A GLOBAL SNAPSHOT OF THE AIR POLLUTION-RELATED HEALTH IMPACTS OF TRANSPORTATION SECTOR EMISSIONS IN 2010 AND 2015

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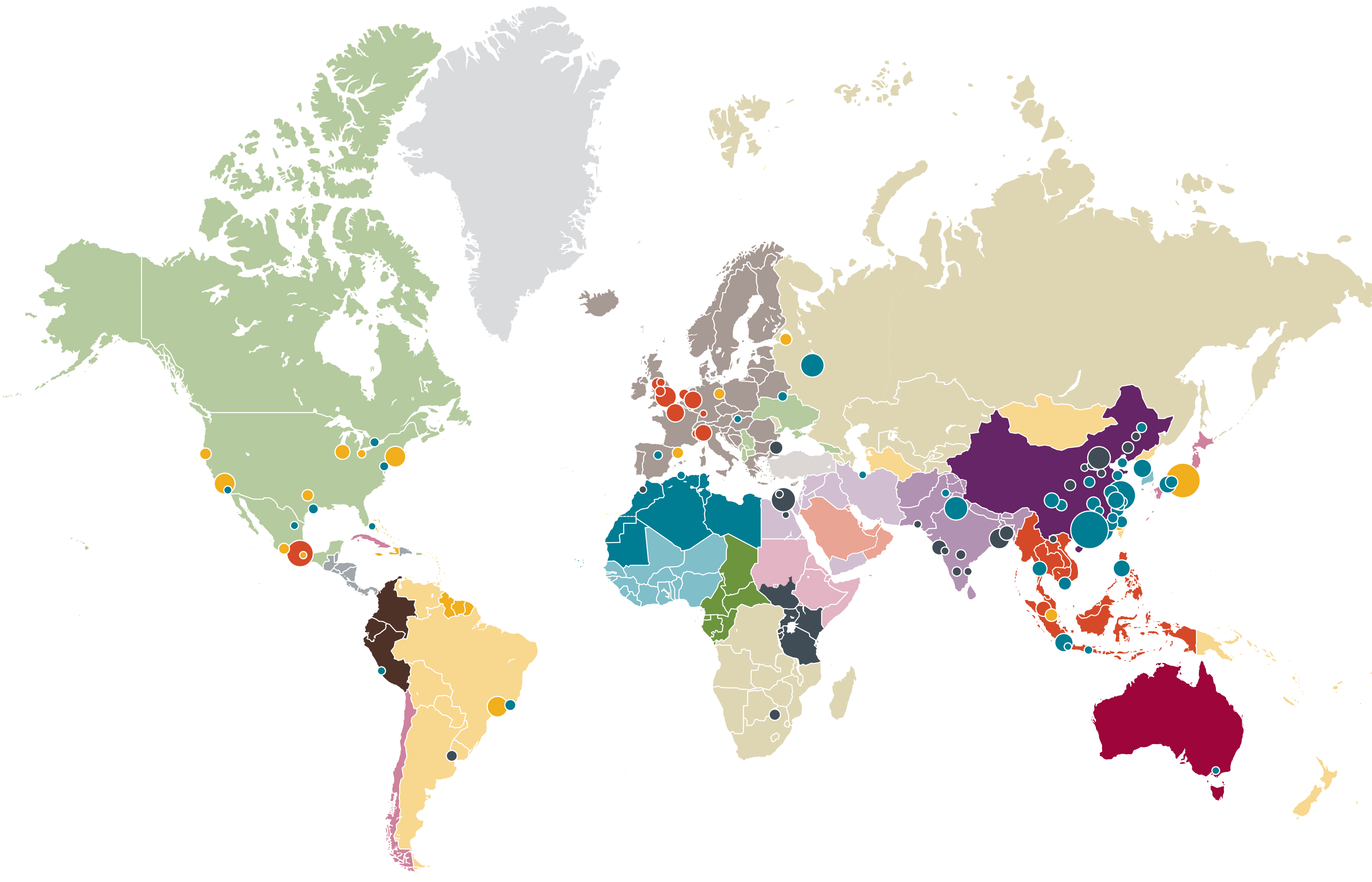

THE INTERNATIONAL COUNCIL ON CLEAN TRANSPORTATION

CLIMATE & CLEAN AIR COALITION TO REDUCE SHORT-LIVED CLIMATE POLLUTANTS

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ABOUT THE CCAC

The CCAC is a voluntary global partnership of governments, intergovernmental organizations, businesses, scientific institutions, and civil society organizations committed to catalyzing concrete, substantial action to reduce short-lived climate pollutants (SLCPs), including methane, black carbon, and many hydrofluorocarbons. The coalition works through collaborative initiatives to raise awareness, mobilize resources, and lead transformative actions in key emitting sectors. The coalition's Heavy-Duty Vehicles and Fuels Initiative works to catalyze major reductions in black carbon through adoption of clean fuel and vehicle regulations and supporting policies. Efforts focus on diesel engines in all economic sectors.

ABOUT THE ICCT

The ICCT is an independent nonprofit organization founded to provide first-rate, unbiased technical research and scientific analysis to environmental regulators. Its mission is to improve the environmental performance and energy efficiency of road, marine, and air transportation to benefit public health and mitigate climate change.

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EXECUTIVE SUMMARY

Ambient air pollution is the leading environmental health risk factor globally, resulting in nearly 3.5 million premature deaths in 2017 from stroke, ischemic heart disease, chronic obstructive pulmonary disease, lung cancer, lower respiratory infections, and diabetes. The global transportation sector is a major source of this health burden through its contribution to elevated fine particulate matter ($PM_{2.5}$), ozone, and nitrogen dioxide concentrations. Transportation activities produce tailpipe emissions, evaporative emissions, resuspension of road dust, and particles from brake and tire wear. Other important health impacts of the sector include noise, physical activity effects, and road injuries. Transportation emissions globally are driven by many factors, including economic development, which often increases personal vehicle ownership and freight activity; changes in fuel quality; and introduction of emission controls on vehicles and engines to comply with tightening environmental standards. For these reasons, transportation emissions have been changing rapidly around the world.

Prior estimates of the health burden from transportation emissions do not reflect recent advancements in global emissions inventories and in air pollution epidemiology. Updated estimates of the transportation-attributable disease burden are needed to inform decision makers tasked with reducing these impacts in international, national, and local jurisdictions. How much of the global public health burden results from transportation emissions around the world? Do certain types of vehicles contribute more than others? Which regions, countries, and cities experience the greatest transportation-related air pollution disease burden? How has this public health burden changed from 2010 to 2015? This analysis addresses these questions by providing updated estimates of the impacts of transportation sector emissions globally on $PM_{2.5}$ and ozone and their health impacts in 2010 and 2015. We restrict our analysis to the air pollution-related health impacts of transportation tailpipe emissions, because of the clear set of policies available to reduce emissions; the existence of updated global inventories of transportation tailpipe emissions; and the rapid changes underway in the determinants of these emissions across regions.

We linked state-of-the-art models on vehicle emissions, air pollution, and epidemiological models to determine how, when, and where transportation emissions are impacting air quality and public health. We advanced beyond examining the transportation sector as a whole, evaluating the health burden attributable to specific subsectors: on-road diesel vehicles, on-road non-diesel vehicles, shipping, and non-road mobile sources that include agricultural and construction equipment and rail transportation.

We estimated that emissions from the transportation sector were responsible for 11.7% of global $PM_{2.5}$ and ozone mortality in 2010 and 11.4% in 2015. Despite growth in vehicle ownership and vehicle distance traveled, the global fraction of air pollution-related premature deaths that are attributable to transportation tailpipe emissions stayed approximately the same, reflecting transportation emissions reductions in leading regulatory markets. Using methods that are aligned with the Global Burden of Disease Study 2017 (GBD 2017) data resources (Institute for Health Metrics and Evaluation, 2018), we estimated that global transportation emissions in 2010 and 2015, respectively, contributed 361,000 and 385,000 $PM_{2.5}$ and ozone-attributable premature deaths. These mortality impacts indicate that vehicle tailpipe emissions were responsible for an estimated 5.43 deaths per 100,000 people globally in 2010 and 5.38 deaths per 100,000 people in 2015. Together, $PM_{2.5}$ and ozone concentrations from transportation emissions resulted in 7.8 million years of life lost and approximately $1 trillion (2015 US$) in health damages globally in 2015.

The majority of estimated transportation emissions-related health impacts occurred in the top global vehicle markets. In 2015, 84% of global transportation-attributable deaths occurred in G20 countries, and 70% occurred in the four largest vehicle markets: China, India, the European Union (EU), and the United States. However, there is substantial heterogeneity even among the four largest markets: From 2010 to 2015, transportation-attributable deaths declined by 14% and 16% in the EU and United States but increased by 26% in China and India. The reductions in the EU and United States are attributable to the implementation of world-class standards for fuel quality and new-vehicle emissions. For example, standards such as Euro VI for heavy-duty vehicles and Tier 3 for light-duty vehicles reduce emissions of $PM_{2.5}$ by 99% or more. Concurrent with reductions in regions with world-class standards, the contribution of transportation emissions to air pollution increased in China, India, sub-Saharan Africa, Central America, parts of the Middle East and Central Asia, and Southeast Asia. The increases in India and China indicate that growth in transportation activity exceeded the reductions from emission control policies from 2010 to 2015. These analysis years are too early to capture the expected benefits of recently adopted world-class standards in countries such as India, China, Mexico, and Brazil, which will take effect between 2020 and 2023.

Of the four transportation subsectors we examined, on-road diesel vehicles contributed the most to pollution and associated disease burdens. This was particularly the case in the EU, where on-road diesel vehicles accounted for 60% of transportation-attributable $PM_{2.5}$ in 2015. The high $PM_{2.5}$ contribution of on-road diesels compared with other regions can be attributed to the high level of dieselization of the passenger car fleet, coupled with chronic elevated emissions of nitrogen oxides (NO_x) from diesel passenger cars certified to Euro 4 through Euro 6 and buses and trucks certified to Euro IV and Euro V. Combining the high contribution of on-road diesel vehicles with progress in reducing pollutant emissions from non-transport sources helps explain why the 10 countries in 2015 with the highest transportation-attributable fractions (TAF) of $PM_{2.5}$ from all sources were all in Europe.

