (**b**), demand for meat (including animal feed crops) (**c**) and demand for food crops for direct human consumption (**d**). These plots correspond to [2010] (**a**), [2010–POP] (**b**), [MEAT–POP] (**c**) and [CROP–POP] (**d**) in Methods.

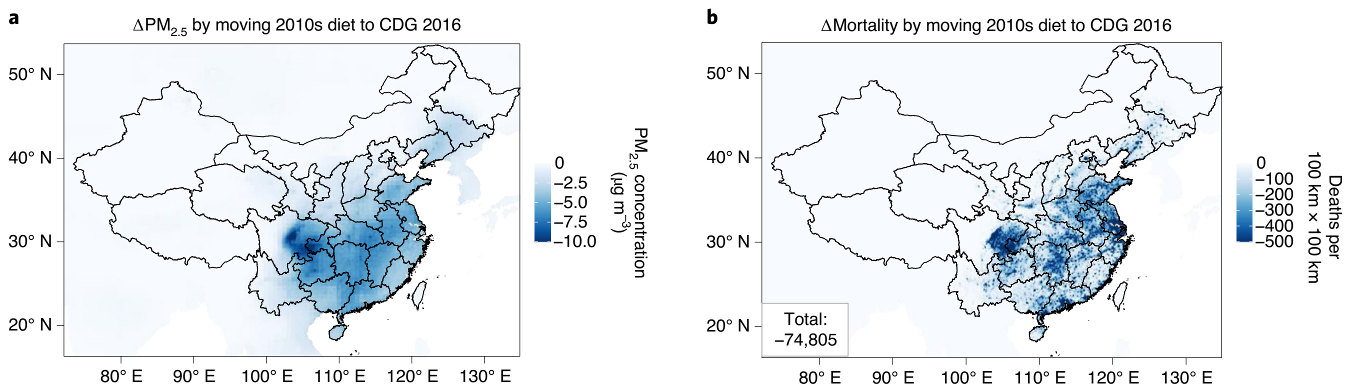

Fig. 5 | Environmental and indirect health benefits of less meat-intensive diets. a,**b**, Potential benefits of moving from the current 2010s diet to the healthier, less meat-intensive direct recommended by CDG 2016 in terms of changes in annual mean $PM_{2.5}$ concentrations (**a**) and $PM_{2.5}$-related premature mortalities (**b**). These plots correspond to [HEAL–2010] in Methods.

We also compare the 'indirect' health cost with the 'direct' diet-related health cost estimated by the Global Burden of Disease (GBD) study[3]. Direct health impact of the changing diet in China is complex, as some aspects of nutritional quality have improved while others worsened. For example, Chinese dietary trends over 1980–2010 (Fig. 1) initially contributed to reduced undernourishment rates and improved nutritional health, as the supplies of protein, iron and zinc from animal products as well as vitamins and fibre from fresh fruits and vegetables improved over time. However, these health benefits were later offset by the continuous shifts towards more unhealthy diets rich in calories and heavily processed and animal-sourced foods. Indeed, many diet-related risk factors increased from 1990 to 2010 according to the GBD. By considering eight diet-related risks related to food intake and one high body mass index risk largely resulted from meat-intensive diets from the GBD database for both 1990 and 2010 (Supplementary Table 8; earlier data were not available), we estimate that a total of 0.77 million premature deaths in 2010 is the direct heath cost of the dietary changes over 1990–2010. Therefore, the indirect health cost of dietary changes via air quality degradation over 1980–2010 is ~12% of the direct health cost in terms of deteriorating dietary quality and overconsumption in China since the 1990s.

Air-quality benefits of less meat-intensive diets

Less meat-intensive diets could, if widely adopted, reduce agricultural NH_3 emission and subsequent $PM_{2.5}$ pollution, offering indirect health benefits for the whole exposed population. In our case, we estimate that, if the average Chinese diet in 2010 is replaced by a less meat-intensive reference diet as recommended by CDG 2016, a 2.1 Tg decrease in NH_3 emission, 2–6 $\mu g\,m^{-3}$ decrease in $PM_{2.5}$ concentration (Fig. 5a) and 74,805 (95% CI 69,805–79,464) avoided $PM_{2.5}$-related deaths (~4% of the total in 2010) (Fig. 5b) can be expected. The reduction in NH_3 emission is mainly driven by reduced emission from meat and animal feed production, partly offset by increased emission from dairy and fruit production (see Fig. 1 for the difference between current and recommended intake). The reduction in $PM_{2.5}$ is particularly pronounced in winter, when the low air temperature favours SNA aerosol formation (Supplementary Fig. 6). Spatially, the $PM_{2.5}$ and mortality reductions are the most pronounced in the southern parts of China, where regional reductions in $PM_{2.5}$ and mortalities can be up to 15% and 10%, respectively.

