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RECEIVED
12 December 2020

REVISED
17 May 2021

ACCEPTED FOR PUBLICATION
28 May 2021

PUBLISHED
8 October 2021

TOPICAL REVIEW

The food we eat, the air we breathe: a review of the fine particulate matter-induced air quality health impacts of the global food system

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Keywords: fine particulate matter, global food system, air quality, emissions, premature mortality, mitigation

Abstract

The global food system is essential for the health and wellbeing of society, but is also a major cause of environmental damage. Some impacts, such as on climate change, have been the subject of intense recent inquiry, but others, such as on air quality, are not as well understood. Here, we systematically synthesize the literature to identify the impacts on ambient $PM_{2.5}$ (particulate matter with diameter $\leqslant$2.5 μm), which is the strongest contributor to premature mortality from exposure to air pollution. Our analysis indicates that the life-cycle of the global food system (pre-production, production, post-production, consumption and waste management) accounts for 58% of anthropogenic, global emissions of primary $PM_{2.5}$, 72% of ammonia (NH_3), 13% of nitrogen oxides (NO_x), 9% of sulfur dioxide (SO_2), and 19% of non-methane volatile organic compounds (NMVOC). These emissions result in at least 890 000 ambient $PM_{2.5}$-related deaths, which is equivalent to 23% of ambient $PM_{2.5}$-related deaths reported in the Global Burden of Disease Study 2015. Predominant contributors include livestock and crop production, which contribute >50% of food-related NH_3 emissions, and land-use change and waste burning, which contribute up to 95% of food-related primary $PM_{2.5}$ emissions. These findings are largely underestimated given the paucity of data from the post-production and consumption stages, total underestimates in NH_3 emissions, lack of sector-scale analysis of $PM_{2.5}$-related deaths in South America and Africa, and uncertainties in integrated exposure-response functions. In addition, we identify mitigation opportunities—including shifts in food demand, changes in agricultural practices, the adoption of clean and low-energy technologies, and policy actions—that can facilitate meeting food demand with minimal $PM_{2.5}$ impacts. Further research is required to resolve sectoral-scale, region-specific contributions to $PM_{2.5}$-related deaths, and assess the efficiency of mitigation strategies. Our review is positioned to inform stakeholders, including scientists, engineers, policymakers, farmers and the public, of the health impacts of reduced air quality resulting from the global food system.

Contents

1. Introduction

Global food demand increased threefold from 1960 to 2010 as a result of increasing population, rising incomes, and shifting dietary choices (Foley *et al* 2011, Tilman *et al* 2011). This demand has been met by intensive agricultural practices associated with 'Green Revolution' technologies, changing land management practices, and resource inputs as evidenced by a 700% increase in nitrogen fertilizer use, a 70% increase in irrigated cropland, and a 110% increase in land cultivation that now accounts for nearly 38% of Earth's terrestrial surface (Foley *et al* 2005, Ramankutty *et al* 2018). Consequently, agricultural intensification has resulted in widespread environmental damage including surface water eutrophication, groundwater contamination, hypoxia and the formation of dead zones in oceans, increased soil acidity associated with reduced crop productivity, biodiversity loss, climate change, and reduced air quality (Vermeulen *et al* 2012, Erisman *et al* 2013, Bauer *et al* 2016, Springmann *et al* 2018a).

Air pollution is the leading environmental risk factor for mortality, linked to 3.9 million premature deaths in 2017 from exposure to ambient fine particulate matter ($PM_{2.5}$, PM with diameter <2.5 μm) (Landrigan *et al* 2018, IHME 2020). Atmospheric $PM_{2.5}$ can result either through direct emissions as primary $PM_{2.5}$ or is formed through chemical reactions as secondary $PM_{2.5}$ from precursors that include ammonia (NH_3), nitrogen oxide (NO_x), sulfur dioxide (SO_2) and non-methane volatile organic compounds (NMVOC). Of all air pollutants, $PM_{2.5}$ is the strongest contributor to premature mortality, resulting largely from respiratory disorders, cardiovascular disease and stroke (Burnett *et al* 2018), and thus is widely regulated with the goal of reducing ambient $PM_{2.5}$ concentrations. $PM_{2.5}$ is short-lived in the atmosphere with a lifetime of a few days to a week, but it can be transported regionally, resulting in human health damage up to several thousand kilometers downwind from the source itself (Wang and Zhang 2014, Goodkind *et al* 2019).

Historically, emissions reductions of $PM_{2.5}$ and precursor pollutants have been achieved by regulating major anthropogenic sources, such as power plants, industries and transportation (Bachmann *et al* 2007). Of emerging concern is agriculture, which has been identified as a significant contributor to global ambient $PM_{2.5}$ (Bauer *et al* 2016, Giannadaki *et al* 2018) and is linked to nearly 20% of all global ambient $PM_{2.5}$-related deaths (Lelieveld *et al* 2015). In the United States, emissions from agriculture have been linked to 15%–25% of all $PM_{2.5}$-attributable deaths (Goodkind *et al* 2019, Thakrar *et al* 2020). Giannadaki *et al* (2018) presented an economic case to mitigate agricultural emissions in Europe, finding that a 50% reduction in emissions could reduce $PM_{2.5}$ premature deaths by 18%, with a saving of 89 billion USD.

Most research examining air pollution from the global food system focuses on agriculture (e.g. Aneja *et al* 2015), but the global food system is expansive and encompasses all life cycle stages of food production, use and disposal (Vermeulen *et al* 2012). Few studies have examined the human health damage that results from air pollution generated by the global food system (Sun *et al* 2017). Here, we present a systematic review and an order of magnitude estimate of the contribution of emissions from the global food system to ambient $PM_{2.5}$-attributable deaths. We expand beyond the historical focus on agricultural production to account for emissions from sectors associated with the pre-production, post-production, consumption and waste management of food. Specifically, we follow the causal chain of emissions to health impacts to (a) describe emission pathways and determine national-scale emission totals for 15 sectors within the food system that emit five pollutants of interest (primary $PM_{2.5}$, NH_3, NMVOC, NO_x, SO_2), (b) summarize studies that estimate impacts of sector-scale emissions on ambient $PM_{2.5}$ formation and $PM_{2.5}$-attributable deaths and (c) identify strategies to reduce $PM_{2.5}$ pollution burden within and outside farms. By adopting a system-scale approach that expands beyond the historical focus on agricultural production, our review establishes emissions contributions and $PM_{2.5}$-attributable deaths resulting over the life-cycle of the global food system.

2. Data and methods

To define the overall scope of this review, we first determined the five key stages that span the life-cycle of the global food system by building on the concept of Vermeulen *et al* (2012). We then identified emissions sectors within each stage of the food system based on the emissions categories defined by the EMEP/EEA (European Monitoring and Evaluation Programme by the European Environment Agency) inventory guidebook, and used the Nomenclature for Reporting to establish system boundaries for each sector (EEA 2016). We also identified and gap-filled the missing sector of land-use change. Our efforts resulted in 15 emissions sectors aggregated by five stages, as shown in figure 1: (a) pre-production: land-use change, fertilizer production, (b) production: on-farm energy use, manure management, grazing, fertilizer use, agricultural waste burning, and other, (on-farm handling of agricultural products, standing crop emissions), (c) post-production: food industry, retail and distribution, transportation, (d) consumption: commercial cooking, residential cooking (not reported in this review), and (e) waste: open burning, controlled disposal (open disposal, uncontrolled incineration, controlled incineration, landfilling and composting).

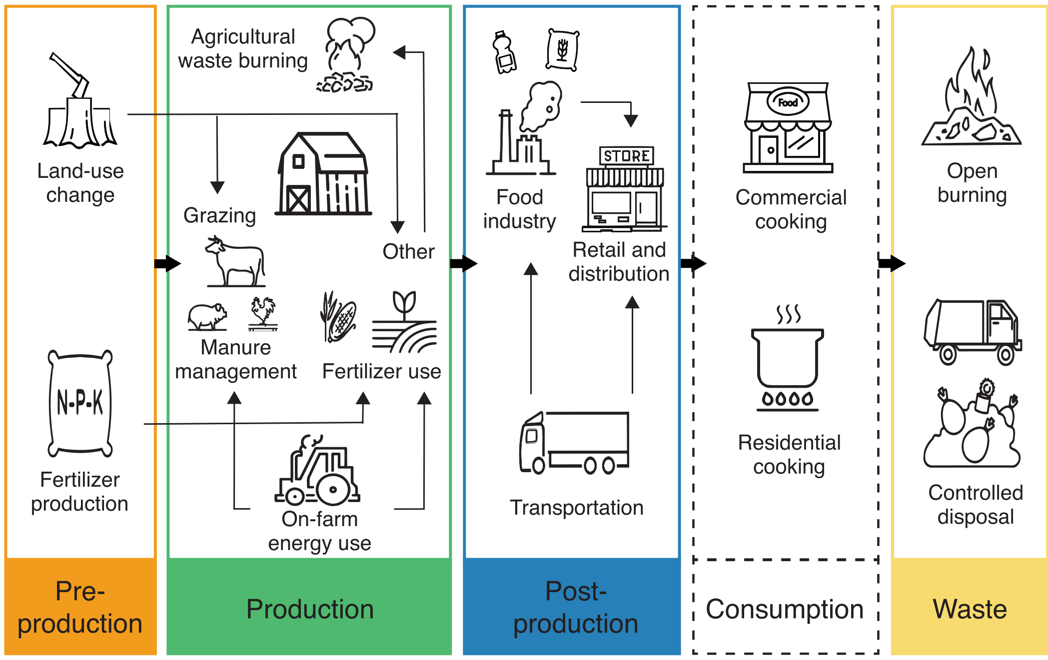

Figure 1. Schematic of the global food system. Identified are 15 emission sources from the following stages: (a) pre-production: land-use change, fertilizer production, (b) production: on-farm energy use, manure management, grazing, fertilizer use, agricultural waste burning, other (on-farm handling of agricultural products, standing crop emissions), (c) post-production: food industry, retail and distribution, transportation, (d) consumption: commercial cooking, residential cooking (marked in the dotted box as emission budgets; not reported in this review), and (e) waste: open burning, controlled disposal (open disposal, uncontrolled incineration, controlled incineration, landfill and composting).

We then adopted a systematic approach to identify and select relevant scientific literature and analyze relevant findings as defined by Uman (2011). First, the literature was located using scientific databases (Scopus, Google Scholar and Web of Science) by iteratively choosing the preliminary keywords of 'agriculture', 'emissions', 'food', 'air pollution', '$PM_{2.5}$' and 'premature mortality'. This search yielded 4746 peer-reviewed English language publications from the past decade (2009–2020). However, this process did not identify several key studies that examined specific emission sectors. Thus, we systematically expanded the search using additional keywords, by using a combination of each of the 15 emissions sectors, pollutants ($PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC), and mitigation strategies (see table 1) to identify an additional 1384 publications. We then applied the following criteria to sub-select relevant studies based on their abstract and introduction sections. Inclusion criteria were: (a) description of mechanisms and magnitudes of primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursor emissions, (b) air quality studies to quantify the enhancement of secondary ambient $PM_{2.5}$ and (c) mitigation strategies to reduce the $PM_{2.5}$ pollution burden. Exclusion criteria were: (a) hazardous air pollutant emissions from agriculture, (b) ground-level ozone formation and (c) sub-national scale studies using both modeling and measurement approaches to study contributions of the food system to ambient $PM_{2.5}$ concentrations. As a caveat, we do not explicitly show trade and associated emissions flows; instead, emissions are attributed to the geographic domains where sources are located. External to the scope of this review are related topics such as air pollution impacts on agricultural productivity, atmospheric deposition impacts of reactive nitrogen on ecosystems, pathways of global food demand and dietary shifts.

In addition to the archival literature, we also obtained data from highly curated institutional repositories to ensure consistency in data quality across geographic domains. The main data set of interest is the Emissions Database for Global Atmospheric Research (EDGAR4.3.2) that reports annual emissions of primary $PM_{2.5}$ and $PM_{2.5}$ precursors that are differentiated by activity, use of fuel and technology, pollutant type and end of pipe abatement (Crippa *et al* 2018). We also scoped the following databases: EMEP/EEA emissions factor database (EEA 2016) for sectoral-scale and pollutant-specific emissions factors, the World Bank for demographic (World Bank 2020) and waste management data (World Bank 2018a), FAOSTAT for data on land-use and land-use change, food production, fertilizer production and livestock management (FAO 2020a), the Global Fire Emissions Database (GFED4) for landscape fire data (van der Werf *et al* 2017), and environmentally extended input-output models including the World Input-Output Database (WIOD)

Table 1. List of search keywords implemented in this study. The search resulted in a database of 6130 records, of which 320 studies, data sets and reports were synthesized in this analysis.

