Climate and Air Quality Impact of Using Ammonia as an Alternative Shipping Fuel

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Abstract

As carbon-free fuel, ammonia has been proposed as an alternative fuel to facilitate maritime decarbonization. Deployment of ammonia-powered ships is proposed as soon as 2024. However, NO\textsubscript{x}, NH\textsubscript{3} and N\textsubscript{2}O from ammonia combustion could impact air quality and climate. In this study, we assess whether and under what conditions switching to ammonia fuel might affect climate and air quality. We use a bottom–up approach combining ammonia engine experiment results and ship track data to estimate global tailpipe NO\textsubscript{x}, NH\textsubscript{3} and N\textsubscript{2}O emissions from ammonia-powered ships with two possible engine technologies (NH\textsubscript{3}–H\textsubscript{2} (high NO\textsubscript{x}, low NH\textsubscript{3} emissions) vs pure NH\textsubscript{3} (low NO\textsubscript{x}, very high NH\textsubscript{3} emissions) combustion) under three emission regulation scenarios (with corresponding assumptions in emission control technologies), and simulate their air quality impacts using GEOS–Chem High Performance global chemical transport model. We find that the tailpipe N\textsubscript{2}O emissions from ammonia-powered ships have climate impacts equivalent to 5.8% of current shipping CO\textsubscript{2} emissions. Globally, switching to NH\textsubscript{3}–H\textsubscript{2} engines avoids 16,900 mortalities from PM\textsubscript{2.5} and 16,200 mortalities from O\textsubscript{3} annually, while the unburnt NH\textsubscript{3} emissions (82.0 Tg NH\textsubscript{3} yr\textsuperscript{-1}) from pure NH\textsubscript{3} engines could lead to 668,100 additional mortalities from PM\textsubscript{2.5} annually under current legislation. Requiring NH\textsubscript{3} scrubbing within current Emission Control Areas leads to smaller improvements in PM\textsubscript{2.5}-related mortalities (22,100 avoided mortalities for NH\textsubscript{3}–H\textsubscript{2} and 623,900 additional mortalities for pure NH\textsubscript{3} annually), while extending both Tier III NO\textsubscript{x} standard and NH\textsubscript{3} scrubbing requirements globally leads to larger improvement in PM\textsubscript{2.5}-related mortalities associated with a switch to ammonia-powered ships (66,500 avoided mortalities for NH\textsubscript{3}–H\textsubscript{2} and 1,200 additional mortalities for pure NH\textsubscript{3} annually). Our findings suggest that while switching to ammonia fuel would reduce tailpipe greenhouse gas emissions from shipping, stringent ammonia emission control is required to mitigate the potential adverse effects on air quality.
Introduction

Maritime shipping burns fossil fuels in large diesel engines for energy (propulsion, heat, and electricity), which leads to emissions of CO$_2$ and air pollutants. The main air pollutants emitted by the maritime transport sector include SO$_x$ (≡ SO$_2$ + SO$_4^{2-}$), NO$_x$ (≡ NO + NO$_2$), non-methane volatile organic compound (NMVOC), CO and carbonaceous aerosols. These are either components or precursors of particulate matter (PM) and ozone (O$_3$). Exposure to PM, particularly the fine PM (aerodynamic diameter < 2.5 µm, named PM$_{2.5}$) that can reach deep inside the respiratory tract, is estimated to have caused 3.7 – 4.8 million deaths in 2015 by increasing the risk of cardiopulmonary and cerebrovascular diseases (Cohen et al 2017). O$_3$ exposure exerts oxidative stress on the respiratory tract (Nuvolone et al 2018), which also leads to increased risk of cardiopulmonary diseases, and therefore another 1.04 – 1.24 millions of respiratory deaths in 2010 globally (Malley et al 2017). Shipping emissions are estimated to account for 2.7% of global energy-related CO$_2$ emissions and caused an estimated 84800 – 103000 annual premature deaths from PM$_{2.5}$ exposure globally in 2015 (Zhang et al 2021b), and account for up to 14 and 25% of PM$_{2.5}$ concentration over East Asia and Mediterranean area, respectively (Contini and Merico 2021).

The International Maritime Organization (IMO) has outlined a goal of reducing greenhouse gas (GHG) emissions from international shipping by at least 40% by 2030 compared to the 2008 level (International Maritime Organization 2018). The uses of alternative fuels (e.g. NH$_3$, H$_2$, methanol) and other energy solutions (e.g. electrification) are essential for reaching such a decarbonization goal (Balcombe et al 2019). NH$_3$ is one of the main candidates for alternative maritime fuels, and could represent up to 43% of the energy mix of shipping in 2050 (IRENA 2021). Since NH$_3$ is mainly manufactured with H$_2$ and N$_2$ through the Haber-Bosch Process, the carbon footprint of NH$_3$ production can be reduced by carbon capture (blue NH$_3$), or using renewable energy for N$_2$ and H$_2$ production and the synthesis process (green NH$_3$) (Valera-Medina et al 2021).

