Hidden Cost of U.S. Agricultural Exports: Particulate Matter from Ammonia Emissions

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ABSTRACT: We use a model of agricultural sources of ammonia (NH₃) coupled to a chemical transport model to estimate the impact of U.S. food export on particulate matter concentrations (PM₂.₅). We find that food export accounts for 11% of total U.S. NH₃ emissions (13% of agricultural emissions) and that it increases the population-weighted exposure of the U.S. population to PM₂.₅ by 0.36 μg m⁻³ on average. Our estimate is sensitive to the proper representation of the impact of NH₃ on ammonium nitrate, which reflects the interplay between agricultural (NH₃) and combustion emissions (NO, SO₂). Eliminating NH₃ emissions from food export would achieve greater health benefits than the reduction of the National Ambient Air Quality Standards for PM₂.₅ from 15 to 12 μg m⁻³. Valuation of the increased premature mortality associated with PM₂.₅ from food export (36 billion US$ (2006) per year) amounts to 50% of the gross food export value. Livestock operations in densely populated areas have particularly large health costs. Decreasing SO₂ and NOₓ emissions will indirectly reduce health impact of food export as an ancillary benefit.

INTRODUCTION

Increasing nitrogen inputs from fertilizer application have contributed to greater agricultural outputs in the last 50 years.¹ They have also resulted in the release of nitrogen to the environment, in particular through emissions of ammonia (NH₃) to the atmosphere. In the U.S., Houlton et al.² estimated that ∼25% of the nitrogen used as fertilizer is lost to the atmosphere as ammonia (NH₃), costing farmers ∼6 billion US$ a⁻¹. Beyond the direct economic liability, this makes agriculture the largest source of NH₃ to the atmosphere with important consequences for human health, ecosystems, and climate.³⁻⁵ The most costly impact, human health,⁶ is due to the production of fine inorganic particulate matter (PM₂.₅) as ammonium–sulfate–nitrate salts, a major contributor to PM₂.₅ mass.⁷ PM₂.₅ is a well-documented factor for premature mortality.⁸⁻⁹

Estimates of the health cost of NH₃ emissions through PM₂.₅ require accurate representation of the sensitivity of PM₂.₅ to changes in NH₃ emissions, of the relationship between PM₂.₅ and the health outcome (e.g., premature mortality), and of the valuation of the health impact. Previous work suggests that the average U.S. annual health cost (morbidity + mortality) of 1 kg of NH₃ emitted to the atmosphere ranges from 3 to 13 US$ (2006) depending on the valuation method, 2 and 9 times greater than the cost of 1 kg of SO₂ and NOₓ, respectively.¹⁰ The cost can also vary depending on the source type. For instance, Muller and Mendelsohn¹¹ found that the cost of 1 kg of NH₃ can vary from 0.1 to 73 US $ (2006). This variability reflects in part the spatial distributions associated with different NH₃ sources, with sources located closer to population centers having a greater impact.

The above estimates rely on simplified source-receptor (S-R) models that do not capture the complex nonlinear relationship between NH₃ emissions and PM₂.₅.¹²⁻¹³ This relationship is controlled by the thermodynamic equilibrium between NH₃ ↔ NH₃(g) + NH₄⁺, NOₓ ↔ HNO₃(g) + NOₓ, and SOₓ₂ ↔ SO₄²⁻ + HSO₄⁻ + H₂SO₄, where (g) denotes the gas-phase and other species are in the aerosol phase.¹⁴ The impact of NH₃ emissions on PM₂.₅ thus depends on meteorological parameters (e.g., temperature, relative humidity), the magnitude of the perturbation to NH₃ emissions, and the abundance of NO₃⁻ and SO₄²⁻, which are the products of the oxidation of SO₂ and NOₓ, two byproducts of combustion.¹⁵⁻¹⁷

Here, we focus on quantifying the cost of NH₃ emission associated with food export. Unlike previous valuation studies, we use a chemical transport model with detailed representation of aerosol thermodynamics and NH₃ losses to calculate the impact of a change in NH₃ emissions on PM₂.₅. We also account for the temporal and spatial heterogeneity of the various NH₃ sources using the model for the magnitude and seasonality of agricultural emissions (MASAGE¹⁸).