First iteration		Subsequent iterations		
Agriculture	SO_2	Land-use change	Deforestation	Mitigation
Food	NO_x		Peatland	Future food demand
Emissions	NH_3		Agriculture driven land-use	Animal-based foods
Air pollution	Primary $PM_{2.5}$	Fertilizer production	Fertilizer	Plant-based foods
$PM_{2.5}$	NMVOC		Pesticide	Dietary choices
Premature mortality	IER functions	Energy use	Fuel use	Food waste
Excess deaths	Air quality model		Fuel type	Yield gap
			Machine units	Crop diversification
		Livestock	Manure storage	Energy-efficiency
			Manure housing	Regulations
			Manure management	Food pricing
			Manure application	Food labeling
			Grazing	Food portioning
		Crop	Fertilizer use	
			Pesticide use	
			Agricultural waste burning	
		Food industry	Food processing	
			Retail and distribution	
			Transportation	
		Cooking	Household air pollution	
			Commercial cooking	
		Waste	Landfill	
			Biogas digestate	
			Open burning	
			Compost	

(Timmer *et al* 2012) and EXIOBASE3.3.17 (Merciai and Schmidt 2016, 2018) for emissions from fertilizer production and the food industry.

Overall, of the 6130 records identified, 322 studies, data sets and reports have been synthesized in this review. Of these, only 19 studies established $PM_{2.5}$-attributable health damage either as premature deaths or economic damage from sectors relevant but not exclusive in terms of contributions to the global food system. Only two studies by Sun *et al* (2017) and Malley *et al* (2021) examine linkages between air quality and the global food system. Sun *et al* (2017) qualitatively linked the air quality impacts on the production and processing of food, human health in the form of productivity, and the role of markets, trade, and agricultural and energy policies, while Malley *et al* (2021) employed an air quality model to estimate the impacts of emissions from global food production on $PM_{2.5}$-related deaths. Here, we explicitly present national-scale emissions contributions of primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursors from the global food system, and synthesize studies that link these emissions to increases in ambient $PM_{2.5}$ exposure and premature deaths. We organize the rest of our review as follows: section 3 provides a description and estimates of sector-specific, national-scale emissions of primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursors; section 4 summarizes the resulting impacts on ambient $PM_{2.5}$ and premature mortality; section 5 identifies tools to reduce $PM_{2.5}$ pollution from the food system, and section 6 presents highlights and conclusions.

3. $PM_{2.5}$ pollution burden from the global food system

3.1. Sectoral-scale emissions: description and estimates

There are multiple approaches to developing air pollutant emissions inventories. A common approach is the use of emission factors, which represent how much pollutant is emitted by a unit of source activity. The emission-factor approach is readily scalable across regions and thus widely implemented, such as in the National Emissions Inventory for the United States (US EPA 2018) and EDGAR4.3.2 (Crippa *et al* 2018). Other approaches, particularly for agricultural production, include the use of process models that simulate physical, chemical and biological processes governing pollutant release at the field scale (Brilli *et al* 2017), and are integrated to develop regional-scale emissions inventories as input to air quality models (AQMs) (Cooter *et al* 2012, Balasubramanian *et al* 2015). Inverse modeling approaches have also been used recently to constrain emissions using observations assimilated from ground or satellite platforms, as in the case of improving the seasonality in NH_3 emissions (Zhu *et al* 2015b, van Damme *et al* 2018). We derive sector-specific, national-scale emissions of $PM_{2.5}$ and $PM_{2.5}$ precursors from EDGAR4.3.2 (Crippa *et al* 2018) that have been widely used as input to AQMs. EDGAR4.3.2 uses the Nomenclature for Reporting to establish system boundaries for sectors that follow initiatives such as the Convention on Long-Range Transboundary Air Pollution to minimize double-counting of emissions (EEA 2016). However, not all the emissions from the food system are accounted for in EDGAR4.3.2, and for many sectors, these emissions are not explicitly reported for the food system. We have thus supplemented data from other databases including GFED4 (van der Werf *et al* 2017) to estimate land-use change emissions and environmentally extended input-output models, such as WIOD (Timmer *et al* 2012) and EXIOBASE3.3.17 (Merciai and Schmidt 2016, 2018) for emissions from fertilizer production, food industry and waste, using similar system boundary definitions. We also identified the share of production for food versus non-food purposes based on data reported in the National Food Balance Sheets (FAO 2020b) and applied the fractional contribution of food to estimate emissions for relevant sectors. We present an in-depth discussion of each sector in section 3.1.1 and provide a summary in table 2.

3.1.1. Pre-production

3.1.1.1. Land-use change

Agriculture is the primary driver of deforestation especially in the tropical regions of South America and Southeast Asia (Fuchs *et al* 2018, Song *et al* 2018), and is largely driven by global food demand and international trade (Pendrill *et al* 2019). As of 2000, 50% of the habitable land has been diverted to grow food for human consumption and feed for livestock production (Ellis *et al* 2010). Increasing demand for food crops, cattle and timber has been linked to recent increases in forest clearing since 2017 in the Brazilian Amazon (De Oliveira *et al* 2020) and industrial oil palm productions in equatorial South-East Asia where 30% of the native peatland has been converted since 1990 (Miettinen *et al* 2016). Land clearing for shifting agriculture or permanent conversion to cropland is typically achieved through fires, while other practices, such as drainage of peatland increase susceptibility of these landscapes to fires (Martin 2019). Fires emit NO_x, $PM_{2.5}$, NH_3 and NMVOC, which are influenced by vegetation type and meteorology (Crutzen and Andreae 1990, Andreae and Merlet 2001, Akagi *et al* 2011), and have been linked to hazardous levels of $PM_{2.5}$ over the Amazon (Reddington *et al* 2015) and in Southeast Asia (Kiely *et al* 2019). Our review did not identify any studies that estimated primary $PM_{2.5}$ and precursor emissions resulting from food-demand driven land-use change. Instead, we designed an approach based on GFED4 that reports emissions that are derived using satellite-derived burned area and vegetation-type specific emissions factors (van der Werf *et al* 2017) and reported for 14 ecological regions that are aggregated to the following categories: savanna, grassland and shrubland, boreal forests, temperate forests, deforestation, peatland and agricultural waste burning.

We adopted the following method to derive $PM_{2.5}$ and precursor emissions totals for land-use change. First, we extracted national-scale emissions from GFED4 for Asia, Africa and South America that experience large-scale deforestation (Carter *et al* 2018) for the categories of savanna, grassland, shrubland, deforestation and peatland. Second, we extracted the extent of forest loss driven by wildfires, shifting agriculture and conversion for agriculture for the years 2012–2015 (World Resources Institute 2014). By combining forest loss data with GFED4, we identified emissions from shifting agriculture and permanent land-use change for agriculture. Finally, we identified the share of production for food versus non-food purposes based on the National Food Balance Sheets (FAO 2020b), and apply the fractional contribution of food to estimate emissions from land-use change. Similarly, GFED4 emissions for peatland were combined with the national-scale fractions of peatland fires on oil palm plantations (Miettinen *et al* 2016, Petersen *et al* 2016) and the fraction of oil palm diverted for food purposes (70%) (Lai *et al* 2012).

3.1.1.2. Fertilizer production

Agrochemicals including fertilizers, herbicides and pesticides have been widely used to maximize crop yields and for disease and pest management. The Haber–Bosch process, which was invented in the early 1900s, enabled the conversion of inert nitrogen to NH_3 to produce nitrogen fertilizers, which has dramatically altered agricultural production (Sutton *et al* 2011). Since 1960, croplands have received 300% more nitrogen from synthetic fertilizers than from natural biological nitrogen fixation, thus supporting nearly 48% of all crop production (Erisman *et al* 2008b). Global fertilizer production increased by 520% between 1960 and 2014 (FAO 2020a), resulting in large on-site emissions of primary $PM_{2.5}$ and NH_3, as well as emissions of $PM_{2.5}$, SO_2, NO_x and NMVOC from embodied energy. Satellite data have identified 158 hotspots of NH_3 emissions over fertilizer production sites in China, Ukraine, Iran and the United States (Van Damme *et al* 2018).

EDGAR4.3.2 aggregate emissions from nitrogen fertilizer production into the 'Industrial Processes and Product Use' category (EEA 2016). While the emissions-factor approach can be replicated by combining emissions factors for the production of NH_3 and other fertilizer types (EEA 2016) and scaled using agrochemical production statistics (FAO 2020a), it is challenging estimating emissions from embodied

Table 2. System-scale emissions inventory of the global food system. Annual emissions of primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursors (NH_3, NO_x, SO_2, NMVOC) are reported by sector. Descriptions of the 15 emissions sectors, choice of global emissions database and year of reporting are also included. N/A refers to pollutants from sectors where emissions are expected but unreported by the emissions databases. Note that this inventory does not include emissions from the consumption stage (commercial cooking and residential cooking) given the challenges in delineating contributions to indoor/household and ambient $PM_{2.5}$ pollution. As a result, the presented emissions budgets are underestimated at the global scale.

				Global estimates (Gg yr^{-1})				
Stage	Sector	Emissions database	Year	Primary $PM_{2.5}$	NH_3	NO_X	SO_2	NMVOC
Pre-production	Land-use change	GFED4	2012	14 000	1400	1450	870	N/A
	Fertilizer production	EXIOBASE3.3.17	2011	300	330	800	940	1700
Production	On-farm energy use	EDGAR4.3.2	2012	250	1	5800	3100	3200
	Manure management	EDGAR4.3.2	2012	339	16 600	325	N/A	19 000
	Grazing	EDGAR4.3.2	2012	0	8400	0	0	
	Fertilizer use	EDGAR4.3.2	2012	0	13 700	0	0	0
	Agricultural waste burning	EDGAR4.3.2	2012	6800	740	2100	360	3800
	Other	Emissions-factor approach	2012	100	50	16	0	1400
Post-production	Food industry	EXIOBASE3.3.17	2012	140	14	1400	870	30
	Transportation	Scaled from food industry emissions	2012	240	23	930	1500	60
	Retail and distribution		2012	93	9	360	580	22
Consumption	Commercial cooking	N/A						
	Residential cooking							
Waste	Open burning	EXIOBASE3.3.17	2012	1400	160	530	529	3200
	Controlled disposal		2012	0.3	520	9	0.7	140

energy use due to the lack of harmonized data on global fuel use for agrochemical production. Instead, we obtained data for emissions of primary $PM_{2.5}$, NO_x, SO_2, NH_3 and NMVOC from EXIOBASE3.3.17 that report emissions from mining of fertilizer minerals, and the production of nitrogen, phosphorus and other fertilizers (Merciai and Schmidt 2018). Emissions are reported for 44 countries and five regions outside those countries, which we distributed by population to gap-fill for the remaining countries. Finally, emissions were reduced proportionately to the percentage of agricultural commodity use for food versus non-food purposes derived from the National Food Balance Sheets (FAO 2020b). Our analysis excludes pesticide manufacturing as it typically occurs in a highly controlled environment to minimize direct health impacts, thus resulting in minimal $PM_{2.5}$-related emissions (EEA 2016).

3.1.2. Agricultural production

3.1.2.1. On-farm energy use

Energy is required on farms to power machinery, livestock housing and storage facilities. Diesel engines are widely used for powering tillage, planting, harvesting, irrigation, crop drying and transport operations. The input of energy to farms increased by 137% between 1961 and 2014 amounting to 2.6% of global energy use, while machinery and associated fuel use doubled (Pellegrini and Fernández 2018). In particular, large increases in machine stocks by 2400% in mainly irrigated countries including Bangladesh, China, India, Pakistan, Egypt and South Korea have been accompanied by a 50% increase in irrigated land. Of interest are emissions of primary $PM_{2.5}$, NO_2 and SO_2 resulting from fuel combustion. Several studies have examined contributions from off-road mobile sources, yet few have exclusively examined emissions from on-farm energy use. The use of agricultural machinery in China is linked to substantial emissions (250 Gg $PM_{2.5}$, 2.1 Tg NO_x and 25 Gg SO_2), coinciding with peak agricultural activities in April, June and October (Wang *et al* 2016, Lang *et al* 2018). However, such explicit emissions accounting for on-farm energy use are unavailable at global scales, as is the lack of harmonized on-farm energy use data classified by technology and end-use. In the United States, diesel was the typical fuel used for on-farm machinery and related operations (38%) with smaller contributions from electricity (16%), gasoline (15%) and natural gas (10%) (Brown and Elliott 2005). Following the emissions-factor approach, we combined national-scale, fuel-specific on-farm energy use (FAO 2020a) and scale using Tier-1 emissions factors for agricultural energy use (EEA 2016) to report national-scale emissions for primary $PM_{2.5}$, NO_x and SO_2. Emissions were reduced proportionally to the percentage of agricultural production for food versus non-food use based on the National Food Balance Sheets (FAO 2020b).

3.1.2.2. Livestock management

Human demand for animal-based food has quadrupled since 1961, with meat production increasing by 200% in Europe and North America, and significantly larger increases in Asia (1500%) and South America (530%) (FAO 2020a). Subsequently, manure-nitrogen production has increased by 520%, with regional contributions dominated by Asia (34%), Africa (17%) and South America (15%) (Zhang *et al* 2017a). Livestock systems are highly nitrogen inefficient, as a large fraction (45%–95%) of nitrogen from the feed is excreted as manure and urine (McQuilling and Adams 2015), which decomposes and is subsequently emitted as NH_3 through volatilization of nitrogen (Behera *et al* 2013). Livestock operations are also associated with primary $PM_{2.5}$ emissions from the movement of livestock within facilities (Ni *et al* 2009, Yang *et al* 2011), and trace emissions of NMVOC (Hobbs *et al* 2004) and SO_2 (Lim *et al* 2003).