Wolfram et al (2022) and Bertagni et al (2023) summarized scientific concerns about the potential environmental impacts of using NH$_3$ as a marine fuel. NH$_3$ combustion may generate additional NO$_x$ and N$_2$O compared to other fuels (Hinokuma and Sato 2021). NH$_3$ emission is one of the major source of global PM$_{2.5}$ pollution (e.g. Gu et al 2021) by neutralizing H$_2$SO$_4$ and HNO$_3$ in the atmosphere (Jacob 1999). Heo et al (2016) find that NH$_3$ emission leads to much higher PM$_{2.5}$ mortality costs per ton ($23000 – 66000$) than SO$_2$ ($14000 – 24000$) and NO$_x$ ($3800 – 14000$) in the United States. These show the potential danger of uncontrolled NH$_3$ emission via worsening PM$_{2.5}$ air quality. Emitted NO$_x$ and NH$_3$ would then deposit to Earth’s surface, causing damages to ecosystems (e.g. soil acidification and eutrophication) and may lead to additional emission of N$_2$O, which is a potent greenhouse gas and contributes to stratospheric ozone depletion.

Here, we explore the possible ranges of air quality and climate impacts of transitioning from using fossil fuels to ammonia as the major shipping fuel under different technologies and policies, aiming to highlight the opportunities and challenges of ammonia combustion as a strategy to decarbonize maritime transport.
Method

We use a bottom–up approach to estimate the global NO\textsubscript{x}, NH\textsubscript{3} and N\textsubscript{2}O emissions from converting the entire fleet into NH\textsubscript{3}–powered ships as a function of engine technologies, emission control strategies and policy under 6 scenarios, using results from ammonia engine experiments and ship Automatic Identification System (AIS) data. We then simulate the associated changes in O\textsubscript{3} and PM\textsubscript{2.5} air quality using a global 3-D chemical transport model (GEOS-Chem High Performance). Finally, we estimate the impacts of simulated changes in O\textsubscript{3} and PM\textsubscript{2.5} on public health (expressed in annual premature mortalities) using concentration functions derived from epidemiological studies.

Scenarios

<table>
<thead>
<tr>
<th>Scenario Name</th>
<th>Emission control inside current ECA</th>
<th>Emission control outside current ECA</th>
<th>Equivalent policy scenario</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baseline</td>
<td>Zhang et al. (2021) inventory for 2015 shipping with 0.5% sulphur cap</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Post-2020 NO\textsubscript{x} baseline</td>
<td>Baseline with Tier III NO\textsubscript{x} (post-2020) standard imposed globally</td>
<td></td>
<td></td>
</tr>
<tr>
<td>[NH\textsubscript{3}–H\textsubscript{2}]\textsubscript{2020}</td>
<td>SCR</td>
<td>SCR</td>
<td>2020 NO\textsubscript{x} limit</td>
</tr>
<tr>
<td>[NH\textsubscript{3}–H\textsubscript{2}]\textsubscript{NH\textsubscript{3},ECA,LIM}</td>
<td>SCR+NH\textsubscript{3} scrubbing</td>
<td>SCR</td>
<td>Additional NH\textsubscript{3} limit in ECA</td>
</tr>
<tr>
<td>[NH\textsubscript{3}–H\textsubscript{2}]\textsubscript{GLOB,LIM}</td>
<td>SCR+NH\textsubscript{3} scrubbing</td>
<td>SCR+NH\textsubscript{3} scrubbing</td>
<td>Global NO\textsubscript{x} and NH\textsubscript{3} limits</td>
</tr>
<tr>
<td>[Pure NH\textsubscript{3}]\textsubscript{2020}</td>
<td>SCR</td>
<td>None</td>
<td>2020 NO\textsubscript{x} limit</td>
</tr>
<tr>
<td>[Pure NH\textsubscript{3}]\textsubscript{NH\textsubscript{3},ECA,LIM}</td>
<td>SCR+NH\textsubscript{3} scrubbing</td>
<td>None</td>
<td>Additional NH\textsubscript{3} limit in ECA</td>
</tr>
<tr>
<td>[Pure NH\textsubscript{3}]\textsubscript{GLOB,LIM}</td>
<td>SCR+NH\textsubscript{3} scrubbing</td>
<td>SCR+NH\textsubscript{3} scrubbing</td>
<td>Global NO\textsubscript{x} and NH\textsubscript{3} limits</td>
</tr>
</tbody>
</table>

Table 1. Description of the engine technology and policy scenarios considered in this study. SCR refers to Selective Catalytic Reduction (assumed to be 90% effective), which converts NO\textsubscript{x} and NH\textsubscript{3} into N\textsubscript{2} in 1:1 ratio under ideal conditions. NH\textsubscript{3} scrubbing is assumed to remove 95% of NH\textsubscript{3} slip after SCR.

In all scenarios, we apply an AIS-based shipping emission model (Zhang et al 2019) to estimate the global spatially-resolved pollutant and GHG emissions for every ship track in 2015 following the technology and policy assumptions of each scenario. The emission model calculates ship emissions as a function of engine power demand, ship specifications, emission factors (EF) and activity time. Missing entries in ship specifications are filled based on the lengths and capacities of the associated ships.

Table 1 shows the scenario design of our study. We choose the emission scenario with 0.5% cap on fuel sulphur content from Zhang et al (2021b) as our baseline. The “post–2020 NO\textsubscript{x} baseline” scenario imposes the most stringent IMO NO\textsubscript{x} emissions (Tier III) limit on top of baseline scenario, which represents the emissions from fossil fuel powered ships if all of them
were retrofitted to follow IMO emission standards for newly–built ships. 6 counterfactual scenarios are designed to examine the possible range of air quality outcomes from total conversion to ammonia-powered ships given the possible engine technologies (and therefore emission management strategies) and emission regulations (current legislation versus additional NH₃ emission regulations).