Because transportation emissions and exposure tend to be co-located in urban areas, we estimated the TAFs and associated air pollution deaths for 100 major urban areas worldwide. The urban areas with the highest number of transportation-attributable air pollution deaths are a combination of those with the largest populations and transportation emissions. The top 10 in order for 2015 were Guangzhou, Tokyo, Shanghai, Mexico City, Cairo, New Delhi, Moscow, Beijing, London, and Los Angeles. By contrast, when normalized by population, the urban areas with the highest number of transportation-attributable air pollution deaths per 100,000 people were mainly in Europe. When considered from this perspective, the top 10 in order for 2015 were Milan, Turin, Stuttgart, Kiev, Cologne, Haarlem, Berlin, Rotterdam, London, and Leeds. Whereas urban areas in regions such as South and East Asia often have large numbers of transportation-attributable deaths, they tend to have lower TAFs due to high overall pollution concentrations from other emission sources, such as coal-fired electric generating units and residential solid fuel combustion for cooking and heating. Large numbers of transportation-attributable deaths but low TAFs indicate the need to include transportation policies in air quality management plans while also ensuring reductions in other sectors. In some cases, a few urban areas contribute a large share of the national transportation-attributable mortality burden. In addition to working with national governments to improve fuel quality and tighten vehicle emissions standards, urban areas are well-positioned to adopt policies that accelerate the introduction of low- and zero-emission vehicles, encourage fleet turnover, and restrict activity of vehicles and equipment with comparatively high emissions.

Despite recent adoption of more stringent vehicle emission regulations in some major vehicle markets, the transportation sector remains a major contributor to the air pollution disease burden globally. This points to the need for reducing emissions from the transportation sector to be a central element of national and local management plans aimed at reducing ambient air pollution and its burden on public health. Wherever world-class vehicle emissions standards have not yet been adopted, countries and trade blocs should avoid continuation of the considerable public health damages highlighted in this report by accelerating their adoption. The experiences of the top vehicle markets underscore the substantial time lag between implementation of new vehicle standards and the realization of their full benefits for the in-use vehicle fleet that results from the long lifetimes of vehicles and equipment. Countries and trade blocs that already have adopted world-class vehicle emission standards should consider accelerating the expected public health benefits with strategies to reduce emissions from in-use vehicles, such as low emission zones, retrofit/replacement/scrappage programs, and targeted fleet renewal. Urban areas and subnational jurisdictions can justify more ambitious actions because of their disproportionate exposure to transportation emissions. Reducing transportation emissions while concurrently improving access to active transportation and public transportation also would have co-benefits in terms of reduced greenhouse gas emissions and promoting more physical activity. Without these actions, the public health impacts from transportation sector emissions may grow in the future.

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1. INTRODUCTION

Ambient air pollution is the leading environmental health risk factor worldwide. In 2017, fine particulate matter ($PM_{2.5}$) was estimated to be associated with 2.9 million premature deaths from ischemic heart disease, stroke, chronic obstructive pulmonary disease (COPD), lung cancer, lower respiratory infections, and diabetes mellitus type 2 (Stanaway et al., 2018). In addition, ground-level ozone was estimated to be associated with 472,000 premature deaths from COPD. Major anthropogenic sources of air pollution globally include transportation, power generation, residential fuel combustion for energy use (e.g. cooking, heating, and lighting), industrial facilities, and agriculture. Information about the contribution of each of these sources to $PM_{2.5}$ and ozone concentrations and disease burdens in each country can highlight which sectors can be targeted to most effectively improve air quality and public health. Such information also can inform the costs and benefits of technology and policy packages capable of mitigating these impacts in individual countries. Tracking how the impact of each sector changes over time can also demonstrate the influence of changes in source sector activity levels and policies to reduce emissions. Since 2012, the Institute for Health Metrics and Evaluation (IHME) and the World Health Organization have systematically assessed the global burden of disease from ambient air pollution, with updated estimates now being produced annually. However, to date, these analyses have not addressed the contribution of individual emission source sectors to the air pollution-attributable disease burden.

Transportation sector emissions are a major source of both $PM_{2.5}$ and ozone around the world. Several studies over the past five years have assessed the impacts of transportation emissions on the global burden of disease, either within the context of all emission sectors (Lelieveld, Evans, Fnais, Giannadaki, & Pozzer, 2015; Silva, Adelman, Fry, & West, 2016) or with a focus specifically on transportation emissions (Anenberg et al., 2017; Chambliss, Silva, West, Zeinali, & Minjares, 2014). These studies consistently estimate that tailpipe emissions from transportation sources, which include on-road vehicles, shipping, and other non-road mobile sources, impose substantial public health damages, particularly in world regions where large populations are co-located with high transportation activity levels and/or lax emission regulations. Previous estimates of the global mortality burden from transportation emissions range from 165,000 (Lelieveld et al. 2015) in 2010 to 376,000 in 2005 (Silva et al., 2016) for mortality attributable to $PM_{2.5}$ and ozone. Those estimated premature deaths correspond to 5%–10% of global $PM_{2.5}$ mortality and 16% of global ozone mortality. The percentage of air pollution-related mortality that is attributable to transportation emissions was estimated to be much higher in some regions compared with the global average. For example, Silva et al. (2016) estimated that transportation emissions contributed 32% and 24% of total $PM_{2.5}$ mortality in North America and Europe, respectively, and 20%–26% of total ozone mortality in North America, South America, Europe, the former Soviet Union, and the Middle East. Lelieveld et al. (2015) also estimated that land traffic contributes 5% of the global $PM_{2.5}$ mortality, but up to 20% in Germany and 21% in the United States.

Chambliss et al. (2014) suggested an approach to estimating the transportation-attributable disease burden that could "facilitate public health surveillance through the ongoing and systematic collection, analysis and interpretation of sector-specific disease burden." Here, we provide updated estimates of the air pollution-related health impacts attributable to global transportation tailpipe emissions in 2010 and 2015.

Transportation emissions globally are driven by many factors, including economic development, which often increases personal vehicle ownership and freight activity; changes in fuel quality; and introduction of emission controls on vehicles and engines in response to tightening environmental standards. For these reasons, transportation emissions have been changing rapidly around the world. In addition, earlier estimates of the health burden from transportation emissions do not reflect recent advancements in global emissions inventories (Klimont et al., 2017; Miller & Jin, 2018) and in air pollution epidemiology (Burnett et al., 2018; Cohen et al., 2017). Estimates of global on-road diesel emissions have large uncertainty due to differences between laboratory and real-world emission factors, which were only recently discovered, and assumptions about the number and activity levels of light-duty and heavy-duty vehicles certified to each iterative emission standard in countries around the world (Anenberg et al., 2017). We address these uncertainties by using the most comprehensive and up-to-date estimates of global on-road diesel emissions. We also advance beyond examining the transportation sector as a whole, evaluating the health burden attributable to specific transportation subsectors: on-road diesel vehicles, on-road non-diesel vehicles, shipping, and non-road mobile sources that include agricultural and construction equipment and rail transportation.

The results of this study can inform public officials and other stakeholders to support decision making to control transportation emissions. In particular, our results may be informative for international, national, and local jurisdictions where current estimates of the transportation-attributable disease burden were previously unavailable, as well as for intergovernmental activities to assess and reduce emissions, such as those undertaken by the CCAC.

2. METHODS

We estimate the $PM_{2.5}$- and ozone-attributable health burden globally from transportation tailpipe emissions in 2010 and 2015. Other air pollution-related health impacts of the transportation sector include evaporative emissions, emissions from brake and tire wear, and resuspension of road dust. Additional important aspects of how the transportation sector interacts with public health include injuries and fatalities from vehicle accidents and physical activity health benefits from active transportation. We restrict our analysis to the air pollution-related health impacts of transportation tailpipe emissions. We estimate the health impacts of transportation emissions in 2010 and 2015 using a series of steps for each year: (a) obtain globally gridded emissions inventories for transportation and all other emissions from existing datasets; (b) simulate the influence of transportation emissions on $PM_{2.5}$ and ozone concentrations using the GEOS-Chem chemical transport model; and (c) estimate health impacts of $PM_{2.5}$ and ozone concentrations using epidemiologically derived health impact functions. Similar methods have been used by many other studies focusing on health impacts of ambient air pollution globally (Anenberg, Horowitz, Tong, & West, 2010; Anenberg et al., 2018; Cohen et al. 2017; Malley, Henze et al., 2017) and from various emitting sectors (Anenberg et al., 2017; Chambliss et al., 2014; Lelieveld et al., 2015; Silva et al., 2016).

2.1. EMISSIONS

In addition to examining the transportation sector as a whole, we evaluate the health burden attributable to emissions of four transportation subsectors (see Table 1). **On-road diesel vehicles** include passenger cars, light commercial vehicles, trucks, and buses with diesel engines. In China and India, this category includes three-wheeled freight vehicles used for on-road applications. Diesel is the principal fuel for these activities; this category also includes a small share of biodiesel, which is typically blended into diesel fuels. **On-road non-diesel vehicles** include passenger cars, light commercial vehicles, two-wheeled vehicles, and three-wheeled vehicles, as well as (a smaller population of) trucks and buses fueled by gasoline, liquefied petroleum gas (LPG), compressed natural gas (CNG), electricity, or other non-diesel fuels. **Non-road mobile sources** include rail transportation, agricultural equipment, construction machinery, inland shipping, and other non-road mobile machinery. Although most of the activities in this category are fueled by diesel, this category also includes activities fueled by gasoline, LPG, electricity, or other fuels. Rail is the principal source of electricity consumption in this category. **International shipping** includes container ships, bulk carriers, cargo ships, tankers, cruise ships, fishing vessels, ferries, and other service vessels. The main fuels for these activities are residual fuels, which include heavy fuel oil; diesel, also referred to as distillates; and a smaller amount of liquefied natural gas (LNG).