Globally, NH_3 emissions from livestock management are attributed to the production of cattle (43%), goats and sheep (33%), swine (11%) and poultry (10%) (Zhang *et al* 2017a), and can occur at multiple stages in the livestock management system: from accumulated manure in housing, yard and storage facilities (31%–55%), land application for crop cultivation (23%–38%) and from livestock grazing (17%–37%) (Beusen *et al* 2008, Dämmgen and Hutchings 2008). The most important factors determining NH_3 emissions are the type of livestock, its age and the nitrogen content in the feed (Beusen *et al* 2008). Emissions from manure storage and handling depend on the surface area and bedding material. As a result, larger losses are observed in open housing with solid or slatted floors compared to cubicle houses, deep litter and closed manure storage systems (Dämmgen and Hutchings 2008). Emissions from manure application to crops are highly dependent on environmental conditions and application mode, with increased emissions positively correlated with higher temperatures, wind speeds and lower moisture content (Webb *et al* 2010).

NH_3 emissions from livestock rearing have received extensive attention in the development of emissions inventories and through multiple, targeted measurement campaigns in Europe and the United States (Slattery 2005). These efforts have resulted in detailed emissions factors that are differentiated by livestock type and manure management operation (housing, storage and handling, grazing and manure application to soils) (Battye *et al* 1994, EEA 2016), that are implemented in EDGAR4.3.2 (Crippa *et al* 2018). Additional approaches have been developed to better capture spatial and temporal heterogeneity in emissions. Semi-empirical models, such as the Farm Emissions Model fine-tune existing emissions factors by estimating NH_3 losses based on mass balances and mass transfer processes that are influenced by meteorology (McQuilling and Adams 2015). Process models have also been implemented to develop emissions inputs from manure management to AQMs (Deng *et al* 2015, Giltrap *et al* 2017). However, given the large data requirements to capture manure management systems and environmental conditions, and the need for calibrated models to capture region-specific variability, these approaches are yet to be scaled globally. Here, we obtain national-scale emissions of primary $PM_{2.5}$, NH_3 and NMVOC from EDGAR4.3.2 that are differentiated by livestock-type for the categories of manure handling and storage, manure application and grazing.

3.1.2.3. Fertilizer use

The application of synthetic fertilizers for crop cultivation is one of the most important land management practices to increase soil fertility and crop yields. Global nitrogen inputs to crops increased by 850% between 1960 and 2013 (Lu and Tian 2017) to meet the demand for food, animal feed and biofuels. Large regional variations exist in nitrogen use, ranging from 0.15–6 kg N ha^{-1} in sub-Saharan Africa to 100–200 kg N ha^{-1} in cropland in Asia (Lu and Tian 2017). Global NH_3 emissions increased from 1.9 to 16.7 Tg N between 1961 and 2010 (Behera *et al* 2013), 67% resulting from the cultivation of rice, corn, wheat and soybeans (Xu *et al* 2019). Depending on the fertilizer type, amount and mode of application, and weather and soil conditions, 1%–64% of the applied nitrogen can volatilize as NH_3 (Sommer *et al* 2004, Balasubramanian *et al* 2017), thus representing a major financial loss to farmers (Pan *et al* 2016). Urea, which is the most commonly used fertilizer globally (Behera *et al* 2013), has a volatilization potential 22%–55% higher than other nitrogen forms (Goebes *et al* 2003, EEA 2016, Pan *et al* 2016).

Similar to livestock rearing, NH_3 emissions from fertilizer use are estimated using the emission-factor approach as in EDGAR based on fertilizer-type specific emission factors (Crippa *et al* 2018). However, this approach introduces large uncertainties as it does not capture the impact of crop management and the resulting spatial and temporal heterogeneity that has been identified at the farm scale (Sommer *et al* 2004, Nelson *et al* 2017, Ti *et al* 2019). Studies have addressed this challenge through the use of process models to characterize region-specific spatial and temporal variations in NH_3 emissions (Cooter *et al* 2012, Balasubramanian *et al* 2015, Xu *et al* 2019), and through the use of inverse models to improve seasonality in NH_3 emissions (Paulot *et al* 2014, Zhu *et al* 2015b). However, global deployment of the process model and inverse model approaches is limited due to scalability issues that result from limited regional-scale data for agricultural management, and resulting uncertainties that are often of the same order of magnitude as the emissions-factor approach (Schiferl *et al* 2016, Balasubramanian *et al* 2020). Similar to the livestock sector, we thus obtained national-scale NH_3 emissions from EDGAR4.3.2 that were proportionally adjusted for contributions for food versus non-food purposes using data from National Food Balance Sheets (FAO 2020b).

3.1.2.4. Agricultural waste burning

Open burning of agricultural waste is a low-cost way to dispose of crop residues left over after harvesting, land clearing and pest control (Crutzen and Andreae 1990, Akagi *et al* 2011). Annual agricultural waste burning increased by 150% between 1960 and 2015 (FAO 2020a), releasing large amounts of primary $PM_{2.5}$ (1.76 Tg), NH_3 (0.6 Tg), SO_2 (0.11 Tg), NO_x (0.08 Tg) and NMVOC (0.11 Tg) (van der Werf *et al* 2017). Several studies have examined the impacts of agricultural waste burning at regional scales. In the United States, the burning of corn, cotton, bluegrass, rice, soybeans, sugarcane and wheat residues was linked to local increases in ambient $PM_{2.5}$ (Pouliot *et al* 2017). Similarly, the burning of rice, corn and wheat straw residue in China was linked to $PM_{2.5}$ emissions (Ni *et al* 2015), which may have been underestimated (Li *et al* 2017a). Burnt agricultural residue in India from managing rice (43%), wheat (26%), sugarcane (10%) and cereal residues (11%) (Ravindra *et al* 2019) has been linked to a 600% increase in ambient $PM_{2.5}$ during the harvest season (Jethva *et al* 2018). In Southeast Asia, rice straw burning dominated $PM_{2.5}$ emissions (95%–98%), largely driven by crop production in Indonesia (25%–39%), Vietnam (17%–30%), Myanmar (8%–19%) and Thailand (7%–16%). Emissions of primary $PM_{2.5}$, NH_3, NO_2, SO_2 and NMVOC from agricultural waste burning have been reported using the emissions-factor approach in both the GFED4 (van der Werf *et al* 2010) and EDGAR4.3.2 (Crippa *et al* 2018). Here, we obtain national-scale emissions from EDGAR4.3.2, which are proportionally adjusted for food versus non-food contributions by using data from the National Food Balance Sheets (FAO 2020b).

3.1.2.5. Other emissions

On-farm operations including plowing, tilling and harvesting, and on-farm handling and storage of agricultural products are typically associated with emissions of coarse PM that result from the attrition of dry plant particles, silica, biological species including molds, pollen, spores, bacteria, fungi and agrochemical residues. On-farm operations also emit primary $PM_{2.5}$ (Aneja *et al* 2009, van Grinsven *et al* 2013), with contributions ranging from 2%–5% of the total anthropogenic, primary $PM_{2.5}$ emissions in Europe (Erisman *et al* 2008a, Oenema *et al* 2012) and Canada (Pattey and Qiu 2012) to 15% in the United States (Penfold *et al* 2005). Crops also naturally emit NMVOC, including isoprene, monoterpenes and sesquiterpenes, among 50 other identified species as a part of normal growth (Lamb *et al* 1993, König *et al* 1995, Laothawornkitkul *et al* 2009) or as a defense mechanism that can be triggered during harvesting (Guenther *et al* 2000). Miscellaneous sources include emissions from pesticides and NH_3 emissions from treated straw that is used as ruminant feed. Here, we follow the EDGAR4.3.2 methodology to estimate on-farm primary $PM_{2.5}$ emissions and NMVOC emissions from standing crops by combining national-scale crop production data (FAO 2020a) with emissions factors (EEA 2016). We exclude emissions from pesticide application and treated straw as they are assumed to be negligible. These estimates are adjusted for food demand using data from the National Food Balance Sheets (FAO 2020b).

3.1.3. Post-production

3.1.3.1. Food industry

Food and beverage manufacturing (here, the 'food industry') includes industrial manufacturing of food ingredients and products that are processed and packaged typically for retail. The global food industry annually consumes 200 EJ energy (Ladha-Sabur *et al* 2019), accounting for 4% of the industrial energy consumption in OECD (Organisation for Economic Co-operation and Development) countries and 2% in non-OECD countries (EIA 2016). The reporting of sub-national-scale energy embodied in the food industry is fragmented and only for select commodities (Ladha-Sabur *et al* 2019). The industrial processing of food products emits primary $PM_{2.5}$, NH_3, NO_x, SO_2 and NMVOC (US EPA 1995) as a result of embodied energy use and on-site operations. While the emissions-factor approach can be implemented to estimate these emissions, the lack of harmonized data on national-scale fuel and technology used to power the food industry, and how food commodities are produced, limit these efforts. Here, we obtain data for emissions of primary $PM_{2.5}$, NH_3, NO_x, SO_2 and NMVOC from EXIBOASE3.3.17 for the production and processing of meat from cattle, poultry and pigs, vegetable oils and fats, dairy products, processed rice, sugar refining, beverages, seafood products and miscellaneous food commodities (Merciai and Schmidt 2016). These emissions are reported for 43 countries and for five regions for all other countries, which we distributed proportionally to the national population to gap-fill data.

3.1.3.2. Retail and distribution

Energy use in food retail is driven by business size, nature of products sold and use of equipment for on-site food preparation and preservation (Vermeulen *et al* 2012, Ladha-Sabur *et al* 2019). Commercial refrigeration is highly energy-intensive, accounting for 15% of global electricity consumption (James and James 2010). We identified only one study (hereafter DEFRA report) that reported primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursor emissions from food retail and distribution. The report provided relative emissions contributions for the food industry, retail and distribution, and food transportation, but was limited to the United Kingdom (Smith *et al* 2005). Given the lack of such data at the global scale, we combined the relative contributions from the DEFRA report with the national-scale food industry emissions derived from EXIOBASE3.3.17 to estimate national-scale food retail and distribution emissions. As a caveat, the United Kingdom is a high-income country. Thus, our approach will result in higher magnitudes of emissions than expected at the global-scale and be reflective of supply chain management trends that low-income countries may adopt in the future.

3.1.3.3. Transportation

The transportation of food or 'food miles,' is a popular albeit often misapplied, indicator to assess the sustainability of food commodities (Schnell 2013). While the impact of food miles on greenhouse gas (GHG) emissions (Pirog *et al* 2001, Weber and Matthews 2008) and along supply chains of specific commodities (Brodt *et al* 2013, Brunori *et al* 2016, Schmitt *et al* 2016) have been studied, the focus on air pollutant emissions is rather limited. We identified only one study reporting $PM_{2.5}$ emissions from food miles, which was limited to the United Kingdom (Smith *et al* 2005). Transportation modes have a significant impact on emissions, with lower reported emissions per km-tonne for food moved by ship and rail in comparison to cars and trucks. While food commodity flows by transportation modes are reported for Europe (Eurostat 2019) and the United States (Federal Highway Administration 2014), limited data coverage on transportation choice and fuel use at the global scale limits the estimation of primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursor emissions. Freight transport of goods including food commodities has been linked to $PM_{2.5}$-related health impacts resulting from emissions of $PM_{2.5}$ and NO_x (Liu *et al* 2019). It is imperative to establish the global-scale air quality impacts of transportation occurring as a result of food trade (Dalin and Rodríguez-Iturbe 2016), given that 25% of the food produced globally is traded (Odorico *et al* 2014). Similar to the retail and distribution sectors, we combine relative contributions of food transportation from the DEFRA report with national-scale food industry emissions estimates, without accounting for miles from retail to home.

3.1.4. Food preparation and consumption sectors

3.1.4.1. Commercial cooking

Several studies have examined the contribution of commercial cooking to ambient $PM_{2.5}$ pollution in urban settings (Robinson *et al* 2006, 2018, Gysel *et al*

2018), through emissions of ultrafine particles (PM with diameter <0.1 μm) that are retained longer in the lungs and cause more pulmonary infections than $PM_{2.5}$ (Schraufnagel 2020), and NMVOC in the form of *n*-alkanes, furans, lactones, polycyclic aromatic hydrocarbons and cholesterol (Rogge *et al* 1991). Commercial cooking often elevates $PM_{2.5}$, especially ultrafine fractions ($PM_{2.5} \leqslant 0.1$ μm) several orders of magnitude higher compared to the urban background and to larger extents than congested roadways (Robinson *et al* 2018) and smoking (Nasir and Colbeck 2013). These emissions are influenced by practices including cooking style, the temperature, duration of cooking and type of cooking oil (Abdullahi *et al* 2013, Torkmahalleh *et al* 2017). Commercial cooking impacts not only in-house workers, but elevates ambient $PM_{2.5}$ concentrations (50%–300%) and drives spatial patterns in $PM_{2.5}$ exposure in neighboring urban areas (Robinson *et al* 2018, Saha *et al* 2019), with disparate socio-economic impacts given the demographics of the population living in proximity to restaurants (Shah *et al* 2020).