Discussion

Global annual meat production has increased more than four fold from 72 Mt in 1961 to 318 Mt in 2014, and is projected to increase further to 374 Mt and 455 Mt by 2030 and 2050, respectively[27,28]. This increase reflects a growth in demand from both a rising global population (by 110% over 1960–2010; 39% over 2010–2050) and an increase in per capita consumption of meat (for example, from 20 kg to 42 kg per year over 1960–2010; from 42 kg to 49 kg per year over 2010–2050) linked to growing income and changing dietary preferences[27]. Although each country and population group has their own trajectory of meat production changes over the past 50 years, the largest change globally has been in East Asia, particularly China. The ongoing Chinese dietary transition characterized by increasing meat and declining whole grain intake has been proven to adversely affect nutritional health[3,29]. Air-quality degradation caused by diet-driven production changes can further threaten respiratory health.

Our work demonstrates a previously unquantified indirect health impact of Chinese dietary changes via air-quality degradation, which we find to be smaller than but within an order of magnitude of the direct health cost via malnutrition. We note specifically that the direct and indirect health costs are unevenly distributed demographically and geographically. The direct health cost is borne mostly by those who can afford more meat consumption, that is, more affluent population in the more developed regions of China such as Guangdong, Shanghai, Zhejiang and Fujian (66–93 kg meat per capita; 94,000–157,000 CNY gross domestic product per capita in 2019[21]). However, the indirect health cost is borne mostly by the population within and downwind of the less developed, major agricultural regions such as Hebei and Henan, who on average have lower income (~50,000 CNY gross domestic product per capita in 2019[21]) and consume less meat (29–34 kg meat per capita in 2019[21]). For example, the indirect health cost is 40–76 deaths per million population for Guangdong, Shanghai, Zhejiang and Fujian, but 82–102 deaths per million population for less developed, agricultural regions Hebei and Henan. This inequality aspect demonstrates an ethical dimension of meat consumption that deserves attention from a public policy perspective.

We show also that ~23% of NH_3 emission from croplands is due to feed crop production, which is an aspect overlooked by most previous studies. By including rising feed production for animal husbandry, we derive a more complete estimate of the NH_3 and $PM_{2.5}$ footprints of meat consumption in China. Globally, ~36% of cereals produced go into animal feed production, a substantial portion of the world's coarse grains[20]. Developed countries account for most of such use of coarse grains so far, but developing countries are catching up, accounting for ~42% of the world total in 2010, up from ~22% in 1980[28]. They will continue to increase their share in the global animal feed production of coarse grains, to ~56% by 2050[28].

China's changing food habits now have great global implications, as China increasingly imports agricultural products from other countries[20,30]. Although the import quantity is less than 5% of the domestic production for meat and major crops (except soybean), the amount and expenditure of import is rising year by year[20]. Of the large amount of agricultural imports, soybeans secure the most weight. In 2017, most (94%) of the soybeans imported into China came from Brazil (53%; 51 Mt), the United States (34%; 33 Mt) and Argentina (7%; 7 Mt)[20]. Meat import begins to show obvious increases in about 1994, which grows at 0.05 Mt yr^{-1} over 1994–2010 and increases to 0.29 Mt yr^{-1} over 2010–2017[20]. With its agricultural imports now topping the world, the indirect health impacts of Chinese dietary choices will become more widespread beyond China's borders. The trajectories of agricultural trade and its environmental and embedded health impacts worldwide should be further explored. For example, shifting domestic production towards crops with higher nitrogen-use efficiency (for example, soybean) while importing more crops with low nitrogen-use efficiency (for example, maize, fruits and vegetables) can relieve domestic nitrogen pollution, but can cause greater environmental burdens on the exporting countries and expose China's food security more to the intermittent risks of the international market[31].