We choose to focus on food exports to reflect their growing importance for the U.S. trade balance. The U.S. is presently the largest world exporter of wheat, corn, soybeans, cotton, pork, and poultry,¹⁹ which makes it a central component of global food security.²⁰ From 2000 to 2009, 20% of U.S. agricultural production was exported, which amounted to 74 billion US$ (2006) per year and accounted for 9% of the total value of U.S.
exports. The value of U.S. agricultural export is increasing faster than other exports (doubling from 2000 to 2010), reflecting in part growing demand from China, which has become the leading export destination for U.S. food.

**MATERIALS AND METHODS**

We use the GEOS-Chem global CTM (v9.1.3) to calculate the sensitivity of PM$_{2.5}$ to NH$_3$ emissions from agricultural exports. GEOS-Chem includes a detailed representation of the photochemical production of SO$_2$ and NO$_x$. Thermodynamic equilibria between SO$_2$, NH$_3$, and NO$_x$ are simulated using ISORROPIA II. Wet scavenging is simulated as described by Liu et al. and Wang et al. for aerosols and by Amos et al. for gases. Comparisons with observations have shown these parametrizations provide an unbiased representation of wet scavenging. Dry deposition is calculated using a standard resistance-in-series model applied to a surface-type database from. GEOS-Chem is driven by assimilated meteorological data from the NASA Goddard Earth Observing System (GEOS-5) with horizontal resolution of 0.5° x 0.67° and 72 vertical levels. We degrade the horizontal resolution to 2° x 2.5° for computational efficiency.

US anthropogenic emissions of NO$_x$ and SO$_2$ are taken from the U.S. Environmental Protection Agency (EPA) National Emission Inventory for 2005 (NEI05). The parametrization of soil NO$_x$ emissions takes into account the effect of N deposition and fertilizer application as described by Hudman et al. Our simulation also includes NO$_x$ emissions from biomass burning (GFED3 with monthly temporal resolution) and lightning.

Agricultural emissions of NH$_3$ are calculated using the MASAGE model. MASAGE estimates the magnitude, seasonality, and spatial distribution of NH$_3$ emissions associated with different commodities. For instance, soybean requires little N input, whereas beef cattle, the largest source of NH$_3$ in the U.S., generates NH$_3$ emissions not only from manure but also from fertilizer used to grow feed crops (40% of corn grown in the U.S. is used as livestock feed). We also include in the model other anthropogenic sources of NH$_3$ from transportation and biofuel, and natural sources from soil, ocean, and wild animals, and open fire emissions from GFED3. U.S. NH$_3$ emissions in MASAGE are 2.7 Tg N a$^{-1}$, to which agriculture contributes more than 80%. MASAGE U.S. NH$_3$ emissions are 15% lower than the emissions from the U.S. EPA National Emission Inventory for 2006. Paulet et al. found good agreement between NH$_3$ emissions estimated by MASAGE and those inferred from inversion of NH$_4^+$ wet deposition fluxes.

We estimate the U.S. NH$_3$ emissions associated with food export by scaling crop acreages and livestock head numbers based on the commodity-specific export fraction by weight reported by the United States Department of Agriculture Economic Research Service (USDA ERS) and the Food and Agriculture Organization (FAO) averaged from 2000 to 2009. The export fraction for each commodity is assumed uniform across the U.S. We use the USDA feed index to account for the indirect export of feed crops through consumption by exported livestock. We also account for the contribution of food export to NO emissions. Food exports account for 20% of soil NO$_x$ emissions from fertilizer but this represents less than 1% of total U.S. NO$_x$ emissions. The effect on surface ozone is small (<0.6 ppbv anywhere) and will not be discussed further.

Table 1 summarizes the export fractions and the gross and net values for the main agricultural commodities exported by the U.S. Together they account for 75% of the gross value of U.S. export. Fruits, vegetables and nuts, soybeans, corn, wheat, and cotton are the most important exported commodities by value. Other commodities such as tobacco and hides account for the remaining 25% and are not considered here.