Table 1. Definition of transportation subsectors evaluated in this study.

Transportation Subsector	Main Fuel Types	Data Source
On-road diesel vehicles	Diesel	ICCT (Miller & Jin, 2018)
On-road non-diesel vehicles	Gasoline, LPG, CNG	IIASA (ECLIPSE v5a)
Non-road mobile sources	Diesel, gasoline, LPG, electricity	IIASA (ECLIPSE v5a)
International shipping	Residual fuels, diesel, LNG	ICCT (Comer et al, 2017)

Note: The International Institute for Applied Systems Analysis (IIASA) is an Austrian-based independent, international research institute. Its ECLIPSE project is developing and evaluating emission abatement strategies for short-lived substances such as ozone and black carbon. More information is at http://www.iiasa.ac.at/web/home/research/researchPrograms/air/ECLIPSE.en.html

Emissions for on-road diesel vehicles were obtained from Miller and Jin (2018). These emissions estimates were derived using bottom-up fleet modeling of six diesel vehicle types in 199 countries covering 99.8% of the world population in 2015. Data on vehicle sales, used vehicle imports, stock, mileage, and fuel efficiency were obtained from the International Energy Agency's Mobility Model (International Energy Agency [IEA], 2017a). Energy consumption estimates were calibrated to match IEA energy balances (IEA, 2017b), which were adjusted for some countries using estimates made by the London-based consultancy CITAC as reported by Naré and Kamakaté (2017) and data from the Joint Organisations Data Initiative (2016). Technology-specific emission factors were derived from a review of emission factor models such as MOVES (U.S. Environmental Protection Agency, n.d.) and COPERT (Gkatzolias, Kouridis, Ntziachristos, & Samaras, 2012). Region-specific emission factor adjustments to account for the effects of high emitters were applied following Klimont et al. (2017). Vehicle survival rates were calibrated to align historical estimates of vehicle sales, used imports, and in-use vehicle stock. The distribution of vehicle fleet activity by vehicle age includes mileage degradation rates obtained from the International Council on Clean Transportation's India Emissions Model. Vehicle emissions characteristics were derived using policy information on new vehicle emissions standards (ICCT & DieselNet, n.d.; United Nations Environment Programme, 2018), used vehicle import restrictions (United Nations Environment Programme and United Nations Economic Commission for Europe, 2017; U.S. Department of Commerce, 2015), and fuel quality (Malins et al., 2016). Additional information regarding emissions methods and uncertainties for on-road diesel vehicles can be found in the appendix. Emissions for on-road diesel vehicles are spatially allocated according to population and road networks, following Anenberg et al. (2017). The appendix also includes a sensitivity analysis in which on-road diesel emissions globally were adjusted up and down by 10% to account for the remaining uncertainty in these emissions estimates.

Spatially allocated emissions for international shipping were obtained from Comer et al. (2017). These emissions are spatially allocated using Automatic Identification System (AIS) data obtained from exactEarth, as described by Comer et al. (2017). All other anthropogenic emissions are from the gridded ECLIPSE emissions inventory (Klimont et al., 2017; Stohl et al., 2015), including primary and precursor emissions for ambient $PM_{2.5}$ and ozone from on-road non-diesel vehicles and non-road mobile sources, as well as from all non-transportation sources. The methods for generating the ECLIPSE emissions inventory are described by Klimont et al. (2017).

2.2. CHEMICAL TRANSPORT MODELING

We simulate ambient $PM_{2.5}$ and ozone concentrations for 2010 and 2015 using the GEOS-Chem chemical transport model at 2° x 2.5° spatial resolution.[1] We use the version of GEOS-Chem within the GEOS-Chem adjoint model v35m, which includes updates to GEOS-Chem up through v9 of the standard forward model. Assimilated meteorological fields are from the GEOS-5 product of the NASA Global Modeling and Assimilation Office, extending from the surface level up to 0.01 hPa across 47 vertical layers. Anthropogenic emissions are described above. Other sources of emissions include natural sources of NO_x from lightning (Murray, Jacob, Logan, Hudman, &

1 GEOS-Chem (http://www.geos-chem.org) is a global 3-D model of atmospheric chemistry driven by meteorological input from the Goddard Earth Observing System (GEOS) of the NASA Global Modeling and Assimilation Office.

Koshak, 2012), fertilizer and soils (Yienger & Levy, 1995), natural emissions of ammonia (NH_3) (Bouwman et al., 1997), sulfur dioxide (SO_2) from dimethyl sulfide and volcanic sources, primary biogenic organic carbonaceous aerosol from MEGAN 2.1 (Guenther et al., 2012), and monthly biomass burning from GFEDv3 (van der Werf et al., 2010). Tropospheric ozone formation includes a HO_x-VOC-NO_x mechanism, with stratospheric boundary conditions provided by the Global Modeling Initiative as implemented in Lee, Henze, Alexander, and Murray (2014). Aerosols are treated as an external mixture of primary organic and black carbonaceous aerosols (Park, Jacob, Chin, & Martin, 2003), secondary inorganic aerosols (Park, Jacob, Field, Yantosca, & Chin, 2004), natural dust (Fairlie, Jacob, & Park, 2007), and sea salt (Jaeglé, Quinn, Bates, Alexander, & Lin, 2011). Thermodynamic partitioning of secondary inorganic species is calculated using RPMARES (Park et al., 2004). Atmospheric tracers are removed through wet scavenging (Liu, Jacob, Bey, & Yantosca, 2001) and dry deposition (Wesely, 1989). Secondary organic aerosols (e.g., Pye, Chan, Barkley, & Seinfeld, 2010) and anthropogenic dust particles (Philip et al., 2017) are not treated in this version of the model. Halogen chemistry (Sherwen et al., 2016) is not included in this version of the model, which may lead to high model biases of up to 10% over land at the surface.

For each year, we conducted six 12-month model runs to simulate $PM_{2.5}$ and ozone concentrations. One base case included all emissions in the model. Five simulations zeroed out emissions of a different transportation subsector or combination of subsectors: (1) on-road diesel vehicles; (2) on-road vehicles of all fuel types; (3) international shipping; (4) non-road sources; and (5) all transportation sources. All simulations used constant non-transportation emissions and 2010 meteorology to isolate the influence of transportation emission changes. Although year-to-year meteorological differences can affect estimated concentrations, we expect that meteorological influences on concentrations would be about 20% or less of annual mean population-weighted surface $PM_{2.5}$, based on satellite-derived $PM_{2.5}$ estimates (Shaddick et al., 2018).

To capture finer scale gradients in co-located high-$PM_{2.5}$ concentrations and population, we redistributed simulated $PM_{2.5}$ concentrations output from the model to 0.1° x 0.1° resolution for input to the health impact assessment, following Anenberg et al. (2017). Within each 2° x 2.5° grid cell, we imposed the 0.1° x 0.1° spatial pattern from van Donkelaar et al. (2016), which integrates satellite aerosol optical depth with vertical aerosol profiles from a chemical transport model, and then calibrates estimated surface concentrations to ground-based monitors. Ozone concentrations are more spatially homogenous compared with $PM_{2.5}$, and previous studies have found that the spatial resolution of concentrations does not substantially affect estimated health impacts (Punger & West, 2013). Therefore, we simply regridded ozone concentrations to 0.1° x 0.1° to estimate health impacts.

For each pollutant ($PM_{2.5}$ and ozone), country, source category, and year, we calculated a transportation-attributable concentration (TAC) and transportation-attributable fraction (TAF). TAC is defined as the difference in source-specific concentrations from the zero out scenario compared to the baseline (in units of concentration). TAF is defined as the fractional difference in total mortality from the zero out scenario compared to the baseline, which is to say the percentage of total air pollution mortality that is attributable to transportation tailpipe emissions and each transportation subsector. Unlike TAC, TAF calculations are influenced by non-transportation emission sources because the denominator—total $PM_{2.5}$ or ozone mortality—is affected by many different emission sources.

2.3. HEALTH IMPACT ASSESSMENT

We first estimated the burden of disease from $PM_{2.5}$ and ozone in each 0.1° x 0.1° grid cell globally in 2010 and 2015, using methods that are consistent with the Global Burden of Disease Study 2017 (GBD 2017). To estimate the $PM_{2.5}$ and ozone disease burdens attributable to transportation emissions, we then multiplied the total $PM_{2.5}$ and ozone disease burdens by the fraction of total concentration that is attributable to transportation emissions in each grid cell, separately for each pollutant.

To estimate the total $PM_{2.5}$ disease burden, we used year-specific concentration estimates reported by Shaddick et al. (2018) for 2010 (see Figure 1) and 2015 (see Figure 2) and integrated exposure-response (IER) curves for 5-year age bands for ischemic heart disease, stroke, COPD, lung cancer, lower respiratory infections, and diabetes from GBD 2017 (Stanaway et al., 2018). The IERs integrate epidemiological associations between combustion particles and each health outcome from studies on ambient air pollution, household air pollution, environmental tobacco smoke, and active smoking.