Figure 1. Load-corrected NH₃ and NOₓ emission factors (EF) of pure NH₃ and NH₃–H₂ engines, as a function of emission control strategy. Red bar (“Engine”) refers to EF from completely untreated engine exhaust. Blue (Post-SCR) and green bars (Post-SCR + NH₃ Scrubbing) refer to EF after implementations of emission control measures. SCR and NH₃ scrubbing are done sequentially. Red dotted lines indicate IMO NOₓ regulations for slow engine speed (<130 rpm), which is typical for large engine.

We consider the emissions from ammonia-powered ships with two types of engine technologies. The first type of engine technology considered is pure NH₃ combustion (Mounaïm-Rousselle et al 2022). The second type (“NH₃–H₂”) is proposed by Imhoff et al (2021) based on the experimental data from Lhuillier et al (2020). Part of the NH₃ is transferred to a catalytic NH₃ cracker to generate H₂ to improve combustion. This balances NH₃ and NOₓ concentration in engine exhaust, allowing both NOₓ and NH₃ emissions to be controlled by Selective Catalytic
Reduction (SCR). The derivations of EF and load dependences for the two types of engines, and a discussion about the uncertainty in engine technologies are given as Supplemental Information.

Given the uncertainty in ammonia engine designs, the engine technology scenarios do not intent to realistically replicate how ammonia combustion would be implemented on ships. Rather, the two engine technologies considered in our study reflects two extremes of, and therefore provide bounding scenarios for NO\(_x\) and NH\(_3\) emission management approaches: 1) with pure NH\(_3\) engine having low NO\(_x\) (currently regulated) and very high NH\(_3\) (currently unregulated) emissions, versus 2) NH\(_3\)–H\(_2\) engine that strictly maintains the NO\(_x\)/NH\(_3\) ratio to allow SCR to simultaneously control both pollutants.

We consider three policy scenarios. The first (“2020”) follows the IMO regulations as of 2020. The untreated NO\(_x\) EF are 32.7 g/kWh for NH\(_3\)–H\(_2\) and 7.08 g/kWh for pure NH\(_3\) engines following the load corrections prescribed by IMO (International Maritime Organization 2008) (fig. 1). Current IMO guidelines (International Maritime Organization 2017) cap NO\(_x\) EF for new vessels at 7.7 – 14.4 g/kWh (Tier II limit) when operating outside the Emission Control Area (ECA, mostly includes North America and United States Caribbean Sea as of 2020, and additionally Baltic Sea and North Sea in 2021) and 2 – 3.4 g/kWh (Tier III limit) within ECA, depending on the engines’ rated speed. Compliance with such a guideline would require SCR that can remove 90% of NO\(_x\) to operate globally for NH\(_3\)–H\(_2\) and within ECA only for pure NH\(_3\) engines. The second (“NH\(_3\) _ECA_LIMIT”) assumes that additional NH\(_3\) scrubbing requirements (assumed to be 95% effective from available technology) (Melse and Ogink 2005, Van der Heyden et al 2015, Boero et al 2023) are implemented within ECA for both types of engines, while the third (“GLOB_LIM”) extends Tier III NO\(_x\) compliance and NH\(_3\) scrubbing requirements to the whole globe.

**Atmospheric Chemistry Modeling**

We use version 13.4.1 of the GEOS-Chem High Performance model (GCHP, https://doi.org/10.5281/zenodo.4429193) (Martin et al 2022, Eastham et al 2018) to simulate the response of O\(_3\) and PM\(_{2.5}\) to pollutant emission changes in each scenario through resolving the chemistry, transport, emission and deposition of relevant chemical species. The model is driven by the Modern-Era Retrospective analysis for Research and Application (MERRA-2) assimilated meteorological fields (Gelaro et al 2017). The model is run at a horizontal resolution of ~200km in cubed-sphere configuration (C48) from 1st Oct 2018 to 31st Dec 2019, with the first 3 months of output discarded as spin-up. O\(_3\) is simulated from a coupled O\(_3\)-NO\(_x\)-VOCs-CO-halogen-aerosols chemical mechanism (Sherwen et al 2016). Anthropogenic emissions are from Community Emission Data System (Hoesly et al 2018) except the shipping sector. Biogenic VOCs, soil NO\(_x\) and sea salt aerosol emissions follow Weng et al (2020) and dust emissions follow Meng et al (2021). Re-emissions of deposited NO\(_x\) and NH\(_3\) are not considered. Formation of secondary inorganic aerosols are simulated by the ISORROPIA II, which considers thermodynamic equilibrium of the NH\(_4^+\)-Na\(^+\)-SO\(_4^{2-}\)-NO\(_3^-\)-Cl\(^-\)-H\(_2\)O (Fountoukis and Nenes 2007). PM\(_{2.5}\) concentrations are derived by summing the mass of its constituents at standard conditions to align with the sampling standard used by the United States Environmental Protection Agency (Latimer and Martin 2019). Ship plume chemistry is parameterized by the
PARANOX scheme (Vinken et al 2011). Model evaluation is provided as Supplemental Information.