Only the United States reports commercial cooking emissions that are classified by the equipment type and amount of food cooked (Roe *et al* 2004, US EPA 2018). Commercial cooking accounts for 1% of national $PM_{2.5}$ emissions resulting from underfired-char broilers (78%), conveyorized charbroilers (10%) and flat griddle frying (12%) (Roe *et al* 2004). Commercial food establishments account for a large fraction of the energy consumption (28%–34%) in the United States (Todd 2017), and this fraction is increasing globally (Fryar *et al* 2018). The lack of similar emissions reporting for other countries limits efforts to develop a global emissions inventory. Here, we do not quantify commercial cooking emissions, given data constraints and endemic challenges in delineating indoor-outdoor emissions contributions. However, given that this sector accounts for 1% of the $PM_{2.5}$ national emissions and an increasing shift towards consumption of food from commercial cooking, this source may be of increasing importance for urban air pollution, and should be revisited.

3.1.4.2. Household cooking

Much of the focus on cooking and $PM_{2.5}$ pollution has been on household air pollution resulting from solid fuel use, which is a major health risk in developing countries (Smith and Pillarisetti 2017, Goldemberg *et al* 2018). In 2017, 3.6 billion people, primarily in South Asia, East Asia and sub-Saharan Africa, were exposed to elevated household $PM_{2.5}$ concentrations resulting from the use of solid fuels, such as wood, charcoal, coal and other biomass (Health Effects Institute 2019). Similar to commercial cooking, household cooking emits primary $PM_{2.5}$, NMVOC and trace levels of NO_x and SO_2, that are dependent on fuel type (Sidhu *et al* 2017) and cooking practices, such as food and oil type, cooking temperature and duration, type and efficiency of cooking appliance, and indoor ventilation (Rehfuess *et al* 2011, Hu *et al* 2012).

A large body of the literature has examined emissions from solid fuel use in various settings. Example studies include laboratory measurements (Roden *et al* 2009, Preble *et al* 2014, Shen *et al* 2017), field measurements from uncontrolled in-home stoves in India (Pandey *et al* 2017, Menghwani *et al* 2019), China (Li *et al* 2007, Jiang and Bell 2008, Shen *et al* 2015), Ethiopia (Mamuye *et al* 2018), Ghana (Zhou *et al* 2011, Dickinson *et al* 2015), sub-Saharan Africa (Tumwesige *et al* 2017) and Mexico, inter-country comparisons (Rose Eilenberg *et al* 2018, Johnson *et al* 2019) and comparative emissions reductions from improved cookstoves (Coffey *et al* 2017, Sonarkar and Chaurasia 2019). The reported $PM_{2.5}$ emissions factors (g MJ^{-1}) are highly variable (0.01–1.5), with lower emissions rates observed for electric and gas stoves, and nearly an order of magnitude higher for natural-draft and traditional cookstoves fueled by charcoal, wood and residue (0.06–1.8) (Arora and Jain 2016). Average emissions factors (g kg^{-1}) for primary organic aerosols, SO_2, NMVOC and NO_x have been compiled for mud stoves (5.7, 0.3, 2.7 and 1.0 respectively), conventional wood stoves (3.9, 0.2, 23.6 and 2.8), wood boilers (1.5, 0.3, 14 and 1.2) and coal-burning stoves (0.8, 0.2, 0.5 and 2.2) (Bond *et al* 2013). Average emission rates for outdoor cooking to model personal exposure were found to be substantially higher than for indoor cooking (Edwards *et al* 2017). Hu *et al* (2012) compiled a $PM_{2.5}$ emissions database for residential environments in the United States and identified lower emissions rates for microwave and oven use (0.64–0.7 mg h^{-1}) and 200%–300% higher for frying irrespective of oil type.

EDGAR4.3.2 does not account for ambient $PM_{2.5}$ and precursor emissions from household cooking. These contributions, which are specific to ambient air pollution, are instead reported by the GAINS emissions model based on the methodology by Chafe *et al* (2014). Household fuel use for cooking and heating is a significant contributor to anthropogenic $PM_{2.5}$ emissions, ranging from 20%–55% globally (Tao *et al* 2016, Pervez *et al* 2019). Here, we do not further compile a global emissions inventory for cooking. There are multiple opportunities to develop further research on the impacts of household cooking on $PM_{2.5}$-attributable premature deaths. Topics of interest to the broader conversation of the sustainability of food systems include (a) cookstove technologies (Arora and Jain 2016) and the impacts on $PM_{2.5}$-attributable health damage (Grieshop *et al* 2011), (b) socio-economic and air quality impacts of carbon-financing schemes and national-scale fuel

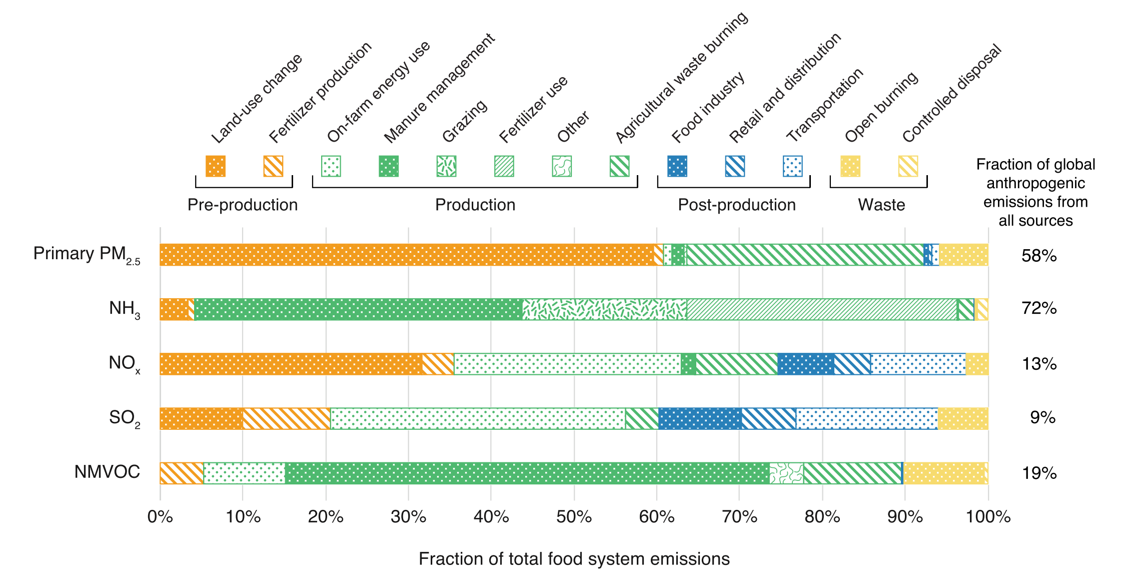

Figure 2. Global emissions of primary $PM_{2.5}$, NH_3, NO_x, SO_2 and NMVOC. Relative contributions by emissions sectors and grouped by stage in the food system: pre-production (orange), production (green), post-production (blue) and waste (yellow). Emissions from the consumption stage are not included. Percentage contribution of emissions from the global food system relative to total anthropogenic contributions are provided to the right of the bars.

interventions (Aung *et al* 2016, Kelp *et al* 2018), and (c) developing spatially explicit global emissions inventories of primary $PM_{2.5}$ and precursor emissions that are currently at national or regional scales (Chafe *et al* 2014) to enable spatially explicit assessment of $PM_{2.5}$-attributable health damage.

3.1.5. End-of-life disposal practices

Food loss and waste occur at all stages of the food supply chain (Parfitt *et al* 2010). Food losses of >40% are common in developing countries during the production and post-harvest stages, typically through agricultural waste burning due to inefficient technologies and poor infrastructure. Food waste of >40% at the retail and consumer stage is typical in developed countries and nearly equals the net food production in sub-Saharan Africa (Lipinski *et al* 2013). Household loss is the most important source of food waste with large per-capita variation, ranging from 6–11 kg yr^{-1} in sub-Saharan Africa to 95–115 kg yr^{-1} in Europe and North America (Lipinski *et al* 2013, Xue *et al* 2017). Waste can be disposed of through open burning or integrated within municipal solid waste systems in the form of controlled incineration, landfilling or composting. Thus, emissions of primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursors are a function of both waste quantity and mode of food disposal. In the only study identified, Grizzetti *et al* (2013) estimated that food waste management emitted 0.21 Tg NH_3 and 0.086 Tg NO_x for Europe. Few studies have provided emissions totals of trace gases and $PM_{2.5}$ from the open burning of domestic waste at national (India: Sharma *et al* 2019) and global scales (Wiedinmyer *et al* 2014). However, this analysis is not exclusive to food waste. Here, we derive emissions of primary $PM_{2.5}$, NH_3, NO_x, SO_2 and NMVOC for 43 countries from EXIOBASE3.3.17, and gap-fill data for other countries by combining national-scale solid waste data that are classified by waste management method (World Bank 2018a) and technology-specific emissions factors (EEA 2016).

3.2. Global emissions inventories

We present national-scale emissions inventories of primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursors from the global food system, reported for the year 2015 or the most recent year of available data, following the methods we describe at the sector-scale in section 3.1. Global emissions totals of primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursors are shown in table 2 with fractional sector contributions shown in figure 2. Figures 3 and 4 show national emissions totals and regional-scale fractional sector contributions, respectively. Overall, we find that the global food system is a major contributor to the anthropogenic emissions of primary $PM_{2.5}$ (58%), NH_3 (72%), SO_2 (9%), NO_x (13%) and NMVOC (19%) in comparison to total anthropogenic emissions reported in EDGAR4.3.2 (Crippa *et al* 2018). We estimate that the global food system emits 24 Tg primary $PM_{2.5}$, driven by fires for land-use change (60%), agricultural waste burning (28%) and open burning of food waste (6%). The dominant emission sources of primary $PM_{2.5}$ vary regionally. Land-use change was identified as the predominant source in South America, Africa and Asia, while crop management

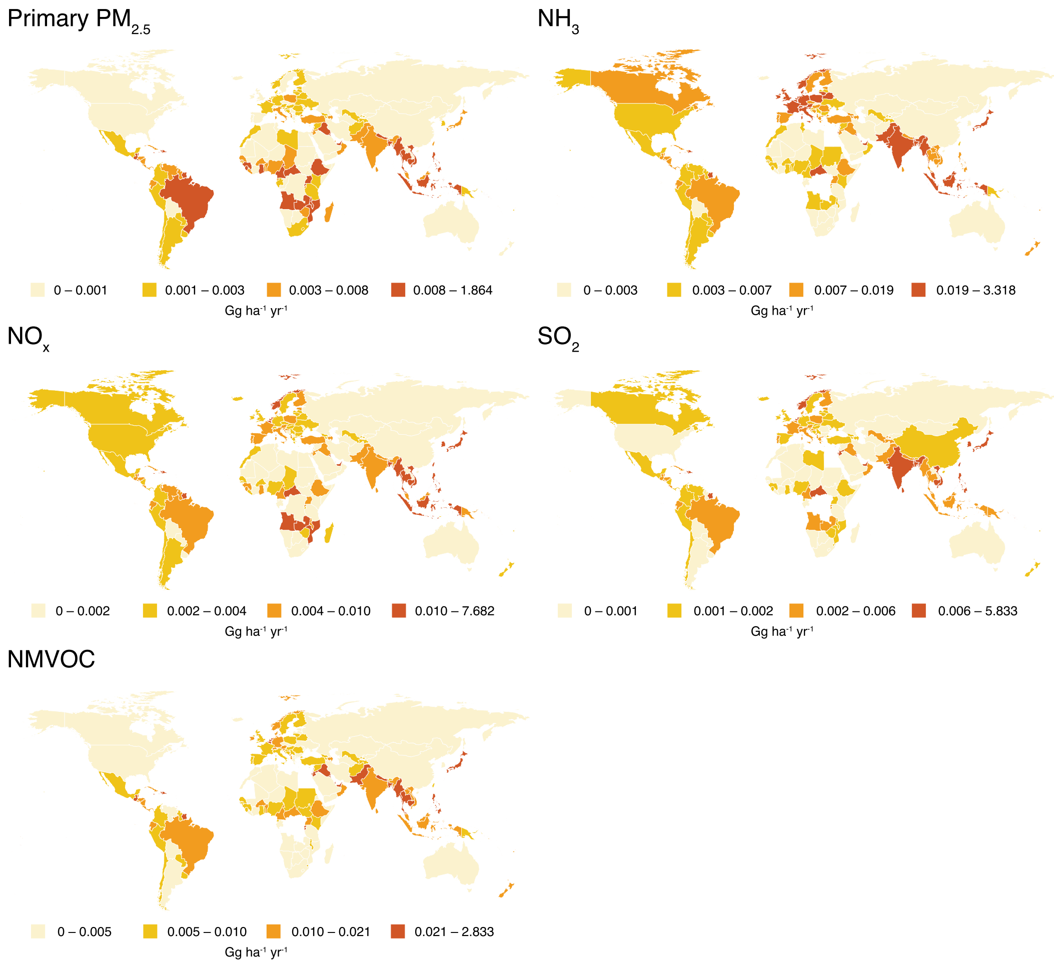

Figure 3. National-scale emissions from the global food system. Estimates are in Gg ha^{-1} yr^{-1} for: (a) primary $PM_{2.5}$, (b) NH_3, (c) NO_x, (d) SO_2 and (e) NMVOC, adjusted by national-scale agricultural land area (crop and pastureland). Estimates exclude emissions contributions from several sectors, including the consumption stage, that were identified as N/A in table 1.

and on-farm energy use dominate primary $PM_{2.5}$ emissions in North America and Eastern Europe, and China and Russia, respectively.