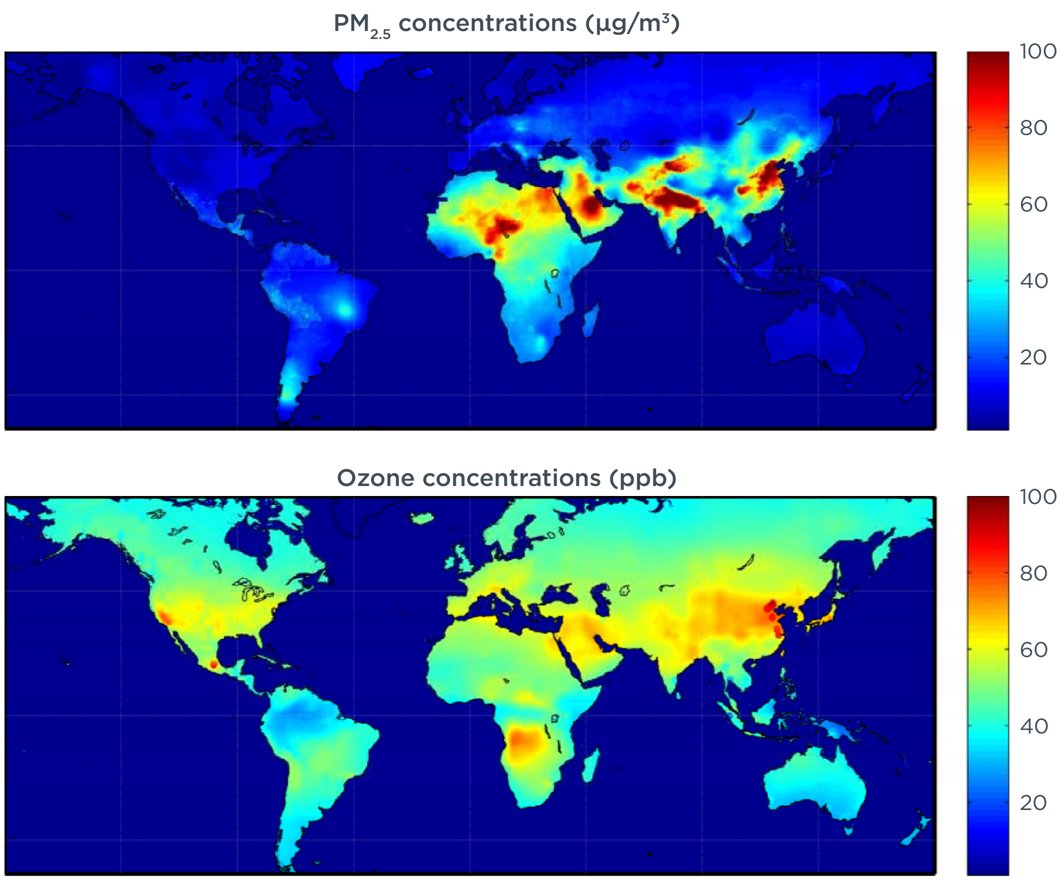

Figure 1. Total $PM_{2.5}$ (annual average) and ozone concentrations (6-month average of the 8-hour daily maximum) in 2010. $PM_{2.5}$ concentrations are from Shaddick et al. (2018). Ozone concentrations are from Chang et al. (2018). Maximum concentrations globally are 387 µg/m³ for $PM_{2.5}$ and 94 ppb for ozone.

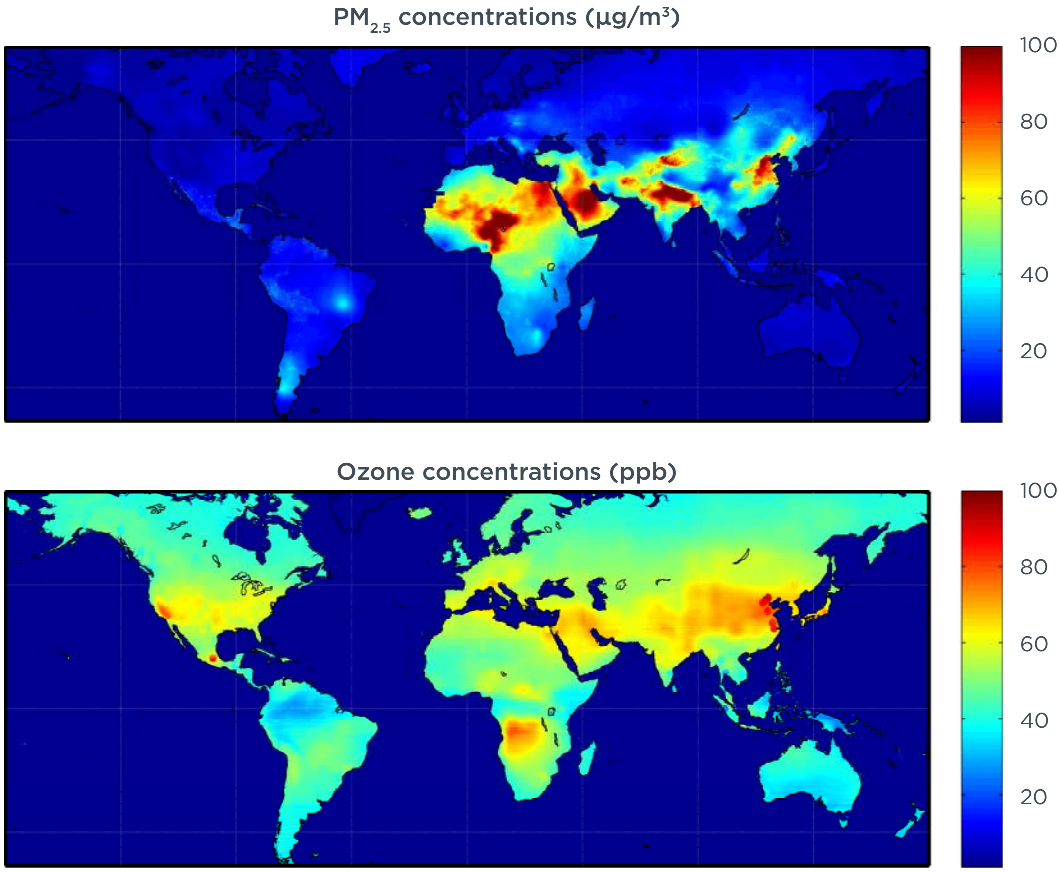

Figure 2. Total $PM_{2.5}$ (annual average) and ozone concentrations (6-month average of the 8-hour daily maximum) in 2015. $PM_{2.5}$ concentrations are from Shaddick et al. (2018). Ozone concentrations are from Chang et al. (2018). Maximum concentrations globally are 329 µg/m³ for $PM_{2.5}$ and 94 ppb for ozone.

The IER curves take the form:

$$RR_{a,i,h} = 1 + \alpha_{a,h} \{ 1 - \exp [-\gamma_{a,h} (z_i - z_{cf}) \wedge \delta_{a,h}] \} \qquad (1)$$

where *RR* is relative risk in grid cell *i* for health endpoint *h* and age group *a*, *z* is the $PM_{2.5}$ concentration in grid cell *i*, z_{cf} is the counterfactual $PM_{2.5}$ concentration below which health impacts are not calculated, and α, γ, and δ are model parameters for health endpoint *h* and age group *a*. Following Anenberg et al. (2017), we obtained 1,000 Monte Carlo draws of the IER model parameters (α, γ, δ, and z_{cf}), which we used to generate 1,000 RR estimates for each 0.1 µg/m³ $PM_{2.5}$ step in a lookup table, for each health endpoint and age group. We then applied the mean of the 1,000 RR estimates for each health endpoint and age group to the grid cell $PM_{2.5}$ concentrations to estimate age- and cause-specific premature mortality in each grid cell.

For ozone, we applied the RR for COPD of 1.06 (95% CI, 1.02–1.10) used by the GBD 2017 study based on five epidemiological cohorts in Canada, the United States, and the UK (Stanaway et al., 2018). We used year-specific 6-month average of the 8-hour daily maximum concentration estimates reported by Chang et al. (2018) for 2010 (see Figure 1) and 2015 (see Figure 2), which fused six chemical transport models and the

Tropospheric Ozone Assessment Report (TOAR) ozone monitor database (Schultz et al., 2017).

The concentration-response relationship for ozone is:

$$RR_i = \exp \hat{} \, [\beta \, (X_i - X_{cf})] \quad (2)$$

where *RR* is relative risk in grid cell *i*, *β* is the model parameterized slope of the log-linear relationship between concentration and mortality, and *X* is the 6-month average of the 8-hour daily maximum ozone concentration in grid cell *i*, and X_{cf} is the counterfactual concentration below which health impacts are not calculated. For ozone, we estimated health impacts only above 32.4 ppb, the midpoint of the uniform distribution of theoretical minimum risk exposure levels (TMREL, or X_{cf} in Equation 2) used by the GBD 2017 study.

We then calculated the $PM_{2.5}$- and ozone-attributable disease burden within each 0.1° x 0.1° grid cell using the common population attributable fraction (PAF) method:

$$M_{a,i,h} = Pop_i * Popfrac_{a,h} * Ya_{a,c,h} * [\ (RR_{a,i,h} - 1)/RR_{a,i,h}\ ]$$

where *M* is the disease burden (pollutant-attributable deaths or years of life lost) in grid cell *i* for age group *a* and health endpoint *h*, *Pop* is the population in grid cell *i*, *Popfrac* is the population fraction for age group *a* for health endpoint *h*, *Y* is the baseline incidence rate (deaths or years of life lost per 100,000 people) in country *c* for age group *a* and health endpoint *h*. The GBD 2017 began using an integrated risk model for ambient $PM_{2.5}$ and household air pollution to account for overlap between the two, resulting in lower estimates for both risk factors. To be consistent with this new approach, we scale our gridded $PM_{2.5}$ burden estimates so that national totals match the GBD 2017 national results. For example, for 2015 our calculated global total of $PM_{2.5}$ deaths was 3.9 million, whereas the GBD 2017 global total was 2.9 million. After our rescaling, our national and global totals match the GBD 2017 results. Our global ozone mortality burden estimate for COPD (427,000 deaths in 2015) is within 3% of the GBD 2017 results. We estimated total chronic respiratory mortality for ozone rather than just COPD because the American Cancer Society study, one of the largest ozone epidemiology studies used to generate the GBD RR estimates, reported RR for total respiratory disease (Turner et al., 2016). We estimate about 20% more ozone-attributable deaths when including all respiratory disease (514,000 deaths in 2015) compared with just COPD, using the same RR (1.06 per 10 ppb). $PM_{2.5}$- and ozone-attributable premature deaths were associated with an estimated 75 million years of life lost (YLL) in 2010, and 74 million YLL in 2015, with 91% of these attributable to $PM_{2.5}$ as opposed to ozone.