**Health Outcome**

We estimate the impacts of air quality changes on public health using the global gridded population data at 30 arc-second resolution from the Gridded Population of the World version 4.11 (Center for International Earth Science Information Network - CIESIN - Columbia University 2018). Country-level age distribution and baseline mortality rates are provided by the World Health Organization (WHO) (WHO 2018). We estimate the risk of relative mortality from chronic O$_3$ and PM$_{2.5}$ exposure under the baseline ($RR_{\text{base}}$) and each alternative scenario $i$ ($RR_i$) for every age group. The change in the annual mortality for scenario $i$ ($\Delta\text{Mort}_i$) due to some disease for that age group is then calculated for each grid cell as:

$$\Delta\text{Mort}_i = \text{Mort}_{\text{base}} \times \frac{RR_i \times RR_{\text{base}}}{RR_{\text{base}}}$$

where $\text{Mort}_{\text{base}}$ is the number of mortalities due to that disease in 2016. The relative risk is calculated by comparing the simulated exposure-relevant concentration under scenario $i$ to that under the baseline scenario using an appropriate concentration response function (CRF). We use a log-linear CRF for O$_3$ from Turner et al (2016), which estimate a 12% increase (95% confidence interval (CI): 8.0 – 16%) in respiratory mortality per 10 ppb increase in annual mean maximum daily 8-hour average (MDA8) O$_3$ concentration. For PM$_{2.5}$ we estimate RR for non-communicable diseases and lower respiratory infections using the age-specific non-linear CRFs from the Global Exposure Mortality Model (Burnett et al 2018).

We estimate the median and 95% confidence interval of changes in mortalities due to O$_3$ and PM$_{2.5}$ for each scenario by performing 1,000 random draws of the CRF parameters in a paired Monte-Carlo simulation.
Result

Modelled Shipping Emissions

<table>
<thead>
<tr>
<th>Scenario</th>
<th>NO$_x$ (Tg/yr)</th>
<th>NH$_3$ (Tg/yr)</th>
<th>CO$_2,e$ (Tg/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baseline</td>
<td>17.2</td>
<td>0.004</td>
<td>867</td>
</tr>
<tr>
<td>Post-2020 NO$_x$ baseline</td>
<td>3.59</td>
<td>2.51</td>
<td></td>
</tr>
<tr>
<td>[NH$_3$–H$<em>2$]$</em>{2020}$</td>
<td>4.43</td>
<td>2.21</td>
<td></td>
</tr>
<tr>
<td>[NH$_3$–H$<em>2$]$</em>{NH3_ECA_LIM}$</td>
<td>4.43</td>
<td>0.125</td>
<td></td>
</tr>
<tr>
<td>[NH$_3$–H$<em>2$]$</em>{GLOB_LIM}$</td>
<td>4.43</td>
<td>50.2</td>
<td></td>
</tr>
<tr>
<td>[Pure NH$<em>3$]$</em>{2020}$</td>
<td>6.84</td>
<td>82.0</td>
<td></td>
</tr>
<tr>
<td>[Pure NH$<em>3$]$</em>{NH3_ECA_LIM}$</td>
<td>6.84</td>
<td></td>
<td>71.7</td>
</tr>
<tr>
<td>[Pure NH$<em>3$]$</em>{GLOB_LIM}$</td>
<td>0.762</td>
<td>3.92</td>
<td></td>
</tr>
</tbody>
</table>

Table 2 Modelled global total nitrogen-based air pollutants (in Tg/yr) and GHG emissions (in Tg CO$_2,e$/yr) from different scenarios. CO$_2,e$ (equivalent amount of CO$_2$ in terms of 100-year Global Warming Potential) is calculated as CO$_2$ emissions + (N$_2$O emissions × 273).

![Spatial pattern of annual total NO$_x$ emissions (kg m$^{-2}$ yr$^{-1}$) under different scenarios.](image)

Figure 2. Spatial pattern of annual total NO$_x$ emissions (kg m$^{-2}$ yr$^{-1}$) under different scenarios.

Table 2 shows the modelled global annual shipping emissions of NO$_x$, NH$_3$, and GHG under different scenarios, and Figure 2 shows the spatial distribution of NO$_x$ emissions. Under current regulations (“2020”), ammonia-powered ships have lower NO$_x$ emissions (4.4 Tg NO$_x$/yr for NH$_3$–H$_2$ and 6.9 Tg NO$_x$/yr for pure NH$_3$). Such comparison mostly reflects regulatory rather than technological differences, since the older ships in the baseline scenario do not follow the newer and more stringent (Tier II or Tier III) NO$_x$ regulations, while all newly built ammonia-powered ships abide the Tier II regulation outside ECA and Tier III regulations within ECA. To comply with Tier II NO$_x$ regulations, SCR is required for the NH$_3$–H$_2$ engine while no NO$_x$ control is needed for the pure NH$_3$ engine. This leads to higher total post-treatment NO$_x$ emissions from pure NH$_3$ engines than that from NH$_3$–H$_2$ engines, despite pure NH$_3$ engines have...
lower pre-treatment NO\textsubscript{x} emissions than NH\textsubscript{3}–H\textsubscript{2} engines. If the Tier III NO\textsubscript{x} regulations is enforced globally (“GLOB_LIM”), the NO\textsubscript{x} emission of fossil fuel (3.6 Tg NO\textsubscript{x}/yr) and NH\textsubscript{3}–H\textsubscript{2} (4.4 Tg NO\textsubscript{x}/yr) engines are similar, while pure NH\textsubscript{3} engines (0.8 Tg NO\textsubscript{x}/yr) produce the lowest NO\textsubscript{x} emissions.