In the 1980s, a top priority of China was to satisfy the people's basic food demand. Over the recent and coming decades, however, as the undernourishment rate is substantially decreasing[20], a more sustainable path for production and consumption in the food system is urgently needed. The current trajectory of the food system in China needs to be altered to reduce its effects on both human and environmental health domestically and worldwide. Efforts to improve food production processes[30,32–34] to cut reactive nitrogen emissions on the supply side will remain important, but we show further that changing food consumption patterns on the demand side can serve as a complementary approach to mitigate the environmental impacts of the agricultural sector. Addressing dietary patterns and food waste, the two most prominent demand-side measures in ours and other studies[35–39], will be key. Similar to our study but for a different region, Domingo et al.[39] found that 10,700 annual premature deaths related to $PM_{2.5}$ in the United States can be prevented by adopting the less meat-intensive EAT–Lancet Commission diets; they did not quantify the historical contribution of dietary changes to US air quality, but examined the implications of various possible supply-side and demand-side food development pathways for the future, which certainly warrant investigation for the Chinese counterpart. Although moving to healthier diets may temporarily decelerate agricultural economic growth, the benefits from cleaner air and healthier nutritional status would potentially enhance productivity and more than offset such economic drawbacks in the long run.

While reversing the trend in rising meat consumption may reap multiple human and environmental health benefits, recommendations regarding dairy consumption may be more nuanced. Dairy consumption is still low relative to the two recommended diets (Fig. 1). However, much of the Chinese population are lactose intolerant[40–42]. The recommendation to increase dairy consumption may reflect Eurocentric nutritional research and policies, and thus should be viewed with scepticism for a population for whom milk consumption can lead to poor digestion and other adverse health outcomes[40–42]. It would also further contribute to emissions from animal husbandry, thereby counteracting the environmental benefits of reducing meat consumption.

Demand-side controls of agricultural NH_3 emission in our study also provide a new perspective to alleviate $PM_{2.5}$ pollution in China. The emissions of SO_2 and NO_x have been reduced by ~60% and ~20% in China over 2013–2017 by stringent control measures[18], and energy- and transport-related air pollutant emissions are expected to be reduced substantially in the future through adoption of renewable energy and electric vehicles[43,44]. This would on the one hand partly alleviate the rise in $PM_{2.5}$ per unit NH_3 emission enhancement, but on the other hand give more urgency to NH_3 control relative to SO_2 and NO_x control. Curbing agricultural NH_3 emission by modifying the food system and consumption patterns should still be emphasized in the efforts to mitigate regional $PM_{2.5}$ pollution[45,46]. China, the largest meat-consuming country[20], is an archetypal case to study the diet–environment–health nexus, and the indirect health impacts revealed in our study have implications for any middle- and high-income countries with growing and high meat consumption. Overall, our results underscore the need for a planetary health framework that incorporates sustainable food production and consumption; only then can we simultaneously achieve food security and human and environmental health.

Methods

Food and dietary data. Food production and dietary consumption data were collected from the FAO Food Balance Sheets (FBS)[20], the 2018 China Statistical Yearbook from the NBSC[21] and the CHNS[22]. Per capita dietary supply data in the FAO FBS were scaled by conversion factors accounting for food waste and inedible parts following the report of Gustavsson et al.[47], to match definitions in the actual food intake data from NBSC and CHNS (see Supplementary Information for details). Reference diets were from both the CDG[19] and the EAT–Lancet Commission[48] (Supplementary Table 1). We took the average of these data over 1980–1984 (denoted as '1980s') and over 2010–2014 (denoted as '2010s') to represent the food and dietary status of early 1980s and early 2010s. The average 2010s diet is composed of around 380 g cereal and tuber, 140 g meat, 600 g vegetables, 150 g fruits, 18 g soybean and groundnut, 80 g dairy products and others, on a daily basis; the average 1980 diet is composed of around 420 g cereal and tuber, 25 g meat, 100 g vegetables, 10 g fruits, 12 g soybean and groundnut, 10 g dairy products and others.

Production scenarios. FAO FBS bridges per capita dietary data with national production by accounting for the allocation of total dietary energy supply among the food and non-food uses and losses, changes in stock, and international trade. As the agricultural trade quantity is less than 5% of the domestic production for meat and major crops (except soybean), we assume that domestic production is largely driven by domestic demands in this study. First, we merged FAO FBS national production with the NBSC crop- or livestock-specific production for 31 provincial-level administrative units in mainland China, for the 1980s (denoted as [1980]) and 2010s (denoted as [2010]). Second, scaling factors accounting for provincial population changes from the 1980s to 2010s were applied to the 1980s food production to obtain [POP]. Starting with [POP], we further applied scaling factors accounting for changes in the 'food' proportion of crop production to obtain [CROP], and in meat production and the 'feed' proportion of crop production to obtain [MEAT] (see Supplementary Table 3 for details). This created a set of counterfactual scenarios to isolate various factors driving the increases in production, such as population growth, changes in per capita consumption and dietary composition, and contribution from different food and crop items. [2010–1980] represents rising agricultural production as a whole (including both dietary changes and population growth). [POP–1980] represents rising total food consumption due to population growth. [2010–POP] represents dietary changes (including demands for both meat and all crops). Within different forms of dietary changes, [CROP–POP] represents rising demand for food crops for direct human consumption, and [MEAT–POP] represents rising demand for meat (including animal feed crops). Next, we adopted CDG 2016 to construct a production scenario with a healthier, less meat-intensive reference diet [HEAL]. [HEAL–2010] represents production changes by moving from the current 2010s to the reference diet. Finally, based on the 2010s production–emission model in Zhang et al.[23], we bridged each of the six production scenarios ([1980], [POP], [CROP], [MEAT], [2010] and [HEAL]) with a corresponding NH_3 emission scenario through factorial model experiments.