Global NH_3 emissions (42 Tg NH_3) largely result from livestock manure management (40%), grazing (20%) and synthetic fertilizer use (33%), with large variations in relative regional contributions. Fertilizer use is also a dominant contributor to NH_3 emissions in Asia and North America (40%–45%) in contrast to smaller contributions in Africa (<10%), where the slower adoption of nitrogen fertilizers and inefficient manure handling practices (Ndambi *et al* 2019) result in more than 50% contributions from livestock management. Of the 32 Tg NMVOC emitted from the food system, the dominant contributors included manure management (58%) and agricultural waste burning (12%). Smaller emissions totals were estimated for SO_2 (9 Tg) and NO_x (16 Tg), which are typically a result of combustion. SO_2 was linked to on-farm energy use (35%), post processing of food (30%) and open burning (15%), with similar trends for NO_x (35%, 30% and 6%, respectively).

4. $PM_{2.5}$ exposure and $PM_{2.5}$-attributable deaths from the food system

We describe the causal pathway of emission impacts on ambient $PM_{2.5}$ concentrations and $PM_{2.5}$-attributable premature deaths in section 4.1, summarize studies that report $PM_{2.5}$-attributable premature deaths from sectors within the global food system to develop an overall estimate of $PM_{2.5}$-attributable premature deaths in section 4.2, and discuss uncertainties in section 4.3.

4.1. Connecting the emissions—$PM_{2.5}$ exposure—premature mortality pathways

Ambient $PM_{2.5}$ concentrations are a result of precursor emissions, and are impacted by transport, chemistry and removal processes in the atmosphere. Of key importance to the discussion here are the emissions of NH_3, 72% of which is emitted from the food system (section 3.2). As the most dominant alkaline component in the atmosphere, NH_3 neutralizes acids formed from atmospheric

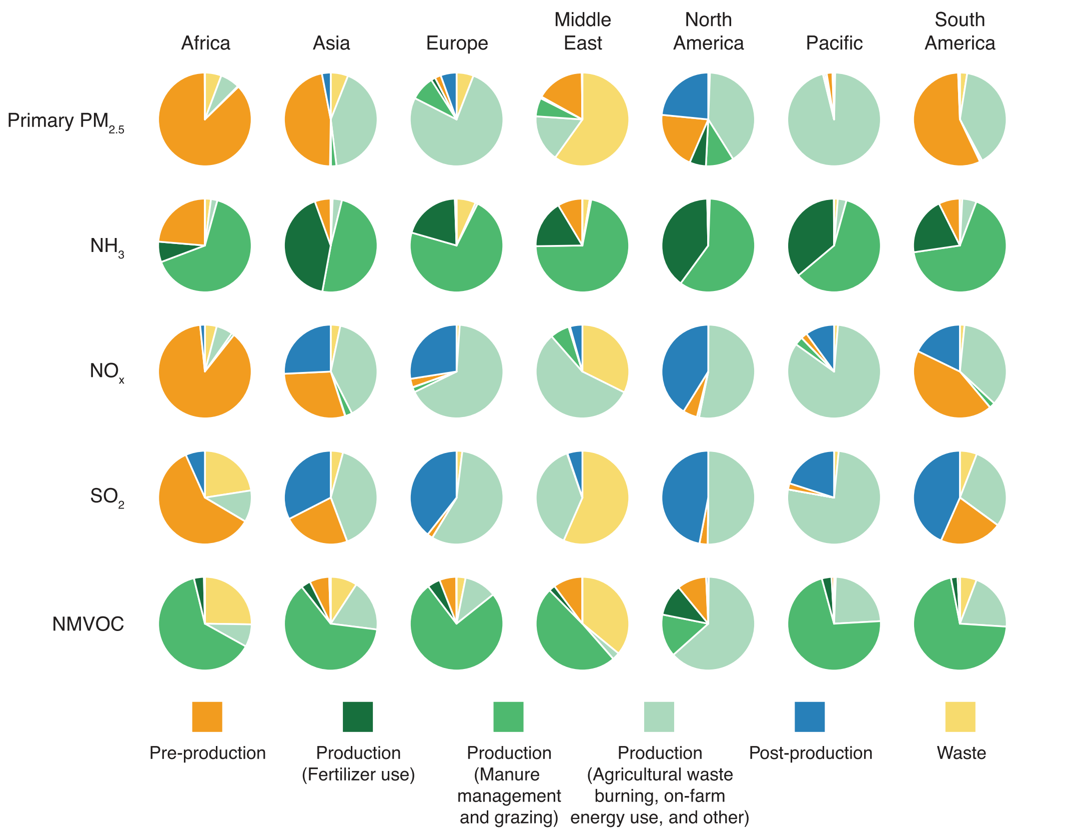

Figure 4. Region- and pollutant-specific emissions contributions from each stage over the life-cycle of the global food system. Emissions are grouped as follows: pre-production (orange) and production (green) that are further classified as fertilizer use, manure management and grazing, and aggregated emissions that include agricultural waste burning, on-farm energy use and other emissions (standing crop emissions and on-site handling of agricultural commodities), post-production (blue) and waste (yellow). Emissions from consumption are excluded in the analysis.

oxidation of precursor gases, such as SO_2 and NO_x, and organic acids to form $PM_{2.5}$ (Behera *et al* 2013). The concentration of ambient $PM_{2.5}$ and chemical partitioning, especially as PM-nitrate, is driven by the relative abundance of NH_3 and acids formed from precursors, such as SO_2 and NO_x in the atmosphere, thermodynamically driven by environmental conditions (Seinfeld and Pandis 2016), and thus can vary regionally and seasonally (Holt *et al* 2015, Weagle *et al* 2018). While $PM_{2.5}$ concentrations are more sensitive to the availability of NO_x in regions of high NH_3 concentrations (Langridge *et al* 2012), such as in India (Kharol *et al* 2013) and China (Lin *et al* 2020), NH_3 is still a major contributor to the overall $PM_{2.5}$ abundance (Warner *et al* 2017).

Despite air quality regulations in most regions of the world, global annual average ambient $PM_{2.5}$ concentrations are still 300% higher than the World Health Organization's recommended healthy air guideline of 10 μg m^{-3} (van Donkelaar *et al* 2016). 92% of the global population lives in countries in Africa, Southeast and East Asia, and the Middle East, where exposure exceeds 10 μg m^{-3} (Health Effects Institute 2019), and has significantly increased since 1998 (Li *et al* 2017b). Studies have identified chronic health risks even in regions of relatively clean air, where $PM_{2.5}$ exposure is lower (Shi *et al* 2016). Ambient $PM_{2.5}$ has been linked to reduced global life expectancy (Apte *et al* 2018), which results from a wide range of health impacts including ischemic heart disease, chronic obstructive pulmonary disease, cerebrovascular disease, lung cancer and non-communicable diseases including lower respiratory tract disease (West *et al* 2016, Pope *et al* 2019). The extent of health damage varies as a result of ambient $PM_{2.5}$ concentration, length of exposure and demographics especially for elderly and vulnerable populations as demonstrated by toxicological, short-term epidemiological and large-scale cohort studies (Shiraiwa *et al* 2017). The impact of $PM_{2.5}$ on mortality has been represented through integrated exposure-response (IER) functions that are developed based on a comprehensive body of cohort and population studies (Pope *et al* 2019).

To date, the analysis of emissions contributions to $PM_{2.5}$-attributable deaths has been limited to economic sectors, such as energy and transport (Lelieveld *et al* 2015, 2019, Silva *et al* 2016) with a few select studies examining agriculture (Bauer *et al* 2016, Pozzer *et al* 2017, Giannadaki *et al* 2018).

The typical approach is to sequentially: (a) generate emissions inputs to AQMs, (b) develop spatially resolved AQM predictions of ambient $PM_{2.5}$ concentrations, (c) estimate population-weighted $PM_{2.5}$ exposure and (d) finally scale $PM_{2.5}$ exposure using IER functions to estimate $PM_{2.5}$-attributable premature deaths. The AQM framework has been implemented using two approaches (Conibear *et al* 2018). In the 'zeroed out' approach, emissions from a sector of interest are zeroed or reduced and the resulting $PM_{2.5}$ deaths are attributed as source contributions. Alternatively, in the 'attribution' approach, sector-specific mortality is estimated in proportion to the fraction of the sectoral contribution to $PM_{2.5}$ concentrations, either by examining emissions contributions or in models that 'tag' $PM_{2.5}$ concentrations as marginal changes in emissions. Given that emissions totals in the two approaches differ and due to the non-linear emissions-$PM_{2.5}$ exposure responses, estimates of premature mortality can vary, especially in populated regions (Conibear *et al* 2018).

The analysis of health damage beyond broad economic sectors has been limited due to the large computational, data and resource requirements when using AQMs. Advances in high-performance computing, the use of alternative statistical approaches and the development of other models, such as reduced complexity models (RCMs) have enabled AQM assessments at high spatial resolution and for multiple scenarios. RCMs use simplified representations of atmospheric processes with variable grid sizes and leverage outputs from an existing AQM simulation to predict marginal changes in ambient $PM_{2.5}$ concentrations at high spatial resolution in response to marginal changes in precursor $PM_{2.5}$ emissions, with reduced computational times (Tessum *et al* 2017). RCMs have been widely implemented to study contributions of emissions to $PM_{2.5}$-attributable premature deaths from various economic sectors at high spatial scales (Goodkind *et al* 2019), delineate contributions at the emissions sector and pollutant scales to inform mitigation efforts (Thakrar *et al* 2020) and to monetize damage (Heo *et al* 2016, Gilmore *et al* 2019). These applications are currently limited to the United States, given the current constraints on the spatial formulations of RCMs.

4.2. Global food system linked to significant $PM_{2.5}$-attributable premature deaths

4.2.1. Summary of studies discussing the impact on ambient $PM_{2.5}$-attributable premature deaths

Given the large contribution of the global food system to primary $PM_{2.5}$ and NH_3 emissions, and the central role of NH_3 in the formation of secondary $PM_{2.5}$, we identify the lack of a system-scale analysis on the contribution of the global food system to $PM_{2.5}$-attributable premature deaths as a key literature gap. Here, we briefly discuss AQM studies that link emissions from different stages and emissions sectors within, but not exclusive to the food system to ambient $PM_{2.5}$-attributable premature deaths. Key findings are summarized in table 3, which highlights differences in the approaches used by the listed studies in terms of spatial extent of analysis, choice of emission inventories and AQM configurations, and the reporting of health damage.

Much of the focus on the impacts of the global food system on air quality has been on agricultural production. Emissions from agricultural production contribute to about 20% of $PM_{2.5}$ deaths worldwide (Lelieveld *et al* 2015), with larger contributions in China, the United States and Europe (45%–55%) and smaller contributions in India and Africa (5%–15%) (Bauer *et al* 2016, Guo *et al* 2018, Crippa *et al* 2019). A recent integrated health and environmental assessment by Malley *et al* (2021) linked global agricultural production to 537,000 $PM_{2.5}$-related deaths. Notably, a 100% reduction in these emissions would reduce 800 000 (95% confidence interval (95% CI): 420 000–980 000) global, annual $PM_{2.5}$-attributable premature deaths (Pozzer *et al* (2017). Achievable health benefits were identified to be the largest for Europe and North America (70%–80%) where significant reductions in NO_x and SO_2 emissions have already been achieved and $PM_{2.5}$ formation is highly sensitive to NH_3 emissions (Pozzer *et al* 2017, Giannadaki *et al* 2018). These responses were smaller in Asia (3%–25%), where $PM_{2.5}$ and PM-nitrate formation are sensitive to NO_x emissions (Bauer *et al* 2016, Giannadaki *et al* 2018). Goodkind *et al* (2019) estimated that NH_3 emissions from the United States agriculture resulted in 16 000 excess deaths and economic damage of 40 000 USD. However, large spatial variations ($\sim$500%) were reported for this marginal damage. The morbidity and mortality costs of 1 kg NH_3 emitted into the atmosphere showed large spatial variability (0.1–73 USD) and were valued to be much larger than the marginal damage that results from emissions of SO_x (0.2–12 USD) and NO_2 (0.02–2 USD) that have been the historic focus for $PM_{2.5}$ regulation (Muller and Mendelsohn 2010, Gilmore *et al* 2019). Overall, these findings suggest that air pollution regulations should consider regional-scale impacts of NH_3 emission reductions that are expected to provide the largest gains in Europe and North America, consistent with Pinder *et al* (2007) and Megaritis *et al* (2013).