Country-, cause-, and age-specific baseline disease rates from the GBD 2017 study were downloaded from the Global Health Data Exchange (Institute for Health Metrics and Evaluation, 2018) and regridded from 30 arsecs (0.0083° x 0.0083°) to 0.1° x 0.1°. Gridded population estimates were from the Gridded Population of the World Version 4 dataset (Center for International Earth Science Information Network [CIESIN], 2017). We used year-specific population and baseline mortality rates to calculate best estimates for year-specific transportation health impacts. For $PM_{2.5}$ mortality, we calculated 95% confidence intervals using the 2.5th percentile and 97.5th percentile of 1,000 Monte Carlo simulations of the IER curves. For ozone mortality, we applied the 2.5th percentile and 97.5th percentile of the RR estimates.

To estimate the transportation-attributable $PM_{2.5}$ and ozone disease burdens, we multiplied the gridded TAFs for each transportation subsector by gridded total $PM_{2.5}$ and ozone disease burdens for each year. We then summed $PM_{2.5}$ and ozone deaths attributable to each subsector according to national boundaries and urban areas. Urban area definitions are from the Global Human Settlement grid (GHS-SMOD) for 2015 at 1km resolution (Pesaresi & Freire, 2016). Scaling the 1km urban definition grid to the 0.1° x 0.1° resolution of our disease burden estimates resulted in loss of urban spatial extent, population, and air pollution-attributable deaths compared with the finer resolution. Therefore, to retain as much data as possible, we multiplied our estimated air pollution-attributable deaths in each urban area at 0.1° x 0.1° by the ratio of population in each urban area calculated at high-resolution (0.0083° x 0.0083°, or ~1km) versus low resolution (0.1° x 0.1°). We used the "urban centers or high density clusters" definition, which is defined by ≥1,500 inhabitants per km^2 or a density of built-up ≥50% and ≥50,000 inhabitants. These urban definitions treat areas with dense contiguous urbanicity as one large city (e.g., Tokyo-Yokohama). GHS-SMOD city identifiers were matched to city names in ArcGIS.[2]

2.4. VALUATION

To estimate the monetary value of health damages, we apply country-specific estimates for the value of a statistical life (VSL), derived using a standard benefit-transfer approach (Viscusi & Masterman, 2017). Our main VSL estimates are calculated using the methodology recommended in Viscusi and Masterman (2017). The authors recommend a base VSL for the United States of $9.6 million, calculated using labor market (revealed preference) estimates from the Census of Fatal Occupational Injuries. This base VSL is similar to the values applied by the U.S. Department of Transportation, U.S. Environmental Protection Agency, and the U.S. Department of Health and Human Services. This base VSL estimate is then adjusted to other countries by multiplying it by the ratio of average income in the target country to income in the United States, measured using World Bank data on gross national income (GNI) per capita (Atlas method), and assuming an international income elasticity of 1.0 (Viscusi & Masterman, 2017). In 2015, the average GNI per capita in the United States was $55,980 (2015 US$).

We conduct sensitivity analysis using a second set of country-specific VSL estimates, calculated using a methodology applied by the World Bank (Narain & Sall, 2016). The World Bank methodology uses a base value of $3.8 million and a base income of $37,350, measured as GDP per capita. Both values are in 2011 international dollars and calculated using purchasing power parity (Narain & Sall, 2016). Purchasing power parity (PPP) accounts for the differences in local prices for the same amount of goods and services across countries. The World Bank methodology also applies different income elasticities depending on the income category of the country; it assumes a central value of 1.2 for low- and middle-income countries and a value of 0.8 for high-income countries. Elsewhere in this paper, we refer to the valuation estimates calculated following Viscusi and Masterman (2017) as unit elasticity (in 2015 US$) and to estimates following the World Bank methodology as differentiated elasticity (in 2011 international $).

We multiply these country-specific VSL values by national estimates of transportation-attributable deaths to yield estimates of monetized health damages. VSL estimates for 2010 are adjusted from 2015 values according to the change in the corresponding

2 ArcGis is a proprietary mapping and analysis solution available online at www.arcgis.com.

measure of real income growth. The Organisation for Economic Cooperation and Development (OECD, 2016) has previously estimated that health damages based on VSL and the number of premature deaths capture more than 80% of the total value of health damages from ambient air pollution, with the denominator including market and nonmarket costs. However, the choice of VSL methodology has a nonuniform impact on valuation estimates.

Table 2 compares the calculated estimates of VSL for G20 economies in 2015 using both VSL methodologies. Differences in these VSL estimates are driven by multiple factors, including the choice of base VSL, base income, the measure of income, and currency units. The comparison in Table 2 does not adjust for market exchange rate (MER) versus PPP, nor does it adjust for the currency year. The reason for showing the unadjusted difference is to give readers a sense of the variation to be expected when comparing the results of different studies that value the health impacts of ambient air pollution, even if restricting the comparison to studies that apply VSL.

Table 2. VSL estimates for G20 economies in 2015

Region	VSL, unit elasticity, MER, 2015 US$ (millions)	VSL, differentiated elasticity, PPP, 2011 intl $ (millions)	Percent difference (unadjusted)
Argentina	2.1	2.2	6%
Australia	10.4	4.4	-58%
Brazil	1.7	1.3	-28%
Canada	8.1	4.3	-47%
China	1.4	1.1	-17%
France	7	3.9	-45%
Germany	7.9	4.4	-44%
India	0.3	0.4	48%
Indonesia	0.6	0.8	40%
Italy	5.7	3.6	-37%
Japan	6.7	3.9	-42%
Mexico	1.7	1.5	-12%
Other EU-28 (average)	4.9	3.4	-29%
Russian Federation	2	2.3	15%
Saudi Arabia	4.1	4.9	20%
South Africa	1	1	-2%
South Korea	4.7	3.6	-24%
Turkey	2.1	2.2	6%
United Kingdom	7.5	4	-47%
United States	9.7	5.1	-48%

3. RESULTS

3.1. TRANSPORTATION-ATTRIBUTABLE FRACTION (TAF) OF AMBIENT CONCENTRATIONS

Figure 3 shows national population-weighted TAF for ambient $PM_{2.5}$, ozone, and black carbon (BC) in 2010 and 2015. As discussed previously, a high TAF means that transportation tailpipe emissions contribute a relatively high fraction of ambient air pollutant concentrations. Because the denominator includes air pollution from all emissions sources, a low TAF indicates the importance of addressing emissions from other sectors in addition to any measures addressing transport emissions.

We estimate that in 2015, the global population-weighted TAF for annual average $PM_{2.5}$ was 11.6%, equivalent to a TAC of 3.0 µg/m^3 (see also Table 3). The global TAF for ozone was 10.7%, equivalent to a TAC of 5.6 ppb (measured as the 6-month average of the 8-hour daily maximum ozone). Some countries experience far greater air pollution impacts from transportation emissions than the global population-weighted average. In 2015, the highest contributions of transportation emissions to national population-weighted annual average $PM_{2.5}$, ozone, and BC concentrations, respectively, were 7.2 µg/m^3 (China), 15 ppb (Qatar), and 0.64 µg/m^3 (Bahrain). In 2015, the global population-weighted mean BC concentration from transportation emissions was 0.2 µg/m^3. Transportation-attributable BC concentrations varied substantially around the world, with national population-weighted mean concentrations ranging from 0–0.64 µg/m^3 (median = 0.08 µg/m^3) and national BC TAFs ranging from 0%–88% (median = 27%). In 2015, the national BC TAF exceeded 50% in 30 countries, indicating that in these countries, more than half of BC concentrations were attributable to transportation tailpipe emissions.

All 10 countries with the highest TAF for $PM_{2.5}$ in 2015 were in Europe. In the EU, the $PM_{2.5}$ TAF in 2015 was 26% and the ozone TAF was 10%. On-road diesel vehicles accounted for 60% of transportation-attributable $PM_{2.5}$ in the EU in 2015. The high $PM_{2.5}$ contribution of on-road diesels in the EU compared with other regions can be attributed in part to the high level of dieselization of the light-duty vehicle (LDV) fleet. Diesels made up one-third of new passenger car sales in 2001 and more than half of new passenger car sales in all but one year from 2006 to 2015 (ICCT 2019); however, EU emission standards did not require new diesel LDVs to be equipped with diesel particulate filters (DPF) until January 2013 (EU: Light-duty: Emissions, n.d.). In addition to directly emitted $PM_{2.5}$, diesel engines emit NO_x and SO_2, which produce secondary $PM_{2.5}$ in the form of nitrates and sulfates (Guerreiro et al., 2018). Since 2009, the EU has limited the sulfur content of diesel and gasoline fuels to 10 parts per million (ppm) for both on-road and non-road applications, resulting in concomitant reductions in SO_2 emissions (European Environment Agency, 2015). Even so, NO_x emissions from on-road diesel vehicles have remained high despite a tightening of regulatory certification limits from Euro 4 to Euro 6 for LDVs and Euro IV to Euro V for heavy-duty vehicles (HDV). These elevated NO_x emissions from on-road diesel vehicles were a key contributor to secondary $PM_{2.5}$ in the EU in 2015 (Anenberg et al., 2017; Jonson et al., 2017). Although Euro VI HDV standards have since proven effective in controlling both $PM_{2.5}$ and NO_x emissions, these were only applied to all new HDV sales starting in 2014 (EU: Heavy-duty: Emissions, n.d.). Similar filter-forcing standards for non-road equipment were adopted in 2016, to be applied to all new equipment sales in 2019 (Shao & Dallmann, 2016).