NH_3 emission scenarios. We used the bottom-up inventory of Chinese NH_3 emission of Zhang et al.[23] to calculate agricultural NH_3 emission from nitrogen-based fertilizer for 21 crops (including early/late rice, spring/summer maize, spring/winter wheat, potato, sweet potato, soybean, groundnut, rapeseed, tobacco, cotton, other crops, vegetables, apple, banana, citrus, grape, pear and other fruits) and from livestock manure from indoor (via housing, storage and spread stages) and outdoor sources, allowing us to derive crop- and livestock-specific NH_3 emission. First, 2010s NH_3 emission ([2010]) as the baseline scenario was modelled using detailed information on crop-specific fertilizer application rates, a process-based mass-flow approach for manure, and meteorological controls of emission factors. Next, we scaled the baseline fertilizer and manure using provincial ratio of food-specific production ([1980]/[2010], [POP]/[2010], [CROP]/[2010] and [MEAT]/[2010]) to obtain four emission scenarios. Finally, we obtained an emission scenario with the CDG 2016 reference diet using the ratio of food supply [HEAL]/[2010]. Emission totals of each scenario are shown in Supplementary Table 4. Our estimates of NH_3 emission in 1980 and 2010 are in reasonable agreement with other emission estimates (Supplementary Table 5). As described in Zhang et al.[23], we used the EarthStat dataset of crop harvest area at 5 min × 5 min resolution in year 2000 (refs. [49,50]) but adjusted to 2010 level according to cropland changes in Moderate Resolution Imaging Spectroradiometer (MODIS), as a static land cover to construct NH_3 scenarios, and did not consider the possible increase in total planting area over the period owing to a lack of detailed cropland maps in the early 1980s. If the 8% national increase in total planting area over 1980–2010 is accounted for (Supplementary Fig. 8), we might expect that NH_3 increases would be more dispersed in the north-east and the south-west but more concentrated in the east and south, compared with the current spatial distribution of NH_3 increases. The hotspot of NH_3 increases in the North China Plain would not be affected much as cropland area there has been overall steady over the decades.

Chemical transport model and surface $PM_{2.5}$. GEOS-Chem global 3D chemical transport model[51] was used to estimate the surface $PM_{2.5}$ based on NH_3 emission from each scenario. We used the GEOS-Chem version 12.2.0 with full tropospheric chemistry, driven by assimilated meteorological fields from the Modern-Era Retrospective analysis for Research and Applications, Version 2 (MERRA-2) produced by the NASA Global Modeling and Assimilation Office, with a horizontal resolution of 0.5° latitude by 0.625° longitude and 47 vertical layers. GEOS-Chem has been used extensively in surface $PM_{2.5}$ simulations and evaluated with in situ and satellite-derived observations in previous studies, both globally[52,53] and in China[54]. Six $PM_{2.5}$ simulations were conducted using GEOS-Chem in accordance with each of the six NH_3 emission scenarios, where we kept anthropogenic non-agricultural emissions of SO_2, NO_x and other $PM_{2.5}$ precursors as well as meteorology at 2010 level to isolate the impacts from these emission changes alone. In addition, we simulated separately an addition scenario [1980*] where meteorology and anthropogenic emissions of SO_2, NO_x and other $PM_{2.5}$ precursors were at 1980 level to estimate the total changes in $PM_{2.5}$ from all sources (Supplementary Table 6). GEOS-Chem simulations of SNA aerosols may present excessive nitrate substitution owing to biases in aerosol pH[25,55,56]. To correct the possible high biases of simulated $PM_{2.5}$ in GEOS-Chem (Supplementary Fig. 7), we first adopted the estimates of $PM_{2.5}$ for 2010 from the Data Integration Model for Air Quality (DIMAQ) at 0.1° × 0.1° spatial resolution as reported by Shaddick et al.[17] for the baseline scenario [2010], and then scaled this by the ratios of GEOS-Chem-simulated concentrations for all other scenarios ([1980], [POP], [CROP], [MEAT] and [HEAL]) to [2010] to obtain the corresponding bias-adjusted $PM_{2.5}$ fields. Applying GEOS-Chem-simulated $PM_{2.5}$ ratios to scale DIMAQ year-2010 $PM_{2.5}$ was intended to reduce the influence of systematic high biases on the $PM_{2.5}$ differences between the two scenarios ($\Delta PM_{2.5}$), and to translate the simulated $PM_{2.5}$ scenarios into fine-resolution concentration fields that are suitable for estimating premature mortality. The final $PM_{2.5}$ results and $PM_{2.5}$-related mortality estimates were based on these bias-adjusted scenarios; if the unadjusted scenarios were used, we estimated that 85,846 premature deaths in China in 2010 could be attributable to dietary changes over 1980–2010; 62,328 deaths are attributable to rising demand for meat; and 72,048 avoided $PM_{2.5}$-related deaths can be expected if the average 2010 Chinese diet is replaced by the CDG 2016 reference diet. These estimates are not statistically different from those using the adjusted scenarios. Furthermore, we tested the sensitivity of diet-driven $PM_{2.5}$ rise to further reductions in SO_2 and NO_x emissions post-2010 using GEOS-Chem at a 2° × 2.5° resolution (Supplementary Fig. 11), but food-related NH_3 emissions had to be kept at 2010 levels because of incomplete data post-2010.