Figure 3. Spatial pattern of annual total NH\textsubscript{3} emissions (kg m\textsuperscript{-2} yr\textsuperscript{-1}) under different scenarios

Figure 3 shows the spatial distribution of modelled NH\textsubscript{3} emissions under different technology and policy scenarios. Under current regulations (“2020”), switching to NH\textsubscript{3}–H\textsubscript{2} engines leads to 2.5 Tg/yr NH\textsubscript{3} emissions, while switching to pure NH\textsubscript{3} engines leads to NH\textsubscript{3} emissions (82.0 Tg/yr) that are 32.8 times higher than that from NH\textsubscript{3}–H\textsubscript{2} engines. For pure NH\textsubscript{3} engines, SCR can only remove 7% of NH\textsubscript{3} from engine exhaust, leading to high tailpipe NH\textsubscript{3} emissions. In the “NH\textsubscript{3} _ECA_LIM” scenario, which requires NH\textsubscript{3} scrubbing over ECA (mostly North American coast and northern Europe), global NH\textsubscript{3} emissions reduce by 12% for both NH\textsubscript{3}–H\textsubscript{2} (2.2 Tg/yr) and pure NH\textsubscript{3} (71.7 Tg/yr ) engines. In the “GLOB_LIM” scenario, with both SCR and NH\textsubscript{3} scrubbing are required globally, NH\textsubscript{3} emissions fall to 0.1 Tg/yr for NH\textsubscript{3}–H\textsubscript{2} engines and 3.9 Tg/yr for pure NH\textsubscript{3} engines.

Table 2 also shows the long-lived GHG emissions from each scenario, given as the equivalent amount of CO\textsubscript{2} (CO\textsubscript{2,e}) in terms of 100-year Global Warming Potential (GWP\textsubscript{100}) using a conversion factor of 273 from N\textsubscript{2}O emission to CO\textsubscript{2,e} (Smith et al 2021). CO\textsubscript{2,e} from the baseline scenario does not include GHG other than CO\textsubscript{2} (mainly CH\textsubscript{4} and N\textsubscript{2}O), which contribute to less than 3% of global shipping CO\textsubscript{2,e} during 2013 – 2015 (Olmer et al 2017). We find that the tailpipe CO\textsubscript{2,e} from the ammonia-powered fleet is 5.8% of that from the current fossil-fuel-powered fleet. Our analysis (see Supplemental Information) also shows that the “secondary N\textsubscript{2}O emissions” from reactive nitrogen deposition (Wolfram et al 2022) is not a problem for NH\textsubscript{3}–H\textsubscript{2} engine as the total reactive nitrogen emissions are lower than current fleets. For pure NH\textsubscript{3} engine, the net climate effects from nitrogen deposition are likely to be smaller than reduction in tailpipe GHG emissions (817.2 Tg CO\textsubscript{2,e}/yr) from switching to ammonia-powered ships, showing the potential of blue and green ammonia as a climate-friendly shipping fuel, though considerable uncertainties exist on how CO\textsubscript{2} uptake and N\textsubscript{2}O emissions respond to nitrogen deposition. This
analysis, however, does not fully consider the life cycle GHG emissions (e.g. energy, methane slip) of NH₃ production.

Impacts on Air Quality

Figure 4. Changes in annual mean MDA8 O₃ concentration (∆O₃, ppb) for different ammonia-powered ship scenarios

Figure 4 shows the modelled global changes in annual mean MDA8 O₃ due to converting current fleet to ammonia-powered ships with different technology and policy options. Generally, the lower NOₓ emissions from ammonia-powered ships reduce annual mean MDA8 O₃. Under all scenarios, global population-weighted average MDA8 O₃ decreases (-0.27 ppb for [NH₃-H₂]₂₀₂₀, -1.13 ppb for [Pure NH₃]₂₀₂₀, -0.37 ppbv for [Pure NH₃]GLOB_LIM). The greatest reductions in population-weighted O₃ are simulated over coastal and island nations (e.g. 1.5 to 1.9 ppb for Sri Lanka and Djibouti, 1.4 to 2.2 ppb for Panama, 1.4 to 1.7 ppb for Jamaica). However, over highly NOₓ-saturated coasts near northern China, northern Europe, and Persian Gulf, local increases in surface O₃ are simulated, especially under the scenarios with greater NOₓ reductions ([NH₃-H₂]₂₀₂₀ and [Pure NH₃]GLOB_LIM). Over North Sea, the NOₓ–saturation leads to further increases in MDA8 O₃ as NOₓ emissions become lower, increasing the population-weighted O₃ from 1 ppb under [Pure NH₃]₂₀₂₀ to up to 1.5 ppb under [Pure NH₃]GLOB_LIM over the Netherlands. Over East Asia, population-weighted MDA8 O₃ decreases by 2.4 ppb under the scenario with least NOₓ reduction ([Pure NH₃]₂₀₂₀), but increases by 0.2 ppb under [Pure NH₃]GLOB_LIM and [NH₃-H₂]₂₀₂₀ as NOₓ emissions become lower. This shows the importance of local chemical environment in controlling the response of O₃ pollution to marine NOₓ control.