Landscape fires (wildfires, prescribed burning and biomass burning but not limited to the global food system) have been linked to 330 000 (interquartile range: 260 000–600 000) excess deaths (Johnston *et al* 2012). Open biomass burning is a significant contributor to $PM_{2.5}$-excess deaths in China (1 million (95% CI: 840 000–1.3 million)), India (990 000 (95% CI: 660 000–1.35 million)) (Reddington *et al* 2019) and Africa (780 000 (95% CI: 760 000–800 000)) (Bauer *et al* 2019). However, these estimates are not delineated for contributions specific to the pre-production (land-use change),

Table 3. List of studies reporting $PM_{2.5}$-premature mortality estimates for sectors relevant but not exclusive to the global food system. Of the 320 studies, data sets and reports synthesized in this review, only 19 were identified that followed the causal chain of emissions-$PM_{2.5}$ exposure-premature mortality for sectors relevant to the global food system. These are listed in chronological order with a focus on the geographic scale of analysis, emissions sector and pollutants of interest, air quality model configuration including the name of the air quality model, emissions database, and horizontal grid resolution, and findings from the studies in terms of $PM_{2.5}$-excess deaths and economic damage. An important caveat is that the excess death estimates are based on different IER functions and are not adjusted for contributions ofor the global food system, given that the scope of all but three of the reported studies (marked with [a]) are not focused on the food system. Reported estimates could thus also include contributions resulting from land-use change for urbanization, mining, timber production and the production of non-food agricultural commodities, such as fiber and biofuels. Air quality and emissions model acronyms are defined alphabetically in the footnote.

					Air quality model (AQM) configuration				Premature mortality	
Study	Region	Sector	Pollutants	Approach	AQM	Emissions	Spatial resolution	Year	Excess deaths (95% CI)	Costs (USD)
Johnston *et al* (2012)	Global	Landscape fires	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Zero-out	GEOS-Chem	GFED	2° × 2.5°	1997–2006	340 000 (260 000–600 000)	
Paulot *et al* (2014)	United States	Food export[a]	NH_3, NO_x	Zero-out	GEOS-Chem	MASAGE	2° × 2.5°	2005	5100 (3400–6700)	36 billion
Lelieveld *et al* (2015)	Global	Production	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Zero-out	ECHAM	EDGAR	1.1° × 1.1°	2010	660 000	
Reddington *et al* (2015)	South America	Land-use change	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Zero-out	GLOMAP	MACCity + GFED	2.8° × 2.8°	2002–2011	2900 (1000–4700)	
Crippa *et al* (2016)	Southeast Asia	Land-use change	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Zero-out	WRF-Chem	EDGAR + FINN	10 km × 10 km	2015	12 000 (6100–17 000)	
Heo *et al* (2016)	United States	Production	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Attribution	EASIUR	NEI	36 km × 36 km	2005		110 billion
Kodros *et al* (2016)	Global	Domestic waste burning	Primary $PM_{2.5}$	Zero-out, attribution	GEOS-Chem	EDGAR + waste burning inventory	2° × 2.5°, 0.5° × 0.66° (Asia)	2010	191 000 (150 000–220 000)	
Lal *et al* (2016)	Agra, India	Domestic waste burning	Primary $PM_{2.5}$	Attribution	AERMOD	Solid waste, dung burning inventory	N/A	2015	700 (400–1050)	
Li *et al* (2017)	China	Agricultural waste burning	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Zero-out	Box model	Measurements	N/A	2012	7800 (3200–12 000)	

(Continued.)

Table 3. (Continued.)

Study	Region	Sector	Pollutants	Approach	Air quality model (AQM) configuration: AQM	Emissions	Spatial resolution	Year	Premature mortality: Excess deaths (95% CI)	Costs (USD)
Pozzer et al (2017)	North America, Europe, Asia	Production	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Zero-out	ECHAM	EDGAR	1.1° × 1.1°	2012	800 000 (420 000–990 000)	
Giannadaki et al (2018)	North America, Europe, Asia	Production	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Derived from Pozzer et al (2017)						407 billion
Gu et al (2018)	China	Crop production	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Zero-out	CMAQ	EDGAR-HTAP + FINN	27 km × 27 km	2010	130 000 (19 000–200 000)	4.5 billion
Crippa et al (2019)	Global	Livestock rearing, fertilizer use	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Attribution	TM5-FASST	HTAP	1° × 1°	2010	240 000	
Goodkind et al (2019)	United States	Production	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Attribution	InMAP	NEI	1 km–48 km grid size	2011	16 000	180 billion
Hill et al (2019)	United States	Corn cultivation	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Attribution	InMAP	NEI	1 km–48 km grid size	2014	4300	39 billion
Reddington et al (2019)	South and East Asia	Biomass burning	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Zero-out	WRF-Chem	EDGAR-HTAP	0.1° × 0.1°	2014	359 000 (228 000–513 000)	
Bauer et al (2019)	Africa	Biomass burning	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Zero-out	GISS	CMIP	2.0° × 2.5°	2007–2017	43 000 (42 000–44 000)	
Domingo et al (2021)	United States	Production[a]	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Attribution	InMAP	NEI	1km–48km grid size	2004	15,900	
Malley et al (2021)	Global	Production[a]	$PM_{2.5}$, NH_3, NO_x, SO_2, NMVOC	Attribution	GEOS-Chem	EDGAR	2.0° × 2.5°	2014–2018	522,500–536,800	

[a] Three studies listed here specifically account for contributions related to the global food system. Estimates for excess deaths for the other studies are not exclusive to the global food system.

Abbreviations: AERMOD (AMS/EPA Regulatory Model), CMAQ (Community Multiscale Air Quality), CMIP (Coupled Model Intercomparison Project), EASIUR (Estimating Air pollution Social Impact Using Regression), ECHAM (Climate Model), EDGAR (Emission Database for Global Atmospheric Research), FINN (Fire INventory from NCAR), HTAP (Hemispheric Transport of Air Pollution Inventory), GEOS-Chem (Goddard Earth Observing System—Chem), GFED (Global Fire Emissions Database), GISS (Goddard Institute for Space Studies General Circulation Model), GLOMAP (Global Model of Aerosol Processes), InMAP (Intervention Model for Air Pollution), MACCity (Monitoring Atmospheric Composition and Climate/CityZEN EU projects emissions data set), MASAGE (Magnitude And Seasonality of Agricultural Emissions), NEI (National Emissions Inventory), TM5-FASST (Tracer Model version 5—FAst Scenario Screening Tool), WRF-Chem (Weather Research and Forecasting model coupled with Chemistry) production (agricultural waste burning) and waste (open burning of food waste) stages in the global food system. Several studies have explored the impact of land-use change on emissions but are limited to a few regions. Ambient $PM_{2.5}$ spikes have been reported in Singapore in August–October (>5 μg m^{-3}) (Reddington *et al* 2014), and annually in Sumatra and Borneo (>120 μg m^{-3}) (Kiely *et al* 2019). The observed increases in ambient $PM_{2.5}$ were linked to oil palm expansion in peatlands and attributed to 12 000 excess annual deaths in Equatorial Asia (Crippa *et al* 2016). Similarly, deforestation fires in South America were linked to 1100–4700 premature deaths (Reddington *et al* 2015). Open burning of domestic waste, which includes commodities in addition to food, has been linked to 190 000 (95% CI: 150 000–270 000) global, annual $PM_{2.5}$-excess deaths (Kodros *et al* 2016), and accounts for 6% of the total $PM_{2.5}$ exposure in India (Rooney *et al* 2019) and 16% in China (Gu *et al* 2018).

In addition to these sectors, food export in the United States was linked to an average increase in $PM_{2.5}$ exposure by 0.36 μg m^{-3}, mostly attributed to NH_3 emissions, and resulted in 36 billion USD damage in 2006, which was equivalent to 50% of the food export value (Paulot and Jacob 2014). Hill *et al* (2019) estimated that maize cultivation, which accounts for 95% of all feed grain production in the United States, was linked to 4300 $PM_{2.5}$-attributed premature deaths. The resulting economic damage valued at 39 billion USD in 2017 exceeded the monetized damage as a result of GHG emissions, and in 40% of the maize growing states the combined $PM_{2.5}$ and GHG economic damage exceeded the market value, indicating large negative externality costs.

4.2.2. Estimate of ambient $PM_{2.5}$-attributable premature deaths resulting from the global food system

The studies summarized in section 4.2.1 collectively highlight the large $PM_{2.5}$-attributable premature deaths from sectors related to but not exclusive to the global food system. Here, we develop the first estimate, to our knowledge, of annual $PM_{2.5}$-attributable premature deaths from the global food system, as summarized in table 4. For the agricultural production stage, we adjust the median $PM_{2.5}$-attributable deaths from agricultural production reported by Pozzer *et al* (2017) with the fraction of global crop area devoted to food production to estimate 750 000 excess $PM_{2.5}$ deaths from food production. Similarly, for the waste stage, we adjust estimates by Kodros *et al* (2016) with the fraction of domestic waste that is composed of food (40%) to conservatively estimate 76 000 median excess $PM_{2.5}$ deaths. We derive estimates for food-demand-driven land-use change using findings for landscape fires by Johnston *et al* (2012), by first deducting $PM_{2.5}$-death contributions from fires resulting from open waste burning (Kodros *et al* 2016) and then further deducting contributions resulting from natural wildfires (23%) and non-agricultural commodity land-use change (30% of prescribed burning) (World Resources Institute 2014), resulting in an average estimate of 70 000 excess $PM_{2.5}$ deaths. We ensure no double counting of deaths occurred by conforming to the system boundaries that were used to describe stages in the food system and by excluding open waste burning, wildfires and non-food commodity land-use change from landscape fires.

Overall, by adding these estimates, we identify that 890 000 median excess deaths can be attributed to the global food system, 84% being a result of emissions from agricultural production. This order of magnitude estimate, developed based on studies with different approaches and IER functions (see table 4), is equivalent to 23% of the overall 3.9 million $PM_{2.5}$-attributable deaths in the Global Burden of Disease Study 2015 (IHME 2020), and is similar or higher in comparison to global contributions from natural sources (18%), power generation (14%) and transportation (5%) (Lelieveld *et al* 2015, Crippa *et al* 2019). Our estimates are higher than the $PM_{2.5}$-related deaths reported by Malley *et al* (2021), as ours accounts for life cycle emissions over the entire food system. Overall, we identify that our estimate of $PM_{2.5}$ deaths from the global food system is underestimated given the limited accounting of contributions from sectors including agrochemical production, post-processing, consumption and inherent uncertainties in the causal pathways of emissions to exposure estimates as identified in section 4.3.

Our analysis has also identified key research gaps: (a) to date, the focus has been on agricultural production, with few studies examining sectors from other stages in the food system, and at national or sub-national scales. There is a dearth of studies examining the impacts of food demand and agricultural production activities in highly populated regions in Africa, South America and Asia, where countries also have a high share of GDP (15%–58% in Africa and Asia) attributed to agriculture (World Bank 2018b). Given that a 10% increase in global NH_3 emissions could result in 22 000 additional excess deaths (Lee *et al* 2015), it is important to focus on these regions that are also expected to see increases in NH_3 emissions in the future. (b) It is important to identify the regional-scale efficacy of NH_3 emissions controls in regulating ambient $PM_{2.5}$ (Pinder *et al* 2007). Notably, Bauer *et al* (2016) demonstrated that emissions from increased food production could be managed without deteriorating future air quality, assuming emission controls on combustion sources of NO_x. Given the substantial uncertainties in the emission inventories from agriculture (Crippa *et al* 2019) (see section 4.3.1), the extent of the impacts of NH_3, NO_x and NMVOC emissions on ambient $PM_{2.5}$ at regional scales needs further investigation.

Table 4. Estimate of $PM_{2.5}$-attributable premature deaths from the global food system as developed in this review. Estimates are derived by using the median estimates reported by the listed studies and applying an allocation factor that identifies food versus non-food contributions from the emissions sector. These findings are largely underestimated given the paucity of emissions from the post-production and consumption stages, reported global underestimates in the magnitude of NH_3 emissions, the lack of sector-scale analysis of $PM_{2.5}$-attributable deaths in several regions including South America and Africa, and different methods and uncertainty in IER functions.

Stage	Sector	Region	Study	IER functions	Non-adjusted $PM_{2.5}$-deaths	Allocation for the food v/s non-food contributions	Adjusted $PM_{2.5}$-deaths from the global food system
Pre-production	Land-use	Global	Johnston *et al* (2012)	American Cancer Society study by Pope *et al* (1995)	330 000	1. Reduce contributions from waste burning (Kodros *et al* 2016) 2. Reduce contributions from wildfires (23%) and non-agriculture commodity driven land-use change (30%) (World Resources Institute 2014)	70 000
	Fertilizer production	N/A					
Production	On-farm energy use Livestock rearing Fertilizer use Agricultural waste burning Other emissions	Europe, Asia, North America	Pozzer *et al* (2017)	Burnett *et al* (2014) as reported in GBD 2010	800 000	93% global crop area diverted for food production (FAO 2020)	750 000
Post-production	Food industry Transportation Retail and distribution	N/A					
Consumption	Commercial cooking Residential cooking	N/A					
Waste	Open burning	Global	Kodros *et al* (2016)	Burnett *et al* (2014) as reported in GBD 2010	190 000	40% domestic waste composed of food (World Bank Group 2018)	76 000
	Controlled disposal	N/A					
			Median $PM_{2.5}$ attributable deaths from the global food system:				890 000

4.2.3. Household cooking impacts on ambient $PM_{2.5}$ pollution burden

In this review, we thus far focus on ambient $PM_{2.5}$. However, it is worthwhile briefly discussing the expansive literature that examines household cooking impacts on both ambient and indoor $PM_{2.5}$ exposure. The use of solid fuels, such as coal, wood, crop residues and animal dung can elevate household $PM_{2.5}$ concentrations by 110–850 μg m^{-3} higher in comparison to the use of gas or electricity (Shupler *et al* 2018). As a result, 2.8 million premature deaths in 2015 have been linked to exposure to household $PM_{2.5}$ (Kodros *et al* 2018), as well as non-fatal cardiovascular and respiratory conditions (Hystad *et al* 2019). Recent studies have estimated that 12% of global population-weighted average ambient $PM_{2.5}$ exposure is attributed to household solid cooking fuels (Smith *et al* 2014), with regional contributions ~10% in East Asia including China, and higher contributions in India (26%) and sub-Saharan Africa (37%) (Chafe *et al* 2014, Smith *et al* 2014). While there are challenges in separating the indoor versus outdoor contributions of emissions to $PM_{2.5}$, household cooking has been attributed to 450 000 excess deaths annually (Chafe *et al* 2014). In China alone, household energy use for cooking was attributed to 182 000–260 000 excess deaths (Archer-Nicholls *et al* 2016, Zhao *et al* 2018). In India, residential energy use was linked to 34% ambient $PM_{2.5}$ exposure (Rooney *et al* 2019) and contributed to 512 000 excess deaths (Conibear *et al* 2018). However, contributions from cooking were not delineated. Given that emissions from residential energy use for heating and cooking dominate the contribution to $PM_{2.5}$ exposure in India, China and sub-Saharan Africa (Butt *et al* 2016), we recommend that further research be directed to understanding the mitigation potential of cleaner fuels and technologies, especially at spatially explicit scales in these regions (Kuhn *et al* 2016).