**RESULTS AND DISCUSSION**

We estimate that U.S. NH$_3$ emissions from food export are 0.3 Tg N a$^{-1}$ or 13% of U.S. agricultural emissions. This is less than the exported fraction of U.S. agricultural production (20%), reflecting the large contribution of crop production to U.S. exports. Livestock production, which is more N intensive than crops, accounts for 50% of export emissions but only 20% of the export value. Beef alone accounts for 25% of U.S. export. Fruits, vegetables, and nuts, soybeans, corn, wheat, and cotton are the most important exported commodities by value. Other commodities such as tobacco and hides account for the remaining 25% and are not considered here.

<table>
<thead>
<tr>
<th>commodity</th>
<th>fraction exported</th>
<th>export value$^a$</th>
<th>net export value$^b$</th>
<th>NH$_3$ emissions$^c$</th>
</tr>
</thead>
<tbody>
<tr>
<td>livestock</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>poultry</td>
<td>13%</td>
<td>2.8</td>
<td>1.1$^d$</td>
<td>75</td>
</tr>
<tr>
<td>pork</td>
<td>12%</td>
<td>2.6</td>
<td>0.6</td>
<td>40</td>
</tr>
<tr>
<td>beef</td>
<td>7%</td>
<td>2.9</td>
<td>0.1</td>
<td>75</td>
</tr>
<tr>
<td>dairy</td>
<td>2%</td>
<td>1.8</td>
<td>0.6</td>
<td>10</td>
</tr>
<tr>
<td>crops</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>cotton</td>
<td>69%</td>
<td>3.8</td>
<td>0.8</td>
<td>15</td>
</tr>
<tr>
<td>soybeans</td>
<td>56%</td>
<td>10.5$^e$</td>
<td>7</td>
<td>5</td>
</tr>
<tr>
<td>wheat</td>
<td>49%</td>
<td>5.5</td>
<td>2.7</td>
<td>45</td>
</tr>
<tr>
<td>rice</td>
<td>48%</td>
<td>1.3</td>
<td>0.5</td>
<td>5</td>
</tr>
<tr>
<td>feed, other</td>
<td>23%</td>
<td>5.3</td>
<td>3.3</td>
<td>20</td>
</tr>
<tr>
<td>grain</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>corn</td>
<td>22%</td>
<td>7.3</td>
<td>3.4</td>
<td>35</td>
</tr>
<tr>
<td>fruits, vegetables, nuts</td>
<td>19%</td>
<td>11.2</td>
<td>4.5</td>
<td>&lt;5</td>
</tr>
<tr>
<td>total</td>
<td>55.1$^f$</td>
<td>23.5</td>
<td>295</td>
<td></td>
</tr>
</tbody>
</table>

$^a$ Averaged from 2000 to 2009 from USDA ERS and FAOSTAT. $^b$ Calculated by multiplying the export value by the ratio of production value minus operating cost to production value (USDA ERS). $^c$ Calculated using the MASAGE model (see text). $^d$ http://extension.umd.edu/publications/publications/pdf/eb373.pdf. $^e$ http://www.soystats.com. $^f$ Total U.S. export value over the 2000–2009 period was 74 billion US$. (2006). The remainder includes a number of minor products (such as tobacco and hides) that are not considered here.
different chemical regimes. This variable sensitivity can be diagnosed using the gas-ratio (GR):\[ GR = \frac{[\text{NH}_3] - 2[\text{SO}_4]_{\text{T}}}{[\text{NO}_3]_{\text{T}}} \] (1)