In addition, we find substantial sensitivity of O₃ response to assumptions in ship plume chemistry (mainly NOₓ lifetime, see Supplemental Material), which could be a major source of uncertainties. This shows the importance of understanding the plume chemistry of NH₃ ship in capturing the O₃ response.
Figure 5 Spatial patterns of changes in annual mean PM$_{2.5}$ concentration ($\Delta$PM$_{2.5}$, $\mu$g m$^{-3}$) for all ammonia-powered ships scenarios.

Figure 5 shows the modelled changes in annual mean surface PM$_{2.5}$. Under [NH$_3$-H$_2$]$_{2020}$, population-weighted PM$_{2.5}$ increases by 0.21 $\mu$g m$^{-3}$ (0.4%) over East Asia (definition of regions follows Giorgi et al (2001)). Smaller increases are simulated over western North America (0.08 $\mu$g m$^{-3}$), though the percentage increase (1.7%) is higher since the baseline population-weighted PM$_{2.5}$ (4.82 $\mu$g m$^{-3}$) is low. PM$_{2.5}$ levels are mostly reduced over other regions in the world, especially over northern Europe and Mediterranean Basin, where population-weighted PM$_{2.5}$ decreases by 0.70 (4%) and 0.16 (0.6%) $\mu$g m$^{-3}$, respectively. Under [NH$_3$-H$_2$]$_{NH3\_ECA\_LIM}$, population-weighted PM$_{2.5}$ is reduced by 0.82 $\mu$g m$^{-3}$ (4.8%) and 0.055 $\mu$g m$^{-3}$ (0.7%) over northern Europe and the United States, respectively, as NH$_3$ emission control is enforced over those regions. Under [NH$_3$-H$_2$]$_{GLOB\_LIM}$, both Tier III NO$_x$ and NH$_3$ emission limit are extended globally, resulting in reduced PM$_{2.5}$ levels over the whole globe. Particularly, the negative impacts from NH$_3$ emission over Mediterranean Basin and East Asia are successfully mitigated, resulting in 0.33 (1.4%) and 0.62 $\mu$g m$^{-3}$ (1.2%) of reduction in population-weighted PM$_{2.5}$, respectively.

Pure NH$_3$ engines have high NH$_3$ emission, leading to higher PM$_{2.5}$ levels than NH$_3$-H$_2$ engines under the same policy scenarios. Under [Pure NH$_3$]$_{2020}$, PM$_{2.5}$ increases globally expect over the North Sea. Reduction in NO$_x$ emissions lead to lower population-weighted PM$_{2.5}$ over Netherlands (1.86 $\mu$g m$^{-3}$, 9.0%), Denmark (0.50 $\mu$g m$^{-3}$, 3.2%), and Belgium (0.35 $\mu$g m$^{-3}$, 2.0%). The largest increases in population-weighted PM$_{2.5}$ are simulated over East Asia (11.4 $\mu$g m$^{-3}$, 21.2%), North Africa (3.40 $\mu$g m$^{-3}$, 5.5%), Mediterranean Basin (3.36 $\mu$g m$^{-3}$, 14.6%), Southeast Asia (2.7 $\mu$g m$^{-3}$, 14.2%), western North America (1.20 $\mu$g m$^{-3}$, 24.8%) and eastern North America (1.88 $\mu$g m$^{-3}$, 21.7%). Under [Pure NH$_3$]$_{NH3\_ECA\_LIM}$, the increase of PM$_{2.5}$ over northern Europe (0.058 $\mu$g m$^{-3}$, 0.34% vs 0.74 $\mu$g m$^{-3}$, 4.3% under [Pure NH$_3$]$_{2020}$), eastern North America (0.35 $\mu$g m$^{-3}$, 7.2%) and western North America (0.55 $\mu$g m$^{-3}$, 6.3%) are partially mitigated by the NH$_3$ emission control. When NH$_3$ emission control is required globally ([Pure NH$_3$]$_{GLOB\_LIM}$), the spatial pattern of PM$_{2.5}$ changes largely resembles that from [NH$_3$-H$_2$]$_{2020}$ due to comparable combined NO$_x$+NH$_3$ emissions (4.7 Tg/yr for [Pure NH$_3$]$_{GLOB\_LIM}$ vs 6.9 Tg/yr for [NH$_3$-H$_2$]$_{GLOB\_LIM}$).
Despite having lower combined NO\textsubscript{x}+NH\textsubscript{3} emissions, [Pure NH\textsubscript{3}]\textsubscript{GLOB_LIM} has higher PM\textsubscript{2.5} levels than [NH\textsubscript{3}-H\textsubscript{2}]\textsubscript{2020} due to higher NH\textsubscript{3} emissions (3.9 Tg/yr for [Pure NH\textsubscript{3}]\textsubscript{GLOB_LIM} vs 2.5 Tg/yr for [NH\textsubscript{3}-H\textsubscript{2}]\textsubscript{2020}) globally except over northern Europe.