4.3. Sources of uncertainties

4.3.1. Characterization of NH_3 emissions and linkages to $PM_{2.5}$ formation

Estimates of $PM_{2.5}$-attributed deaths can vary around $\pm$1 million globally due to uncertainties in emissions inventories alone (Crippa *et al* 2019). Reducing uncertainties in NH_3 emissions inventories is critical for more accurate estimates of ambient $PM_{2.5}$ exposure. While NH_3 emissions from EDGAR are within a factor of three in comparison to satellite-derived emissions fluxes, at least 67% of the sources were underestimated by one order of magnitude or more (van Damme *et al* 2018). Studies report large underestimates in total NH_3 emissions over agricultural areas that are as high as 40% in China (Zhang *et al* 2017b, 2017c) and 200%–450% across the United States (Heald *et al* 2012, Battye *et al* 2016, Bray *et al* 2017). These underestimates result from limited representations of the total magnitude and spatial and temporal distribution of NH_3 emissions from manure management and fertilizer use (Appel *et al* 2011, Paulot *et al* 2014, Hendriks *et al* 2016, Balasubramanian *et al* 2020, Ge *et al* 2020), and are subsequently linked to large biases in the predictions of ambient $PM_{2.5}$ concentrations especially for PM-NO_3 formation (Punger and West 2013, Paolella *et al* 2018).

A wide range of approaches have been adopted to reduce uncertainties in NH_3 emission inventories, including the use of inverse models that use observation data to constrain seasonality in NH_3 emissions (Paulot *et al* 2014, Zhu *et al* 2015a), process models that capture interactions between crop, soil and weather to predict NH_3 emissions at site and regional scales (Cooter *et al* 2012, Balasubramanian *et al* 2017, Xu *et al* 2019), and meta-analysis of field measurements (Pan *et al* 2016). In addition, continued advances in capturing emissions from sources, such as small fires, domestic burning and peatland fires through products like GFED4, further research delineating emissions contributions from agriculture-driven land-use change, and estimating emissions from food waste will help improve our understanding of the $PM_{2.5}$ pollution burden from the food system.

4.3.2. Resolving uncertainties in AQMs and choice of model parametrization

Uncertainties in air quality modeling that result from model formulation and model parametrization can introduce uncertainties in estimates of $PM_{2.5}$ premature mortality. However, these concerns are not specific to the analysis of the global food system. It is infeasible to examine the entire extent of formulations and parametrizations to quantity embedded uncertainties (Solazzo *et al* 2017). However, marginal $PM_{2.5}$ responses to additional emissions have smaller biases than $PM_{2.5}$ predictions in response to the absolute magnitude of emissions (Hogrefe *et al* 2008). An important aspect of CTMs (Chemical Transport Models) is the choice of spatial resolution (Reddington *et al* 2014). Kushta *et al* (2018) found that premature mortality estimates varied by less than 3% when using a coarser CTM resolution (>100 km) in comparison to a finer population-scale spatial resolution (<20 km). Similarly, a fine spatial scale analysis (4–36 km grid dimensions) over the United States constrained $PM_{2.5}$-attributable mortality to $\pm$10% (Thompson *et al* 2014). In contrast, Punger and West (2013) found higher differences (~30%) when scaling $PM_{2.5}$ exposure from a coarser scale of global models (>250 km) to 12 km $\times$ 12 km, with similar differences reported (27%) when switching from coarsest (~69 km) to finest (~5.9 km) grids for the United States using an RCM (Paolella *et al* 2018). Despite similar methodologies, (Kodros *et al* 2016) estimates of total annual, global $PM_{2.5}$-deaths were 13% lower in comparison to (Lelieveld et al 2015) as a result of coarse AQM configuration. Thus, rigorous $PM_{2.5}$ evaluation on a case-by-case basis is recommended in comparison to standard model performance benchmarks (Emery et al 2017) before further evaluation for health assessment. Further model improvements should also focus on reducing uncertainties in capturing $PM_{2.5}$ formation that is non-linear in response to NH_3 emissions, as well as representations of secondary organic aerosol formation (Fuzzi et al 2015).

4.3.3. Exploring IER functions to link $PM_{2.5}$ exposure to $PM_{2.5}$-attributable deaths

Many studies use log-linear IER functions, wherein a given reduction in $PM_{2.5}$ concentrations would yield the same gains in health benefits (Marshall et al 2015). Supralinear IER functions, however, better represent premature mortality outcomes as a function of $PM_{2.5}$ exposure (Goodkind et al 2014), thereby resulting in greater benefits at lower $PM_{2.5}$ concentrations (Marshall et al 2015). The IER responses at relatively high levels of $PM_{2.5}$ represent a source of uncertainty as they are derived based on studies for North America and Europe where the annual average $PM_{2.5}$ exposure is less than 30 μg m^{-3} and have different baseline health conditions compared to several parts of the world. Recent studies now account for impacts from regions with high $PM_{2.5}$ exposure, such as in China (Shiraiwa et al 2017, Yin et al 2017). Burnett et al (2018) estimated that global $PM_{2.5}$ excess deaths could be as high as 8.9 million if the IER functions were derived using cohort studies covering the entire range of global $PM_{2.5}$ exposure. Goodkind et al (2019) estimated that varying the IER functions resulted in a 21% difference in mortality estimates for the United States. In addition, $PM_{2.5}$-attributable damage should consider both chronic and sporadic exposure for episodic emissions sectors, such as fires (Johnston et al 2012), and account for toxicity resulting from $PM_{2.5}$ components (Shaffer et al 2019). Lelieveld et al (2015) identified that when carbonaceous $PM_{2.5}$ was assumed to be more toxic than inorganic $PM_{2.5}$, the resulting mortality attributed to agricultural emissions reduced from 20% to 7%. However, similar analysis for fires from land-use change and waste combustion could result in large estimates of $PM_{2.5}$-attributable deaths. The responses of human health to $PM_{2.5}$ toxicity, especially to components that are carcinogens or allergens, and the synergistic interactions resulting from organic fractions remain active areas of research (West et al 2016, Shiraiwa et al 2017, Landrigan et al 2018, Bates et al 2019).

5. Opportunities for $PM_{2.5}$ mitigation and policy implications

If the current shifts in diets, affluence and population growth trends continue, agricultural production will need to increase by 60%–100% by 2050 to meet future food demand (Tilman et al 2011, Tilman and Clark 2014, FAO 2018). This demand is expected to increase the environmental burden through increases in GHG emissions by 87%, cropland demand by 67%, water withdrawals by 65% and nitrogen fertilizer inputs by 860% (Springmann et al 2018a), but the potential increase in $PM_{2.5}$-health damage is less well understood. Likewise, few studies have evaluated the emissions reduction potential of farm-scale interventions (Kupper et al 2015, Xu et al 2017, Guthrie et al 2018), and the impact of reductions of emissions from agricultural production on ambient $PM_{2.5}$ (Bauer et al 2016, Pozzer et al 2017) and $PM_{2.5}$-attributable premature deaths (Giannadaki et al 2018, Crippa et al 2019). In addition, there is concern with regard to the inequity in air pollution exposure impacts that occur from demographic differences in emissions attributed to the consumption of goods and spatially distant impacts of emissions on $PM_{2.5}$ exposure (Tessum et al 2019). These environmental justice implications are of particular interest in the global food system, dependent on where and how food is cultivated, and further exacerbated by socioeconomic differences in access to adequate and nutrient-rich foods. Reducing these environmental and health impacts will require a 'third Green Revolution' that focuses on the adoption of sustainable diets, improved agricultural practices and the implementation of regulatory instruments (FAO 2018). Here, we briefly identify instruments that have been proposed for the global food system to meet climate targets (Bryngelsson et al 2016, Wollenberg et al 2016, Niles et al 2018) that have potential co-benefits in minimizing ambient $PM_{2.5}$ health burden, both within and beyond the farm gate (Kanter et al 2020).

5.1. 'Eating enough' and 'eating right'

Individual dietary demands play a key role in determining the impacts of the global food system (Kearney 2010). Since 1961, global food consumption has increased by 400 kcal d^{-1}, with the largest increases in South Asia (>50%) and Latin America (>30%) (FAO 2020a), and is projected to further increase by 25% by 2030 (FAO 2017). Despite improvements in food supply equity in the past century, a triple burden of malnutrition exists in the form of undernutrition (690 million), obesity (1.9 billion adults and 42 million children) and micronutrient deficiencies (2 billion) (WHO 2018, FAO 2020c). Agricultural production will need to increase to meet global food demand, while also accounting for shifts towards animal-based foods that are expected to increase by nearly 30% for meat and 20%–58% for eggs and dairy by 2050 (Clark et al 2018). These increases will likely be accompanied by increases in $PM_{2.5}$ and precursor emissions, especially in Asia and Africa, which face the largest increases in food demand (Godfray et al 2010).

The paradigm of 'eating right' and 'eating enough' could be the key to mitigating environmental damage, including air quality impacts. Consuming only the required calories that meet individual metabolic and nutritional demands could improve health and climate outcomes (Niles *et al* 2018). Producing crops only for human consumption (i.e. plant-based foods) can increase caloric availability by 70% (Cassidy *et al* 2013), thus meeting not just current, but future global food demand (Berners-Lee *et al* 2018). Plant-based foods have been identified to have lower environmental impacts per serving in comparison to animal-based foods, especially ruminant meats from cattle, sheep and goats that have larger contributions compared to pork, poultry, eggs and dairy (Clark *et al* 2018, Poore and Nemecek 2018, Willett *et al* 2019). Springmann *et al* (2016) estimated that a complete shift to vegetarian diets and increasing vegetables, fruits, lentils and grain consumption by >50% would reduce GHG emissions (3–11 Gt yr^{-1}) by 2050. An examination of emissions factors for animal type-specific manure management suggests that similar trends could be expected for $PM_{2.5}$ pollution burden. However, the extent of the impact of shifts in diets needs further investigation. Reducing dependencies on animal-based foods could maximize both health and environmental benefits (Clark and Tilman 2017, Godfray *et al* 2018). It is thus imperative to establish spatially explicit impacts of the global food system on $PM_{2.5}$ health impacts, with a focus on NH_3 emissions, as well as emissions resulting from land-use change.

5.2. Managing food waste

Globally, food waste tripled between 1960 and 2011 (FAO 2011, Porter *et al* 2016) and is a contributor to emissions of primary $PM_{2.5}$, NMVOC and SO_2 as a result of disposal practices. Reducing consumer food waste by 50%, either by individual choice or through supply chain interventions, could result in a 10% reduction in fertilizer and land use while improving food security through 1300 trillion kcal savings yr^{-1} by 2050 (Clark *et al* 2018). Developing policies and infrastructure to shift the open burning of waste to controlled disposal, possibly coupled with energy recovery, could provide benefits in reducing $PM_{2.5}$ pollution (Coventry *et al* 2016). However, tradeoffs in the form of increases in NH_3 emissions that result from organic waste decomposition should be carefully evaluated (Wang and Zeng 2018).

5.3. Farm-scale interventions

The demand for food is expected to increase substantially, along with subsequent emissions, especially in Asia (by 40%) and Africa (by 47%) by 2050. While the agricultural contributions to ambient $PM_{2.5}$ in these regions are small (3%–9%), in comparison to residential (27%–45%) and power generation (17%–26%) (Crippa *et al* 2019), even a 50% reduction in agricultural production emissions could reduce up to 130 000 $PM_{2.5}$-attributable premature deaths (Pozzer *et al* 2017). It is thus imperative to balance the need for food security with resulting health impacts, to reduce the expected large externalities and economic losses, through improvements in agricultural productivity as well as farm-scale mitigation strategies.