The United States also had high TAFs of 25% for $PM_{2.5}$ and 12% for ozone in 2010. As in the EU, one potential factor in a high TAF is a low denominator, indicating successful efforts to control emissions from other sources. The U.S. $PM_{2.5}$ TAF declined 13% from 2010 to 2015 (from 25% in 2010 to 21% in 2015), indicating that the introduction of world-class standards in the United States (e.g., Tier 2 LDV standards and EPA 2010 HDV standards) resulted in the contribution of transportation sources declining faster than other sectors over that period.

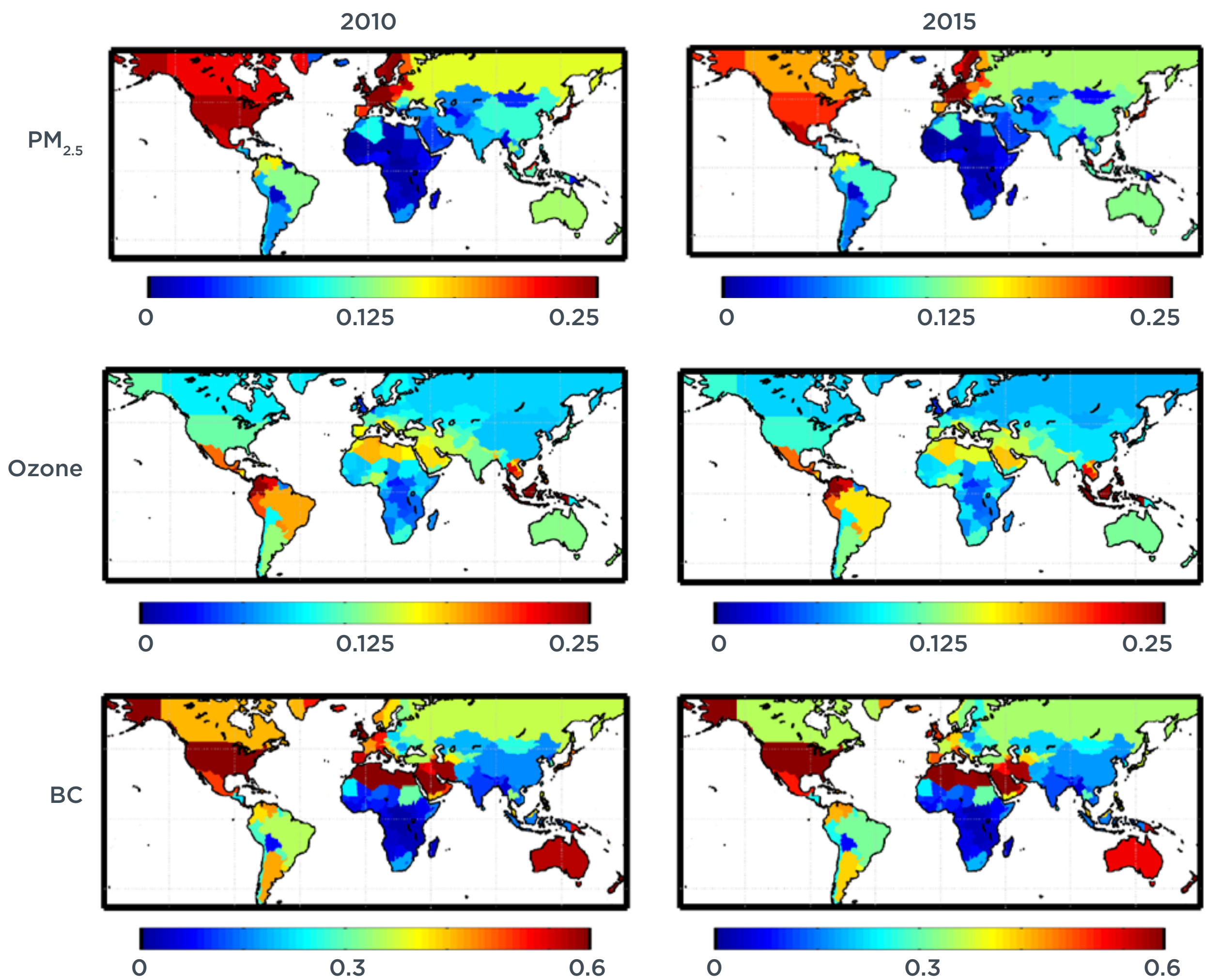

Figure 3. National population-weighted TAF (unitless) for $PM_{2.5}$, ozone, and BC in 2010 and 2015. BC TAFs are estimated from simulated concentrations because we did not estimate BC-specific mortality separately from $PM_{2.5}$ mortality.

3.2. CHANGES IN TRANSPORTATION-ATTRIBUTABLE CONCENTRATION (TAC) FROM 2010 TO 2015

Whereas TAF is an indicator of the relative contribution of transportation tailpipe emissions compared with other sources of air pollution, the change in transportation-attributable concentrations (TAC) is a more direct indicator of progress in reducing air pollution from transportation emissions over time. Figure 4 shows the change in national population-weighted average TAC from 2010 to 2015 and Figure 5 shows scatterplots of national TACs in 2010 versus 2015. Shifts in transportation emissions are driven by changes in transportation activity (vehicle ownership, vehicle miles traveled), advancing technology, and tightening transportation emission policies in some countries. Whereas improvements in fuel quality have immediate fleetwide benefits in the form of reduced SO_2 and sulphate emissions, reductions in other key pollutants such as BC and NO_x rely on the introduction of corresponding vehicle emission controls.

In key countries globally, the contribution of transportation emissions to $PM_{2.5}$, ozone, and BC concentrations has declined substantially from 2010 to 2015, suggesting that progress has been made in terms of reducing the impact of the transportation sector on air pollution. From 2010 to 2015, TACs of $PM_{2.5}$, ozone, and BC declined in all regions that implemented world-class standards for fuel quality and new vehicle emissions, including the United States and Canada, EU and European Free Trade Association countries, and Japan. For example, in the United States, transportation-attributable $PM_{2.5}$, ozone, and BC concentrations declined from 2010 to 2015 by 17%, 10%, and 22%, respectively. Declines were even larger in Canada (25%, 11%, and 34% for $PM_{2.5}$, ozone, and BC) and Japan (19%, 15%, and 43%). TACs in regions with world-class standards can be expected to decline further with fleet turnover. TACs also are estimated to have declined in other regions that implemented progressive stages of emission controls, including Brazil, Argentina, Chile, Russia, and Australia.

In contrast, TACs are estimated to have increased in regions where growth in transportation activity outpaced the implementation of transportation emission control policies. Estimated increases in TACs are most evident in sub-Saharan Africa, Central America, parts of the Middle East and Central Asia, and Southeast Asia. For example, in India, transportation-attributable $PM_{2.5}$, ozone, and BC concentrations increased from 2010 to 2015 by 8%, 4%, and 10%. In China, transportation-attributable $PM_{2.5}$, ozone, and BC concentrations increased by 20%, 8%, and 9% over the same period. Although China and India have progressively tightened their transportation emission control policies, the most stringent policies have not yet taken effect. In particular, India's Bharat VI standards apply in 2020 (Dallmann & Bandivadekar, 2016), and China 6 LDV and China VI HDV standards apply between 2020 and 2023 (He & Yang, 2017; Yang & He, 2018). Increases in TACs from 2010 to 2015 in India and China indicate that the effect of growth in transportation activity exceeded the reductions from emission control policies over that period.

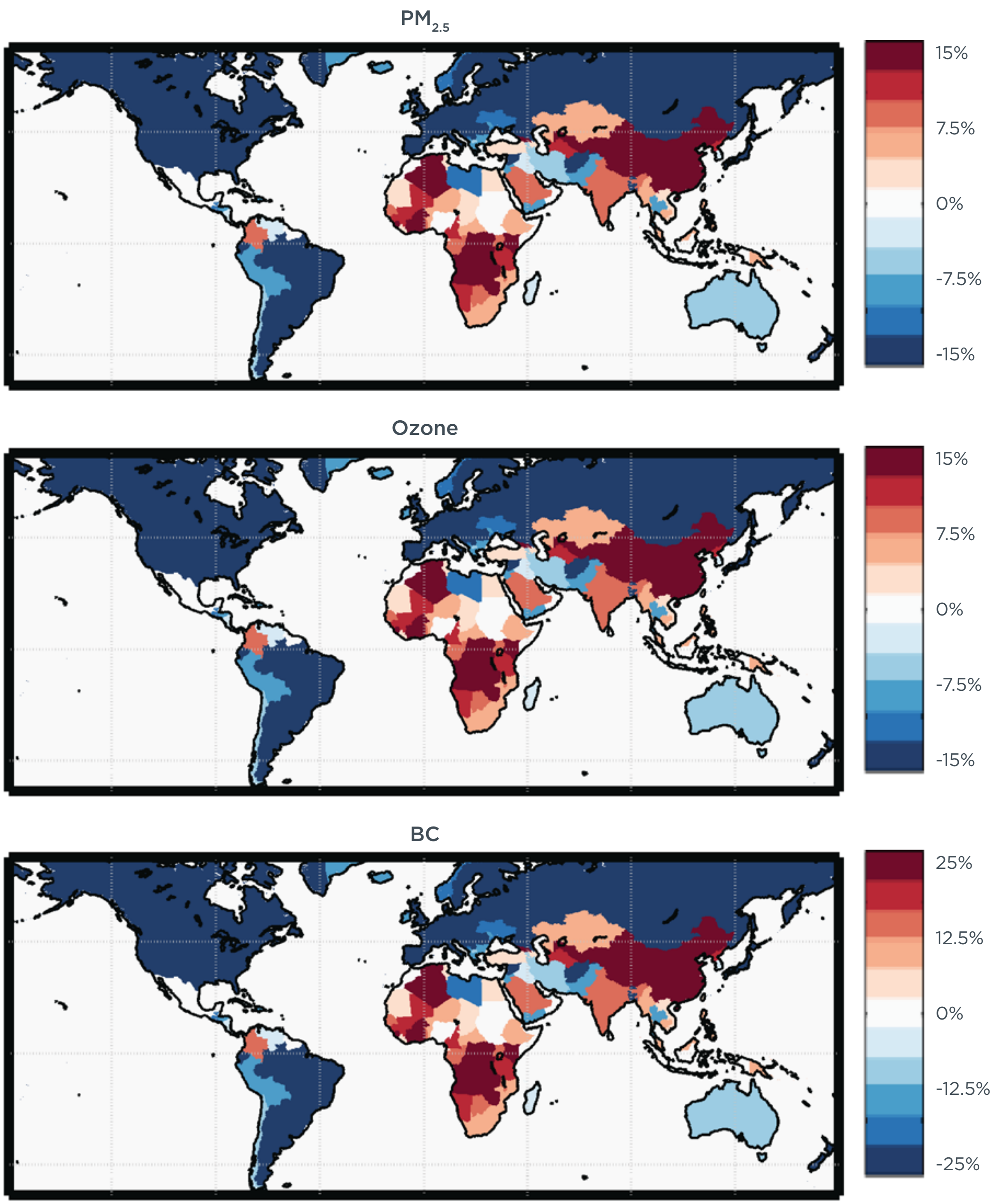

Figure 4. Maps showing the change in national population-weighted average transportation-attributable concentrations from 2010 to 2015 (annual average concentration for $PM_{2.5}$ and BC, 6-month average of the 8-hour daily maximum for ozone).