Indirect health cost. The health impacts of ambient $PM_{2.5}$ were estimated using the concentration–response functions with the GEMM[27], which determined the association between $PM_{2.5}$ exposure and the probability of death. We used the GEMM for non-communicable diseases (NCDs) plus lower respiratory infections (LRIs) (denoted as GEMM NCD + LRI). Population count, population age and baseline mortality rates were kept constant among scenarios to estimate the variations due to changes in exposure only. As a comparison, we also adopted the GEMM for five specific causes (chronic obstructive pulmonary disease, ischaemic heart disease, lung cancer, stroke and LRIs) of death (denoted as GEMM 5-COD)[27]. In the main text, we presented estimates from the GEMM NCD + LRI, and its comparison with the GEMM 5 COD can be found in Supplementary Table 7.

Direct health cost. The direct health cost of dietary changes via malnutrition was calculated with eight dietary risks (including diet low in fruits, diet low in vegetables, diet low in whole grains, diet low in nuts and seeds, diet low in milk, diet high in red meat, diet high in processed meat and diet low in legumes) that are related to food intake and high body mass index risk (Supplementary Table 8), excluding those related to nutrient intake (for example, diet high in sodium, diet low in seafood omega-3 fatty acids, diet low in fibre, diet low in polyunsaturated fatty acids, diet low in calcium, diet high in *trans* fatty acids and diet high in sugar-sweetened beverages) from Afshin et al.[3]. We obtained the mortality rate in 2010 and 1990 for each of the above-mentioned dietary risks, and calculated the mortality change due to dietary changes alone over 1990-2010 (excluding the effect of population growth) as the product of the change in mortality rate from 1990 to 2010 times the year-2010 population.

Uncertainty estimates. We quantified the uncertainties for each part of the methods. First, the percentage errors for food production arising merely from the differences between the two databases, FAO and NBSC, are 15%, 7.6%, 27%, 39%, 21% and 32% (using FAO as baseline) for staples, meat, vegetables, fruits, soybean and groundnut, and dairy, respectively (Supplementary Table 2). The errors coming from cross-database differences (that is, production differences between black and grey curves in Fig. 2) are generally smaller than the relative temporal changes in production for each database (that is, production differences between 1980 and 2010 for each of the black and grey curves in Fig. 2). Whichever database we used as the baseline, the outcoming relative temporal changes from the baseline, which we focused on, would be similar because the two databases show very similar trends. Second, the error of NH_3 estimates arising from inter-inventory differences amounts to 22%; considering all relevant inventories, the 1980–2010 percentage change in national NH_3 emission is +78% ± 29% (+89% in this study; Supplementary Table 5). If the influences of changing fertilizer types are accounted for, we expect a 14% larger NH_3 emission from fertilizer and thus a 6% larger total NH_3 emission back in the 1980s (Supplementary Table 4 and Supplementary Figs. 9 and 10); this source of errors is much smaller than that from inter-inventory differences. Next, the normalized mean error of the DIMAQ-adjusted $PM_{2.5}$ (relative to observations) is ~16% (Supplementary Fig. 7), and simulating $PM_{2.5}$ using alternative baseline NH_3 inventories would add another ~3% to the error (Supplementary Fig. 12); these can be treated as the uncertainty associated with $PM_{2.5}$ simulations. The excessive nitrate substitution in the model also causes uncertainty in $PM_{2.5}$ estimates. Last, the uncertainty for $PM_{2.5}$-related deaths is given as 95% confident intervals, which account for variability of mortality parameters[27]. If errors in NH_3 and $PM_{2.5}$ estimates are propagated, the mortality estimates can have an error of up to ±50%, which does not alter the major conclusions of this study. Improvements in future studies include consolidating food and dietary databases from various sources, considering decadal changes in planting areas and fertilizer types in NH_3 emission inventory, and addressing excessive nitrate substitution in $PM_{2.5}$ simulations, among others.