In addition, we find that NH\textsubscript{3} could potentially form PM\textsubscript{2.5} with anions and acids in sea spray, which implies extra sensitivity of PM\textsubscript{2.5} to NH\textsubscript{3} emissions that could not be controlled by reducing NO\textsubscript{x} and SO\textsubscript{x} emissions alone (see Supplemental Information).

### Health Impacts

Table 3. Estimated changes in annual global mortality attributable to PM\textsubscript{2.5} (\textDelta M\textsubscript{PM2.5}) and O\textsubscript{3} (\textDelta M\textsubscript{O3}) from each scenario. Parentheses indicates 95% confidence interval (CI) of the estimates from 1000 Monte-Carlo simulations.

<table>
<thead>
<tr>
<th>Scenario</th>
<th>\textDelta M\textsubscript{PM2.5}</th>
<th>\textDelta M\textsubscript{O3}</th>
</tr>
</thead>
<tbody>
<tr>
<td>[NH\textsubscript{3}-H\textsubscript{2}]\textsubscript{2020}</td>
<td>-16,900 (-24,000; -10,000)</td>
<td>-16,200 (-23,300; -9,000)</td>
</tr>
<tr>
<td>[NH\textsubscript{3}-H\textsubscript{2}]\textsubscript{NH3_ECA_LIM}</td>
<td>-22,100 (-29,800; -8,700)</td>
<td>-15,900 (-23,000; -8,700)</td>
</tr>
<tr>
<td>[NH\textsubscript{3}-H\textsubscript{2}]\textsubscript{GLOB_LIM}</td>
<td>-66,500 (-78,800; -54,400)</td>
<td>-12,600 (-19,900; -5,200)</td>
</tr>
<tr>
<td>[Pure NH\textsubscript{3}]\textsubscript{2020}</td>
<td>+668,100 (+542,600; +797,300)</td>
<td>-73,100 (-94,600; -51,100)</td>
</tr>
<tr>
<td>[Pure NH\textsubscript{3}]\textsubscript{NH3_ECA_LIM}</td>
<td>+623,900 (+504,000; +747,300)</td>
<td>-69,700 (-90,300; -48,700)</td>
</tr>
<tr>
<td>[Pure NH\textsubscript{3}]\textsubscript{GLOB_LIM}</td>
<td>+1,200 (-10,200; +12,700)</td>
<td>-22,400 (-31,600; -13,000)</td>
</tr>
<tr>
<td>Post-2020 NO\textsubscript{x} Baseline</td>
<td>-46,200 (-54,800; -37,700)</td>
<td>-13,000 (-21,100; -4,800)</td>
</tr>
</tbody>
</table>

Table 3 shows the changes in annual global mortality attributable to O\textsubscript{3} (\textDelta M\textsubscript{O3}) and PM\textsubscript{2.5} (\textDelta M\textsubscript{PM2.5}) for each scenario. We estimate that current shipping emissions leads to 87,400 and 16,900 mortalities from PM\textsubscript{2.5} and O\textsubscript{3}, respectively. The lower NO\textsubscript{x} emissions from ammonia-powered ships provide significant O\textsubscript{3} air quality benefit, reducing annual O\textsubscript{3}-related mortality by 12,600 to 73,100. Despite the lack of primary PM (BC, OC) and secondary PM precursors (SO\textsubscript{2}, NMVOC) emissions other than NO\textsubscript{x} and NH\textsubscript{3}, ammonia-powered ships lead to worse \textDelta M\textsubscript{PM2.5} (-22,100 to +668,100) than fossil fuel powered ships with similar NO\textsubscript{x} regulation (“Post-2020 NO\textsubscript{x} Baseline”, -46,200) except the scenario with lowest NH\textsubscript{3} emissions ([NH\textsubscript{3}-H\textsubscript{2}]\textsubscript{GLOB_LIM}, -66,500). This highlights the importance of NH\textsubscript{3} as a PM\textsubscript{2.5} precursor in coastal environment, and therefore minimizing tailpipe NH\textsubscript{3} emission to mitigate the negative air quality impacts from ammonia-powered ships.

Under currently legislation (“2020”), switching to NH\textsubscript{3}–H\textsubscript{2} engine reduces annual global mortalities from PM\textsubscript{2.5} (16,900) and O\textsubscript{3} (16,200) in comparable magnitudes. While providing substantial benefits from reducing O\textsubscript{3}–related mortality (-73,100), switching to pure NH\textsubscript{3} engines
increases in PM$_{2.5}$–related mortality (+668,100). Since current ECA are mostly over North America and northern Europe, additional NH$_3$ emissions control over current ECA (“NH$_3$ ECA _LIM”) only provides marginal benefits in terms of PM$_{2.5}$–related mortalities (5,200 (31%) for NH$_3$–H$_2$ engines and 44,200 (7%) for pure NH$_3$ engines) since most of the increases in PM$_{2.5}$ occur over East Asia, North Africa, Southeast Asia and Mediterranean region. In contrast, when both Tier III NO$_x$ and NH$_3$ emission controls are extended globally (“GLOB _LIM”), the negative impacts of pure NH$_3$ drivetrains on PM$_{2.5}$ (1,200 additional mortalities) can be mitigated to a level that could be offset by the benefits on O$_3$ (22,400 avoided mortalities). For NH$_3$–H$_2$ engines, the low NH$_3$ emissions, and therefore global reduction in PM$_{2.5}$ level, lead to substantial reduction in PM$_{2.5}$-related mortalities (-66,500) equivalent to 79% of that from current shipping emissions.