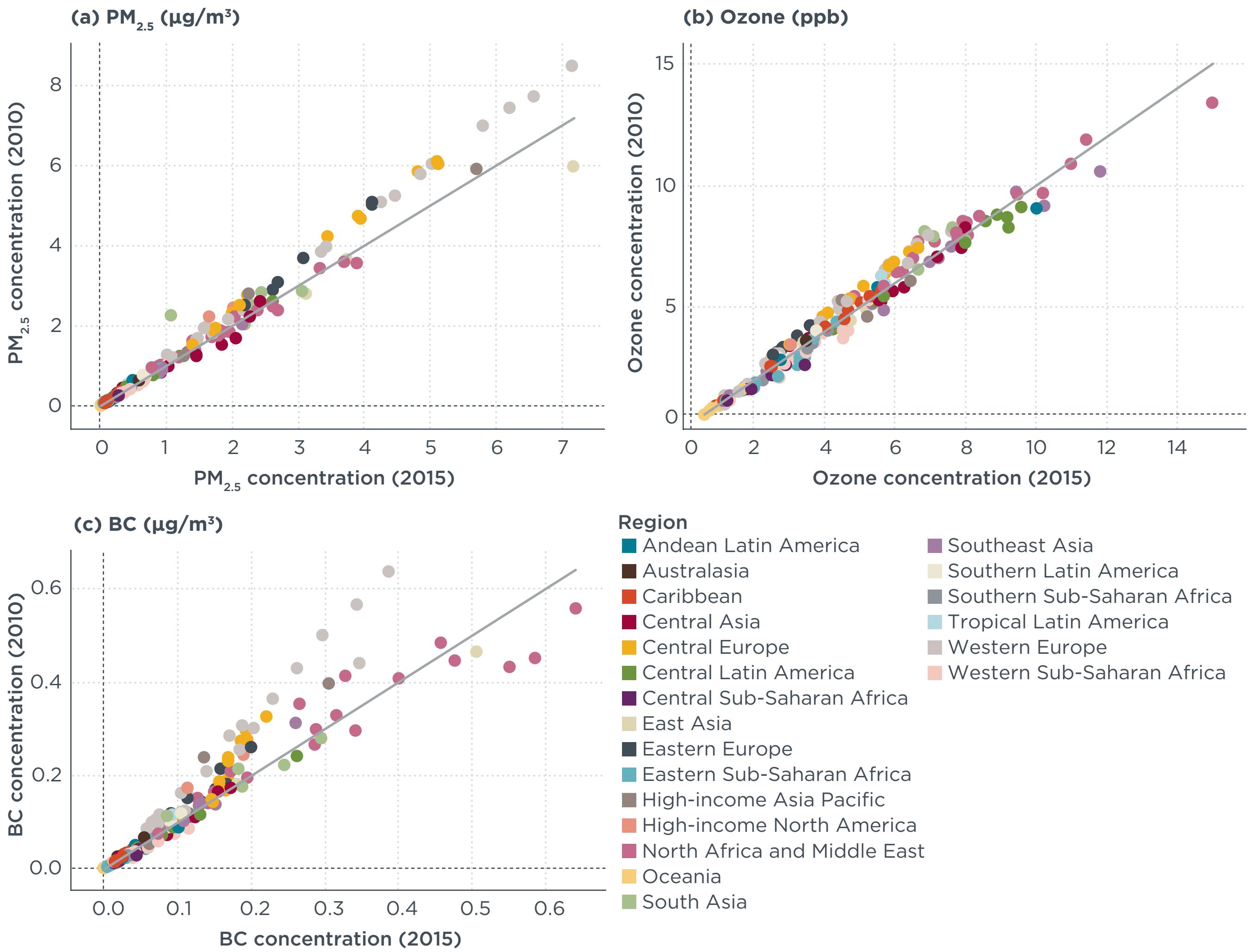

Figure 5. Scatterplots of annual average $PM_{2.5}$, 6-month average of the 8-hour daily maximum ozone, and annual average black carbon (BC) concentrations in 2010 versus 2015 by country.

3.3. GLOBAL IMPACTS OF TRANSPORTATION TAILPIPE EMISSIONS

Table 3 summarizes our estimates of the global air quality and health impacts of transportation tailpipe emissions in 2010 and 2015. At the global level, the impacts of transportation tailpipe emissions have changed little from 2010 to 2015. Global population-weighted TACs of $PM_{2.5}$ and ozone increased by just under 3.5% and 2%, respectively, from 2010 to 2015, using 2010 meteorology for both years to isolate the influence of transportation emission changes. Over this period, global transportation-attributable deaths from tailpipe emissions increased by 6.6% while total global population increased by 7.7% and total $PM_{2.5}$ and ozone mortality increased by 9.4%. These estimated transportation-attributable health burdens represent 11.7% of global $PM_{2.5}$ and ozone deaths in 2010 and 11.4% in 2015. We estimated that vehicle tailpipe emissions were associated with 361,000 (95% CI, 258,000–462,000) premature deaths globally in 2010, and 385,000 (95% CI, 274,000–493,000) in 2015. Confidence intervals here represent uncertainty in the concentration-response function only (see Section 4 for additional details on other sources of uncertainty). Central estimates of mortality impacts translate into 5.43 deaths per 100,000 people globally in 2010 and 5.38 deaths per 100,000 people in 2015. Global transportation-attributable deaths were associated with an estimated 7.9 million YLL in 2010, and 7.8 million YLL in 2015, with 88% of these from $PM_{2.5}$. The global welfare loss associated with transportation-attributable deaths was approximately $1 trillion (in 2015 US$) in 2010 and 2015. These damages in 2015 are approximately 0.6% higher than in 2010 and represent 17% of total ambient $PM_{2.5}$- and ozone-related health damages in 2015. Applying the World Bank methodology for VSL yields a 2015 estimate of approximately $670 billion in PPP-adjusted 2011 international dollars. Overall, $PM_{2.5}$ and ozone mortality from on-road diesel emissions increased by 2%, while those from on-road non-diesel emissions decreased by 9%. However, these global results mask substantial heterogeneity among regions and countries around the world (see Figure 6), with some countries exhibiting substantial changes in transportation contributions to air pollution within just these five years (see Figure 7).

Table 3. Global air quality and health impacts of transportation tailpipe emissions in 2010 and 2015. For premature deaths, 95% confidence intervals reflect uncertainty in the relative risk estimate only.

Measure	Description	Metric	2010	2015
Transportation-attributable concentration (TAC)	How much do tailpipe emissions from transportation sources contribute to global population-weighted air pollutant concentrations? Units: depends on pollutant	annual average $PM_{2.5}$	2.9 μg/m³	3.0 μg/m³
		6-month average of the 8-hour daily maximum ozone	5.5 ppb	5.6 ppb
		annual average BC	0.2 μg/m³	0.2 μg/m³
Transportation-attributable deaths	How many premature deaths are associated with global transportation-attributable concentrations of $PM_{2.5}$ and ozone? Units: thousands (95% confidence interval)	ambient $PM_{2.5}$ deaths	312 (240–386)	330 (255–408)
		ambient ozone deaths	49 (18–76)	55 (20–85)
		total ambient $PM_{2.5}$ and ozone deaths	361 (258–462)	385 (274–493)
Transportation-attributable fraction (TAF)	What fraction of ambient air pollution deaths are attributable to tailpipe emissions from transportation sources? Units: percent	$PM_{2.5}$	11.9%	11.6%
		ozone	10.4%	10.7%
		total $PM_{2.5}$ and ozone	11.7%	11.4%
Transportation health damages	What is the welfare loss due to global transportation-attributable deaths? Units: 2015 US$	$PM_{2.5}$	$900 billion	$891 billion
		ozone	$70 billion	$85 billion
		total $PM_{2.5}$ and ozone	$970 billion	$976 billion