Data availability

The datasets generated during this study are available on the Zenodo repository (https://doi.org/10.5281/zenodo.5643549). Source data are provided with this paper.

Code availability

Programming language R version 4.1.0 (ref. [57]) was used for analysis and visualization in this study. A publicly released map of China was obtained from the National Geomatics Center of China (http://www.ngcc.cn/ngcc/html/1/391/392/16114.html), and all map-related operations were performed using programming language R version 4.1.0 (ref. [57]). The R code is available on the Zenodo repository (https://doi.org/10.5281/zenodo.5643549). The code for the GEOS-Chem chemical transport model is available on the GitHub repository (https://github.com/geoschem/geos-chem).

Received: 26 May 2020; Accepted: 10 November 2021;
Published online: 16 December 2021

References

1. He, P., Baiocchi, G., Hubacek, K., Feng, K. & Yu, Y. The environmental impacts of rapidly changing diets and their nutritional quality in China. *Nat. Sustain.* **1**, 122–127 (2018).
2. Lam, H.-M., Remais, J., Fung, M. C., Xu, L. & Sun, S. S. M. Food supply and food safety issues in China. *Lancet* **381**, 2044–2053 (2013).
3. Afshin, A. et al. Health effects of dietary risks in 195 countries, 1990–2017: a systematic analysis for the Global Burden of Disease Study 2017. *Lancet* **393**, 1958–1972 (2019).
4. Stocks, T. et al. Trends in adult body-mass index in 200 countries from 1975 to 2014: a pooled analysis of 1698 population-based measurement studies with 19.2 million participants. *Lancet* **387**, 1377–1396 (2016).
5. Zhou, M. et al. Mortality, morbidity, and risk factors in China and its provinces, 1990–2017: a systematic analysis for the Global Burden of Disease Study 2017. *Lancet* **394**, 1145–1158 (2019).
6. Tilman, D. & Clark, M. Global diets link environmental sustainability and human health. *Nature* **515**, 518–522 (2014).
7. Poore, J. & Nemecek, T. Reducing food's environmental impacts through producers and consumers. *Science* **360**, 987–992 (2018).
8. Springmann, M. et al. Options for keeping the food system within environmental limits. *Nature* **562**, 519–525 (2018).
9. Clark, M., Springmann, M., Hill, J. & Tilman, D. Multiple health and environmental impacts of foods. *Proc. Natl Acad. Sci. USA* **116**, 23357–23362 (2019).