When GR is greater than 1, there is enough NH\textsubscript{3} to neutralize both NO\textsubscript{3}T and SO\textsubscript{4}T and NH\textsubscript{4}NO\textsubscript{3} is only weakly sensitive to changes in NH\textsubscript{3} emissions. When 0 < GR < 1, all SO\textsubscript{4}T is neutralized and NH\textsubscript{4}NO\textsubscript{3} formation is limited by NH\textsubscript{3}, such that an increase in NH\textsubscript{3} emission correspondingly increases NH\textsubscript{4}NO\textsubscript{3} mass. When GR is lower than 0, (NH\textsubscript{4})\textsubscript{2}SO\textsubscript{4} and NH\textsubscript{4}HSO\textsubscript{4} formation are limited by NH\textsubscript{3}, and increasing NH\textsubscript{3} emissions has a small effect on PM\textsubscript{2.5} by adding mass to pre-existing sulfate aerosol. This latter regime is not found in our simulations anywhere in the U.S., consistent with observations that PM\textsubscript{2.5} is in general fully neutralized.\[ \text{PM}_{2.5} \text{ exposure of the contiguous US population} = 0.36 \mu g m^{-3} \] Based on commodity-specific export fluxes over the 2000–2009 period,\[ \Delta M = \mathcal{P} M_0 (1 - \exp(-\beta \Delta C)) \] (2)

where \( \mathcal{P} \) is the population over 30 years old, \( M_0 \) is the annual mortality rate (a\textsuperscript{-1}), \( \Delta C \) is the change in annual mean PM\textsubscript{2.5} (in \( \mu g m^{-3} \)), \( \beta \), the impact parameter, is taken from Krewski et al.\[ \beta = 5.8 \times 10^{-3} \text{ m}^3 \mu g^{-1} \] The Krewski et al. study used the U.S. EPA\textsuperscript{23,24} to estimate the health impact of SO\textsubscript{2} and NO\textsubscript{X} emissions. It shows a significantly lower health impact for PM\textsubscript{2.5} than the Harvard Six-Cities Study (\( \beta \approx 14 \times 10^{-3} \text{ m}^3 \mu g^{-1} \)) and thus provides a more conservative estimate of the excess mortality from food export. \( \Delta M \) is calculated using the US EPA BenMAP version 4.0 software.\[ \text{PM}_{2.5} \text{ from food export is responsible for} \]
Figure 3. Comparison between annual gross revenue and health cost of agricultural export for individual states. The health cost as computed here is solely driven by increased exposure to PM$_{2.5}$ due to NH$_3$ emissions from agricultural export.

$100 premature deaths (3400–6700; 95% confidence interval associated with the uncertainty in $\beta$) per year. This corresponds to ~4% of the health impact of all anthropogenic PM$_{2.5}$ and is much larger than the reduction in premature mortality (460 premature deaths) that would be achieved by a reduction of the National Ambient Air Quality Standards for PM$_{2.5}$ from 15 to 12 $\mu g\ m^{-3}$.46

We estimate that the resulting annual health cost of PM$_{2.5}$ from food export is 36 (4–100) billion US$ (2006) or 100 US$ (2006) per kg of NH$_3$. Our estimate reflects the “willingness to pay” of individuals in the U.S. for a small reduction in the risk of premature death, which is summarized through the value of a statistical life (VSL). Here we use a VSL of 4.8 million US$ (1990) (7.9 million US$ (2006)) based on 26 wage-risk and contingent valuation studies.42 Premature mortality is expected to lag PM$_{2.5}$ exposure therefore we need to discount the cost.49

Our estimate is based on a discount rate of 3%, which reflects the rate of return of long-term government debt,42 and a 20 year lag-stage structure as recommended by the U.S. EPA Science Advisory Board.46

Our estimate of the health cost of 1 kg of NH$_3$ is greater than previous estimates: 12 € in the European Union,9 0.1 to 73 US $ (2006) in the U.S.10,11,50 Part of this discrepancy can be attributed to differences in valuation methods. A discount rate of 7%, which reflects the rate of return of private capital in the U.S.,49 would reduce our estimate by 10%, while a 15 year homogeneous lag would reduce our estimate by 27% and 57% for a discount rate of 3% and 7% respectively.

The toxicity of PM$_{2.5}$ is also uncertain. Here, we assume that the toxicity of PM$_{2.5}$ is independent of its speciation. In contrast, the ExternE model, which is used to evaluate the costs and benefits of European policies (http://www.externe.info) assumes that nitrate are 50% less toxic than sulfate. Unlike metals,51 neither the health impact of nitrate or sulfate has been conclusively established by toxicological studies.52 Based on epidemiological studies53 nitrate is associated with more cardiovascular hospital admissions than PM$_{2.5}$ but fewer respiratory hospital admissions.