An increase in crop yields and reductions in farm-scale inefficiencies that are prevalent in lower-income countries in Sub-Saharan Africa, Mexico, India and Southeast Asia could reduce nitrogen and energy inputs to meet future food demand (Mueller *et al* 2012). Of high priority is the reduction of yield gaps (Lobell *et al* 2009, van Ittersum *et al* 2013) that are prevalent in 43 countries where crop yields are less than a third of their potential (Clark *et al* 2018). Suggested strategies include improving access to agricultural inputs, such as fertilizers, seeds and pesticides, especially in sub-Saharan Africa (Pradhan *et al* 2015). However, tradeoffs in increased yields and economic gains must be carefully weighed against increases in GHG, NH_3 and $PM_{2.5}$ emissions, and other environmental concerns. The impacts of alternative agricultural practices on air quality should be further evaluated, by exploring crop diversification through leguminous crops, intercropping and crop rotation (Garrity *et al* 2010, Ponisio *et al* 2015, Hunt *et al* 2019, 2020), organic cultivation (De Ponti *et al* 2012), the use of companion crops and exploiting the agronomic potential of natural NMVOC (Brilli *et al* 2019), and integrated pest management practices (Khan *et al* 2014, Hunt *et al* 2017).

Global NH_3 emissions are expected to increase as a result of livestock production by 2050 (Bouwman *et al* 2013). Reducing NH_3 emissions in current food systems will not only benefit air quality, but reduce economic losses for farmers that result from nitrogen volatilization (Good and Beatty 2011). Guthrie *et al* (2018) compiled a comprehensive list of mitigation interventions for Europe that include improvements in livestock management by modifying animal feed (NH_3 reductions of 30%–45%) and increasing grazing time (<50%), structural interventions, such as redesigning animal housing and manure storage (>80%), adding control technologies, such as wet scrubbers (25%–65%), and modifying crop cultivation practices, including changes in nitrogen fertilizer type from urea to other forms of nitrogen, the use of fertilizer inhibitors and changing fertilizer application timing, loading rate and application mode (20%–70%). Similar assessments for emissions reduction potential and costs have also been reported by other studies in Europe (Kilmont and Winiwarter 2015) and the United States (Pinder *et al* 2007). The current suite of engineering solutions and best management practices could result in a 30% reduction in livestock-NH_3 emissions and 20% in fertilizer-NH_3 emissions (total 0.7 Tg yr^{-1}) for the United

States alone (US EPA 2011). However, further region-specific studies are required.

5.4. Technological interventions

Technological solutions can reduce $PM_{2.5}$ pollution and have climate co-benefits within and beyond the post-production stage. Proposed interventions include improvements in energy efficiency by 20%–50% in food processing, distribution and retail, through correct specification and equipment use, cold chain-based food supplies, modal shifts in food transportation (Wakeland *et al* 2012, Pelletier 2015, Niles *et al* 2018, and new packaging technologies (Heller *et al* 2019). Reducing household and ambient $PM_{2.5}$ exposure in regions that are reliant on solid fuel use for cooking have been identified as an important area of research. Ongoing efforts have focused on reducing disparity through the widespread adoption of cleaner fuels and cleaner technologies by the World Health Organization (WHO 2016) and by governments in India, China and across Africa (Aung *et al* 2016, Anenberg *et al* 2017). Recommended guidelines include switching from dirty household fuels including kerosene and coal to cleaner fuels higher on the energy ladder, such as LPG, ethanol, biogas and electricity, and introducing cheaper and cleaner cookstoves as promoted by the Global Alliance for Clean Cookstoves (Lewis and Pattanayak 2012, Anenberg *et al* 2013, Pachauri *et al* 2013) to reduce emissions of primary $PM_{2.5}$, NO_x and SO_2, and the resulting health burden.

5.5. Regulatory instruments

Two regulatory instruments are of interest to minimize the impacts of food system emissions on $PM_{2.5}$ deaths. First, unlike economic sectors such as electricity generation and transportation, not all emissions sources within agriculture have been considered for emissions regulation in most parts of the world. While agriculture is not explicitly excluded from regulations in the United States, emissions regulation on primary $PM_{2.5}$ or secondary $PM_{2.5}$ precursors from farms is required only in non-attainment areas (US EPA 2017). For example, state regulations are imposed in California on crop growers, poultry, dairy and cattle farms, and agri-businesses (CARB 2019). However, on-farm emissions typically do not exceed the specified threshold and are thus exempt from most regulatory programs in the United States (US EPA 2017). Second, in the United States, the Clean Air Act regulations consider six criteria, air pollutants including NO_x, SO_2 and primary $PM_{2.5}$; NH_3 is currently not regulated. Given the large body of evidence identifying the key role NH_3 plays in regulating atmospheric chemistry, the US EPA Science Advisory Board has recommended regulatory approaches to treat NH_3 as a harmful $PM_{2.5}$ precursor (US EPA 2011). Such a regulatory approach should be considered worldwide.

Programs to study nitrogen management strategies and impacts on the environment and agricultural productivity have been adopted in Europe. The Convention on Long-Range Transboundary Air Pollution and the Gothenburg Protocol that have set targets to reduce SO_2, NO_x, NMVOC and NH_3 by 63%, 41%, 40% and 17%, respectively, by 2010 compared to 1990 to reduce acidification and surface water eutrophication, as well as preventing 48 000 excess deaths from $PM_{2.5}$ and ozone exposure (UNECE 2017). NH_3 emissions have already been reduced by 24% between 1990 and 2008, facilitated through multiple programs ranging from the adoption of alternative fertilizer types in Germany to providing financial incentives for improved nitrogen use in the Netherlands (EEA 2015). The United Kingdom also recently announced a plan to reduce NH_3 emissions by 15% by 2030 (Plautz 2018), demonstrating increased attention to cost-effective $PM_{2.5}$ abatement through the regulation of NH_3. A multi-faceted regulation policy should be considered at the national scale to optimize the economic and environmental costs of farm-scale practices and alternative approaches.

5.6. Legislation, environmental and health protections

Legislation and environmental protections are important drivers of reducing the demands of agriculture on land-use change (Nolte *et al* 2017, Seymour and Harris 2019). However, these strategies rarely account for the nature of agricultural commodities and consumption patterns (Henders *et al* 2018). Deforestation rates decreased between 2004 and 2014 in Brazil following the establishment of conservation zones (Anderson *et al* 2016). However, recent increases since 2017 (Amigo 2020) are a result of non-compliance with conservation agreements to meet the increased demand for soy, cattle and timber (Carvalho *et al* 2019) and the impact of export-driven trade demands (Tester 2020). In Indonesia, national moratoriums as well as pledges from corporations to increase sustainable products in their supply chains have helped reduce conversions of primary forests and peatland for industrial palm production (Carlson *et al* 2018, Gaveau *et al* 2019). However, the impacts of such policies on air pollution and health exposure are relatively unexplored and limited to a few studies (Marlier *et al* 2015, 2019).

In addition, health protection policies to promote healthier diets through dietary guidelines and legislation could offer co-benefits to both health and the environment (Clark *et al* 2018). The implications of these policies on GHG emissions have been the subject of recent inquiry, with examples

Table 5. Key findings from the system-scale analysis of excess deaths occurring from exposure to $PM_{2.5}$ from the global food system.

Findings	Implications
The global food system is a significant contributor to total anthropogenic emissions of primary $PM_{2.5}$ (58%), NH_3 (72%), NO_x (13%), SO_2 (9%), and NMVOC (19%).	1. Important sources include land-use change, livestock and crop production, and agricultural waste burning. 2. Sectoral emission contributions vary by region. Agricultural production emissions dominate contributions in North America and Europe, while land-use change, manure management and agricultural waste burning emissions dominate contributions in Asia, Africa and South America.
Emissions from the global food system are linked to at least 890 000 annual $PM_{2.5}$-attributable deaths, which is equivalent to 23% of $PM_{2.5}$-deaths reported in the Global Burden of Disease Study 2015.	1. 84% of estimated $PM_{2.5}$- attributable deaths from the global food system are a result of agricultural production. 2. Estimates of excess deaths are likely underestimated given the paucity in emissions data from the post-production and consumption stages, underestimated global NH_3 emissions, and lack of sector-scale analysis of $PM_{2.5}$-attributable deaths in South America and Africa.
There are uncertainties in establishing the impact of emissions from the global food system on $PM_{2.5}$-attributable deaths.	1. It is important to fill critical information gaps in emissions inventories and reduce existing uncertainties (30%–50%) in primary $PM_{2.5}$, NH_3 and NO_x emissions from the food system. 2. Developing emissions inventories for land-use change and food waste disposal are recommended as high priority research focus, with spatially explicit analysis for South America, Asia and Africa. 3. Continued air quality model development and evaluation is required to reduce current biases of ∼30% in ambient $PM_{2.5}$ predictions. 4. Using a homogenous set of IER functions across studies that also account for toxicity of $PM_{2.5}$ components will reduce existing biases in estimates of excess deaths.
There are likely many cost-effective opportunities to mitigate $PM_{2.5}$ pollution by reducing emissions from the global food system.	1. Air pollution regulations should include $NH_{3,}$ given the key role in formation of ambient $PM_{2.5}$. 2. Shifts in dietary patterns and regulatory instruments can mitigate $PM_{2.5}$ emissions from land-use change. 3. Reduced dependence on fertilizers, improvements in crop yields and better technological practices can help reduce NH_3 emissions from livestock and crop production. 4. Emissions of NO_x and SO_2 from the food industry including retail, distribution and transportation could be mitigated through clean and energy-efficient technologies. 5. Reduction in food waste and shifts to controlled waste disposal is key to mitigating emissions from food waste.

including the evaluation of national dietary recommendations (Behrens *et al* 2017), national-scale strategies to reduce dependencies on animal-based foods (Springmann *et al* 2018b), and expanding sustainability metrics to also account for macro and micronutrient delivery from food production (DeFries *et al* 2015, de Ruiter *et al* 2018). In addition, managing food pricing has been recommended as a tool to reduce GHG emissions, namely through GHG taxes (Springmann *et al* 2017), taxes on less healthy foods, such as refined sugar (Briggs *et al* 2016), and through subsidies and tax revenues (Hadjikakou 2017) albeit with concerns about disproportionate effects on those of lower socioeconomic status. $PM_{2.5}$-related pollutant emissions could also be achieved through programs that target reductions in waste (Porter *et al* 2016), food portioning (Story *et al* 2008) and food labeling akin to calorie labeling (Upham *et al* 2011, Leach *et al* 2016). Expanding environmental impact assessment to include impacts on $PM_{2.5}$ pollution burden deserves further study, given the downstream impacts of such policies on shifts in diet, food production, processing and waste disposal. Finally, it is essential to consider the environmental justice implications of food and agricultural systems for the world at large.

6. Highlights and research needs

The recent growth in understanding the global food system and its complex interplay with energy, material, water and land use has expanded our understanding of the large burden it places on the environment (Springmann *et al* 2018a). Indeed, a comprehensive accounting of food sustainability requires further consideration of major diets and food commodities, the processes that drive the food system and expanding the suite of environmental impacts (Halpern et al 2019). Our review adds to the conversation about global food system sustainability by identifying the large health burden resulting from exposure to ambient $PM_{2.5}$. Here, we show that $PM_{2.5}$-related emissions from the global food system are linked to 890 000 $PM_{2.5}$-attributable premature deaths annually, which is equivalent to 23% of the 3.9 million ambient $PM_{2.5}$-attributable deaths reported in the Global Burden of Disease Study 2015. These findings are, however, underestimated, given the paucity of emissions from food post-production and consumption stages, the overall global underestimate in emissions of NH_3 and the lack of $PM_{2.5}$ exposure impact studies for several emissions sectors and in regions including South America and Africa. A summary of our key findings is listed in table 5.

Additional empirical research is needed to reduce uncertainties in the characterization of emissions across multiple spatial and temporal scales to support air quality forecasting, and with a focus on expected future trends in production, consumption and food losses in low- and middle-income countries. Research opportunities abound in identifying improvements in energy and resource use in the food industry, retail and distribution, and transportation. Furthermore, systematic and region-scale efforts, especially in Asia, Africa and South America, are required to establish how the identified emissions mitigation strategies could deliver cost-effective reductions in ambient $PM_{2.5}$ concentrations and $PM_{2.5}$-attributable premature deaths. With diets shifting towards animal-based and more processed foods, and increases in global caloric consumption, additional environmental and health burdens resulting from degrading air quality are expected. However, by considering variability in regional shifts in future food demand and production, strategies that encompass a wide range of regulatory, technological and educational tools that encourage health and environmentally conscious diets can be implemented to sustainably manage these increases with minimal impacts on air pollution. Given the recent interest in food system research in the context of climate and other environmental impacts, we argue that these studies should further account for damage from $PM_{2.5}$ pollution as a key indicator of both human and environmental health.

Data availability statement

All data that support the findings of this study are included within the article (and any supplementary files).

Acknowledgments

This publication was developed as part of the Center for Air, Climate, and Energy Solutions (CACES), which was supported under Assistance Agreement no. R835873 awarded by the U.S. Environmental Protection Agency (EPA). It has not been formally reviewed by the EPA. The views expressed in this document are solely those of the authors and do not necessarily reflect those of the EPA. The EPA does not endorse any products or commercial services mentioned in this publication. This research was also supported by the U.S. Department of Agriculture (MIN-12-083).

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