10. Steffen, W. et al. Planetary boundaries: guiding human development on a changing planet. *Science* **347**, 1259855 (2015).
11. Ciais, P. et al. in *IPCC Climate Change 2013: The Physical Science Basis Carbon* (eds Stocker, T. F. et al.) Ch. 6 (Cambridge Univ. Press, 2013).
12. Paulot, F. et al. Ammonia emissions in the United States, European Union, and China derived by high-resolution inversion of ammonium wet deposition data: Interpretation with a new agricultural emissions inventory (MASAGE_NH3). *J. Geophys. Res. Atmos.* **119**, 4343–4364 (2014).
13. Lelieveld, J., Evans, J. S., Fnais, M., Giannadaki, D. & Pozzer, A. The contribution of outdoor air pollution sources to premature mortality on a global scale. *Nature* **525**, 367–371 (2015).
14. Bauer, S. E., Tsigaridis, K. & Miller, R. Significant atmospheric aerosol pollution caused by world food cultivation. *Geophys. Res. Lett.* **43**, 5394–5400 (2016).
15. Apte, J., Marshall, J. D., Cohen, A. J. & Brauer, M. Addressing global mortality from ambient PM2.5. *Environ. Sci. Technol.* **49**, 8057–8066 (2015).
16. Aunan, K., Hansen, M. H. & Wang, S. Introduction: air pollution in China. *China Q.* **234**, 279–298 (2018).
17. Shaddick, G. et al. Data integration for the assessment of population exposure to ambient air pollution for global burden of disease assessment. *Environ. Sci. Technol.* **52**, 9069–9078 (2018).
18. Zhang, Q. et al. Drivers of improved $PM_{2.5}$ air quality in China from 2013 to 2017. *Proc. Natl Acad. Sci. USA* **116**, 24463–24469 (2019).
19. Chinese Nutrition Society *Chinese Dietary Guidelines* (People's Medical Publishing House, 2012).
20. *FAOSTAT: FAO Statistical Databases*. http://www.fao.org/faostat/en/ (Food and Agriculture Organization of the United Nations, 2017).
21. *National Bureau of Statistics of China*. http://www.stats.gov.cn/ (China Statistics Press, 2020).
22. *The China Health and Nutrition Survey*. https://www.cpc.unc.edu/projects/china (The Carolina Population Center at the University of North Carolina at Chapel Hill and the National Institute for Nutrition and Health at the Chinese Center for Disease Control and Prevention, 2002).
23. Zhang, L. et al. Agricultural ammonia emissions in China: reconciling bottom-up and top-down estimates. *Atmos. Chem. Phys.* **18**, 339–355 (2018).
24. Schiferl, L. D. et al. An investigation of ammonia and inorganic particulate matter in California during the CalNex campaign. *J. Geophys. Res. Atmos.* **119**, 1883–1902 (2014).
25. Nenes, A., Pandis, S. N., Weber, R. J. & Russell, A. Aerosol pH and liquid water content determine when particulate matter is sensitive to ammonia and nitrate availability. *Atmos. Chem. Phys.* **20**, 3249–3258 (2020).
26. Yang, G. et al. Rapid health transition in China, 1990–2010: findings from the Global Burden of Disease Study 2010. *Lancet* **381**, 1987–2015 (2013).
27. Burnett, R. et al. Global estimates of mortality associated with long-term exposure to outdoor fine particulate matter. *Proc. Natl Acad. Sci. USA***115**, 9592–9597 (2018).
28. Alexandratos, N. & Bruinsma, *J. World Agriculture Towards 2030/2050: The 2012 Revision* (Food and Agriculture Organization of the United Nations, 2012).
29. Godfray, H. C. J. et al. Meat consumption, health, and the environment. *Science* **361**, eaam5324 (2018).
30. Du, Y. et al. A global strategy to mitigate the environmental impact of China's ruminant consumption boom. *Nat. Commun.* **9**, 4133 (2018).
31. Huang, G. et al. The environmental and socioeconomic trade-offs of importing crops to meet domestic food demand in China. *Environ. Res. Lett.* **14**, 094021 (2019).
32. Chen, X. et al. Producing more grain with lower environmental costs. *Nature* **514**, 486–489 (2014).
33. Fung, K. M., Tai, A. P., Yong, T., Liu, X. & Lam, H. M. Co-benefits of intercropping as a sustainable farming method for safeguarding both food security and air quality. *Environ. Res. Lett.* **14**, 044011 (2019).
34. Guo, Y. et al. Air quality, nitrogen use efficiency and food security in China are improved by cost-effective agricultural nitrogen management. *Nat. Food* **1**, 648–658 (2020).
35. Foley, J. et al. Solutions for a cultivated planet. *Nature* **478**, 337–342 (2011).
36. Bajželj, B. et al. Importance of food-demand management for climate mitigation. *Nat. Clim. Change* **4**, 924–929 (2014).
37. Hasegawa, T., Havlík, P., Frank, S., Palazzo, A. & Valin, H. Tackling food consumption inequality to fight hunger without pressuring the environment. *Nat. Sustain.* **2**, 826–833 (2019).
38. Rosenzweig, C. et al. Climate change responses benefit from a global food system approach. *Nat. Food* **1**, 94–97 (2020).
39. Domingo, N. G. G. et al. Air quality-related health damages of food. *Proc. Natl Acad. Sci. USA* **118**, e2013637118 (2021).
40. Wang, Y. G. et al. Prevalence of primary adult lactose malabsorption in three populations of northern China. *Hum. Genet.* **67**, 103–106 (1984).
41. Yang, Y. et al. The prevalence of lactase deficiency and lactose intolerance in Chinese children of different ages. *Chin. Med. J.* **113**, 1129–1132 (2000).
42. Smith, H. in *Moral Foods* (eds Leung, A. K. C, Caldwell, M. L., Ku, R. J. S. & Yano, C. R.) Ch. 12, 262-284 (University of Hawaii Press, 2000).
43. Cai, W. et al. The Lancet Countdown on $PM_{2.5}$ pollution-related health impacts of China's projected carbon dioxide mitigation in the electric power generation sector under the Paris Agreement: a modelling study. *Lancet Planet. Health* **2**, e151–e161 (2018).
44. Liang, X. et al. Air quality and health benefits from fleet electrification in China. *Nat. Sustain.* **2**, 962–971 (2019).
45. Liu, M. et al. Ammonia emission control in China would mitigate haze pollution and nitrogen deposition, but worsen acid rain. *Proc. Natl Acad. Sci. USA* **116**, 7760–7765 (2019).
46. Xu, Z. et al. High efficiency of livestock ammonia emission controls in alleviating particulate nitrate during a severe winter haze episode in northern China. *Atmos. Chem. Phys.* **19**, 5605–5613 (2019).
47. Gustavsson, J., Cederberg, C., Sonesson, U., Van Otterdijk, R. & Meybeck, A. *Global Food Losses and Food Waste* (Food and Agriculture Organization of the United Nations, 2011).
48. Willett, W. et al. Food in the Anthropocene: the EAT–Lancet Commission on healthy diets from sustainable food systems. *Lancet* **393**, 447–492 (2019).
49. Monfreda, C., Ramankutty, N. & Foley, J. A. Farming the planet: 2. Geographic distribution of crop areas, yields, physiological types, and net primary production in the year 2000. Glob. Biogeochem. Cycles 22, GB1022 (2008).
50. EarthStat. University of Minnesota's Institute on the Environment and the Ramankutty Lab at the University of British Columbia, Vancouver, http://www.earthstat.org/ (2015).
51. GEOS-Chem (https://geos-chem.seas.harvard.edu/).
52. Van Donkelaar, A., Martin, R. V. & Park, R. J. Estimating ground-level $PM_{2.5}$ using aerosol optical depth determined from satellite remote sensing. *J. Geophys. Res. Atmos.* **111**, D21201 (2006).
53. Shaddick, G. et al. Data integration model for air quality: a hierarchical approach to the global estimation of exposures to ambient air pollution. *J. R. Stat. Soc. Ser. C* **67**, 231–253 (2018).
54. Geng, G. et al. Estimating long-term $PM_{2.5}$ concentrations in China using satellite-based aerosol optical depth and a chemical transport model. *Remote Sens. Environ.* **166**, 262–270 (2015).
55. Vasilakos, P., Russell, A., Weber, R. & Nenes, A. Understanding nitrate formation in a world with less sulfate. *Atmos. Chem. Phys.* **18**, 12765–12775 (2018).
56. Weber, R. J., Guo, H., Russell, A. G. & Nenes, A. High aerosol acidity despite declining atmospheric sulfate concentrations over the past 15 years. *Nat. Geosci.* **9**, 282–285 (2016).
57. The R Project for Statistical Computing (R Core Team, 2021).