Discussion

Using blue and green NH$_3$ to facilitate decarbonization of maritime transport has been gaining traction among the industry, while concerns have been raised about the consequences (e.g. secondary N$_2$O emissions, air pollution, eutrophication, soil acidification) of such large additional reactive nitrogen production and emission into the Earth System (Baessler et al 2019, Wolfram et al 2022). Despite the uncertainties in the drivetrain design, fuel mix, emission factors and plume chemistry of ammonia-powered ships as they are not yet deployed in real world, an early evaluation using currently available information can provide information to help stakeholders identify the potential climate and air quality issues and formulate mitigation measures.

We combine results from engine experiments and ship activity data to estimate the possible GHG and air pollutant emissions and impacts from ammonia-powered ships. We find that the GWP attributable to tailpipe N$_2$O emissions from ammonia-powered fleet is a small fraction (5.8%) of that of the current fleet. Our findings confirm the potential of blue and green NH$_3$ as a climate-friendly shipping fuel. However, the impacts of large reactive nitrogen deposition over land ecosystems on GHG balance remain highly uncertain.

We find that the public health impacts of switching from fossil fuel to ammonia depends largely on the technology and policy choices. If tuned to balance NO$_x$ and NH$_3$ concentration from engine exhaust to allow simultaneous reduction of NO$_x$ and NH$_3$ emissions using well-optimized exhaust post-treatment systems with highly efficient combustion modes, deployment of ammonia combustion technology can lead to net health benefits by reducing both O$_3$ and PM$_{2.5}$ levels. If the engines are tuned to have lower NO$_x$ emissions than NH$_3$–H$_2$ combustion, which is more compatible with current NO$_x$–focused regulatory framework, the unburnt NH$_3$ emission, if unmitigated, can lead to large increases in PM$_{2.5}$, and consequently 668,100 additional global PM$_{2.5}$–related mortalities annually. Imposing NH$_3$ emission regulation over current ECA only mitigates 7% of the increases in annual PM$_{2.5}$–related mortalities from pure NH$_3$ engines, since the largest negative impacts are expected over East Asia, which is not
currently part of any ECAs. Extending stringent control of NO\textsubscript{x} and NH\textsubscript{3} emissions to the globe provides substantial air quality benefits.

Our study assumes total conversion to ammonia-powered ships, while in reality ammonia-powered ships will operate alongside SO\textsubscript{x}-emitting fossil fuel powered ships, which would increase the sensitivity of PM\textsubscript{2.5} to NH\textsubscript{3} emissions. This shows the urgency of updating shipping emission regulations in anticipation of the real-world deployment of ammonia-powered ships. Particularly, given the availability of effective (> 95%) NH\textsubscript{3} removal strategies, priority should be given towards developing and enforcing working NH\textsubscript{3} emission regulations. More stringent control of SO\textsubscript{x} and NO\textsubscript{x} emissions, which is foreseeable in the future, could be another viable strategy to reduce the PM\textsubscript{2.5} formation from unburnt NH\textsubscript{3} emissions (Bauer et al 2016).

The practicality and efficacy of SCR for ammonia engines remain highly uncertain. The lack of sulfur and particulate poisoning of catalyst, and not requiring a separate NH\textsubscript{3} source to operate could potentially lead to cheaper SCR operation since catalyst and urea recharge are estimated to account for at least 61% of the total cost of SCR ownership and operation (Zhang et al 2021a). However, NH\textsubscript{3} combustion generates more H\textsubscript{2}O than diesel combustions (see Supplemental Information), which limits the efficacy of SCR (Kuta et al 2023, Xiang et al 2024). Excessive tailpipe N\textsubscript{2}O emissions can result from mistuned SCR and ammonia oxidation systems (Yates et al 2005), which could potentially offset the climate benefits. Optimizing the SCR systems for ammonia engines is crucial to limiting their potential air quality and climate impacts.

Our study shows the feasibility of NH\textsubscript{3} to be a climate-friendly shipping fuel despite the concern of tailpipe N\textsubscript{2}O emission, and highlights the adverse effects of unburnt NH\textsubscript{3} emissions on PM\textsubscript{2.5} air quality, which can be mitigated by emission control measures feasible under current technology. Apart from tailpipe emissions, NH\textsubscript{3} leakages also occur over the whole value chain (e.g. production, distribution, bunkering, fueling) (Bertagni et al 2023), which can deteriorate the PM\textsubscript{2.5} air quality over localities near the NH\textsubscript{3} supply chain if unabated (Rathod et al 2023). Development and enforcement of new NH\textsubscript{3} emission regulations is critical for ammonia-powered ships to provide positive impact on air quality and prevent negative impacts from excessive nitrogen deposition, alongside reducing GHG emissions.

**Data Availability**

EF, shipping emission maps, GCHP modelled annual mean MDA8 O\textsubscript{3} and PM\textsubscript{2.5}, and the scripts for data analysis are available on Zenodo (https://zenodo.org/records/11237986)

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