The largest difference with previous valuations of the cost of NH$_3$ emissions lies in the characterization of the relationship between NH$_3$ and PM$_{2.5}$. Previous studies relied on $S$–$R$ relationships derived from reduced-form techniques, which allow to explore many different scenarios through a simplified treatment of transport, wet and dry deposition, emissions, and photochemistry.12,13 For instance, NH$_3$ emissions are assumed to have no seasonality10 and ammonium nitrate production is reduced by 75% to account for the effect of temperature on the NO$_3^{-}$–NH$_3$–SO$_4^{2-}$ equilibrium.54 Because ammonium nitrate is very sensitive to wintertime NH$_3$ concentrations,28,55 these simplifications suggest that $S$–$R$ models may not be suitable to characterize the sensitivity of ammonium nitrate to NH$_3$ emissions.13 Indeed the CRDM model, a $S$–$R$ model used for many valuation studies, suggests a near linear relationship between NH$_3$ emissions and cost in the U.S. over a large range of emissions,50,56 but this is incorrect given the heterogeneity of the gas-ratio. Recent studies in Europe using detailed chemical transport models and time-resolved NH$_3$ emissions clearly illustrate the strong nonlinearity between PM$_{2.5}$ and NH$_3$ emissions.57 found that PM$_{2.5}$ was more sensitive to a 50% change in NH$_3$ emissions than to a 50% change in NO$_x$ or SO$_2$ emissions. In contrast, Pay et al.58 found that PM$_{2.5}$ was only weakly sensitive to small changes in NH$_3$ emissions.

Comparison between the cost of the increased health risk (36 (4–100) billion US$ (2006) for NH$_3$ emissions alone) associated with agricultural exports and the gross ($55 billion US $ (2006) and net value (23.5 billion US$ (2006)) of these exports (Table 1) indicates extensive negative externalities. Taking into account other impacts of agriculture (e.g., eutrophication,59,60 loss of biodiversity,61,62 and greenhouse gases emissions from production and transportation64) would further diminish the net value of agricultural exports.

Figure 3 highlights the different geographical distribution of the PM$_{2.5}$ health cost and direct gross revenue associated with agricultural exports. Regions with large agricultural activities (high NH$_3$ emissions) and low population densities (low SO$_2$ and NO emissions) clearly benefit from food exports. Most of the cost is born by populated states in the Northeast and Great Lakes region, where PM$_{2.5}$ formation is promoted by upwind NH$_3$ sources. In the Northeast, where agricultural production is small, the cost is driven by interstate transport of NH$_3$.17

Export data for 2010 onward suggest that the exported fraction of the U.S agricultural production is stable or increasing. For instance, 21% of the U.S. pork production was exported in 2011,38 a near-doubling over the 2000–2009 average fraction used in this study (Table 1). This trend may foretell the continuing growth of the U.S. NH$_3$ emissions
attributable to food export as world food demand increases.\textsuperscript{65} Greater focus on N-efficient crops (e.g., soybeans) would reduce the health impact of food exports. Previous studies have also shown that NH\textsubscript{3} emissions could be reduced through changes in fertilizer types and applications as well as manure management.\textsuperscript{6,55,56} Such measures have proven effective in Europe, where NH\textsubscript{3} emissions have decreased by nearly 30\% from 1990 to 2010.\textsuperscript{56} Reduction of the health impact of NH\textsubscript{3} emissions could also be achieved indirectly through reduction of NO\textsubscript{x} and SO\textsubscript{2} emissions, which would lower the sensitivity of PM\textsubscript{2.5} to NH\textsubscript{3}. Our work further suggests that the health impact of food production could be diminished through greater spatial segregation between food production and densely populated regions. Such an approach would result in greater food transport (“food miles”) but the impact on the carbon footprint of the food chain would likely be small.\textsuperscript{57}

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\textbf{Notes}

The authors declare no competing financial interest.

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