Acknowledgements

This work was supported by funding (MD18518) from the CUHK–Exeter Joint Centre for Environmental Sustainability and Resilience (ENSURE) awarded to the whole team, Research Grants Council (RGC) General Research Fund (14323116) awarded to A.P.K.T., and RGC Area of Excellence Scheme (AoE/M-403/16) awarded to H.-M.L.

Author contributions

A.P.K.T. and H.-M.L. conceived the concepts and strategies. A.P.K.T. devised the overall methodology and supervised the writing of the manuscript. X.L. processed and analysed food and agricultural data, conducted GEOS-Chem simulations, analysed results and drafted the manuscript. Y.C. and L.Z. provided the NH_3 emission model and scenarios. G.S. provided $PM_{2.5}$ for 2010 at $0.1° \times 0.1°$ spatial resolution, and participated in the writing of the manuscript. X.Y. and H.-M.L. participated in the discussion and writing of the manuscript.

Competing interests

The authors declare no competing interests.

Additional information

Supplementary information The online version contains supplementary material available at https://doi.org/10.1038/s43016-021-00430-6.

Correspondence and requests for materials should be addressed to Amos P. K. Tai.

Peer review information *Nature Food* thanks Eri Saikawa, Matthew Hayek and the other, anonymous, reviewer(s) for their contribution to the peer review of this work.