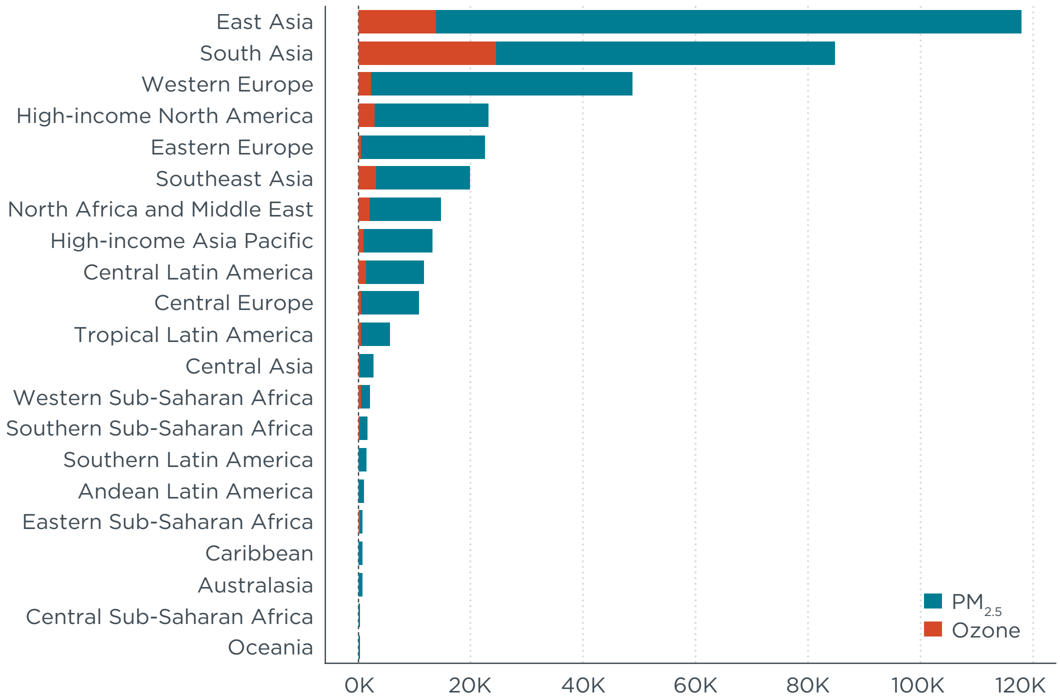

Figure 6. Total number of transportation-attributable $PM_{2.5}$ and ozone deaths in 2015 by world region.

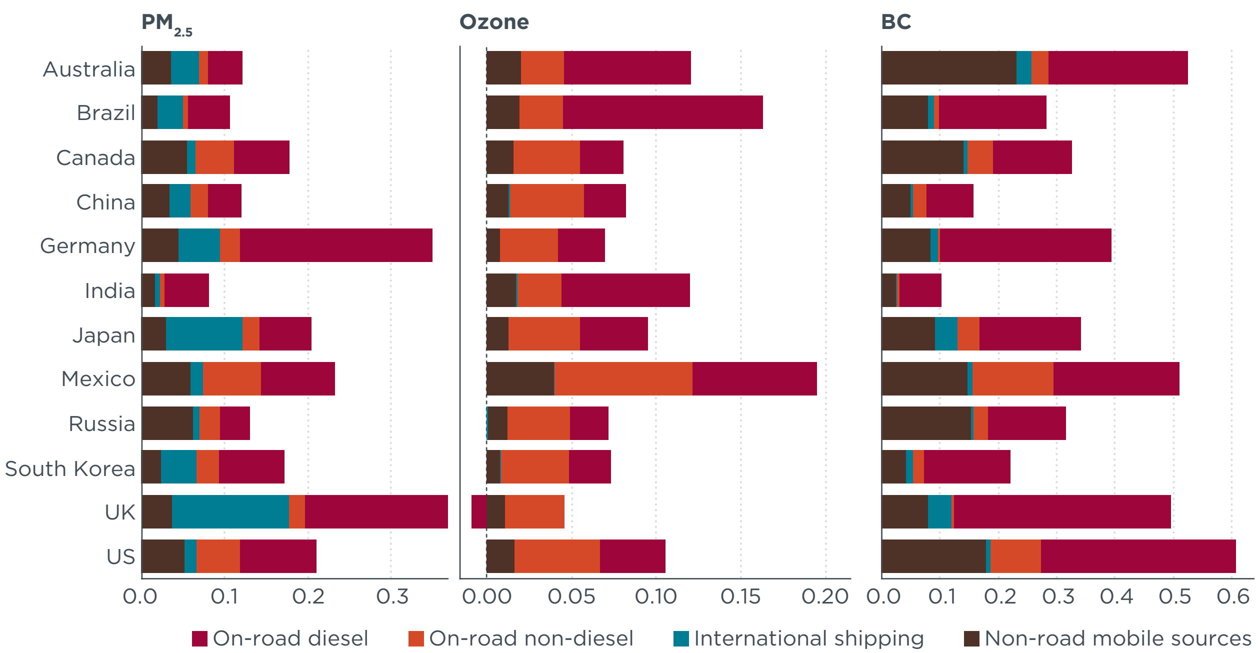

Figure 7. Subsector-specific TAFs in key vehicle markets in 2015.

3.4. IMPACTS IN G20 ECONOMIES

G20 economies collectively account for approximately two-thirds of the world's population; half of global economic output; and 80% of transportation energy demand, on-road vehicle stock, and vehicle activity. Of the top 100 most populated urban areas in the world, 70 are located in G20 economies. In this sense, it may come as no surprise that G20 economies are responsible for 77% of ambient $PM_{2.5}$ and ozone deaths from all emissions sources and 84% of transportation-attributable deaths (see Table 4). Because per capita income is a primary determinant of country-specific VSL estimates, G20 economies account for an even greater share of the global valuation of health damages from transportation tailpipe emissions: an estimated 94% in 2015. These damages amount for a sizeable fraction of total income in G20 economies in 2015 (measured as a share of GNI): up to 2.83% in Germany and an average of 1.37% among G20 economies.

These impacts are further concentrated in the top four vehicle markets: China, the EU, the United States, and India. In 2015, these four markets accounted for 70% of global transportation-attributable $PM_{2.5}$ and ozone deaths but just under half of the global population. China, the EU, the United States, and India had total TAFs of 11%, 24%, 19%, and 9% in 2015, respectively, including $PM_{2.5}$ and ozone contributions.

Within transportation, on-road diesel vehicles were the leading contributor to transportation health damages in nearly all G20 economies in 2015; in Germany, France, Italy, and India, they accounted for two-thirds of transportation health damages. On-road non-diesel vehicles, non-road mobile sources, and international shipping were each the second-largest subsector in one or more G20 economies. International shipping accounted for a particularly high share of transportation health damages in Japan and the United Kingdom; on-road non-diesel vehicles had a particularly high share in North America; and non-road mobile sources had particularly high shares in South Korea and Argentina.

Table 4. Air quality and health impacts of transportation tailpipe emissions in G20 economies in 2015. Estimates are rounded to two or three significant figures.

	Ambient $PM_{2.5}$ and ozone deaths			Transportation health damages		Share of transportation-attributable deaths by subsector			
Region	Transportation-attributable deaths	All emission sources	Transportation-attributable fraction (%)	Transportation health damages (billion 2015 US$)	As a share of GNI, Atlas method (%)	On-road diesel vehicles	On-road non-diesel vehicles	Non-road mobile sources	International shipping
Argentina	970	16,000	6%	2.1	0.38%	38%	9%	38%	15%
Australia	620	4,800	13%	6.4	0.45%	36%	9%	29%	25%
Brazil	5,700	52,000	11%	9.9	0.47%	50%	7%	18%	25%
Canada	1,400	8,700	16%	12	0.68%	37%	28%	30%	5%
China	114,000	1,020,000	11%	160	1.43%	34%	21%	27%	18%
France	6,400	20,000	32%	45	1.65%	66%	5%	12%	18%
Germany	13,000	43,000	31%	110	2.83%	66%	8%	13%	14%
India	74,000	800,000	9%	20	0.96%	66%	10%	19%	5%
Indonesia	7,100	54,000	13%	4.2	0.47%	34%	29%	10%	27%
Italy	7,800	32,000	25%	44	2.21%	66%	6%	11%	17%
Japan	9,900	52,000	19%	66	1.34%	32%	13%	15%	41%
Mexico	8,100	36,000	23%	14	1.11%	39%	31%	25%	6%
Russian Federation	13,000	104,000	13%	27	1.55%	27%	21%	48%	5%
Saudi Arabia	420	8,800	5%	1.7	0.23%	55%	17%	10%	18%
South Africa	1,400	20,000	7%	1.5	0.44%	48%	24%	11%	16%
South Korea	2,900	18,000	16%	14	0.97%	45%	18%	13%	24%
Turkey	2,900	41,000	7%	6.0	0.64%	46%	11%	19%	25%
United Kingdom	8,400	25,000	33%	63	2.20%	46%	6%	10%	38%
United States	22,000	115,000	19%	210	1.16%	43%	28%	24%	6%
Other EU	22,000	119,000	18%	110	1.68%	58%	9%	15%	19%
G20 subtotal	323,000	2,590,000	12%	914	1.37%	47%	16%	22%	15%
Global	385,000	3,370,000	11%	976	0.72%	47%	17%	21%	16%
G20 share of global	84%	77%		94%					

Population size is also a key factor in the absolute number of transportation-attributable $PM_{2.5}$ and ozone deaths in G20 economies (see Figure 8). These estimates are shown on a logarithmic scale to account for the substantial differences in population size between China and India compared with smaller countries such as Australia and Saudi Arabia. The size of each point corresponds to the transportation-attributable mortality rate per 100,000 population. In 2015, most G20 economies had higher transportation-attributable mortality rates than the average of non-G20 economies. Some countries—notably Germany, the United Kingdom, Italy, and France—had transportation-attributable mortality rates between 4 (France) and 7 (Germany) times the average of non-G20 economies. These elevated rates compared to non-G20 economies are likely attributable to a combination of factors, including high levels of motorization and freight activity in all G20 economies; high levels of LDV dieselization in the EU and India; and relatively high baseline incidence rates for diseases that are affected by air pollution, particularly in Russia, the EU, China, the United States, and Japan. As discussed earlier, the relatively recent implementation of world-class standards in the United States, Canada, the EU, Japan, South Korea, and Turkey means that the benefits of these standards—which apply only to new vehicles—have yet to be fully realized for the existing (in-use) vehicle fleet. The proven benefits of advanced vehicle emission control technologies also demonstrate that absent such policies and earlier stages of emission control, the situation in 2015 would have been considerably worse (Bishop, Schuchmann, & Stedman, 2013; Chambliss, Miller, Façanha, Minjares, & Blumberg, 2013).

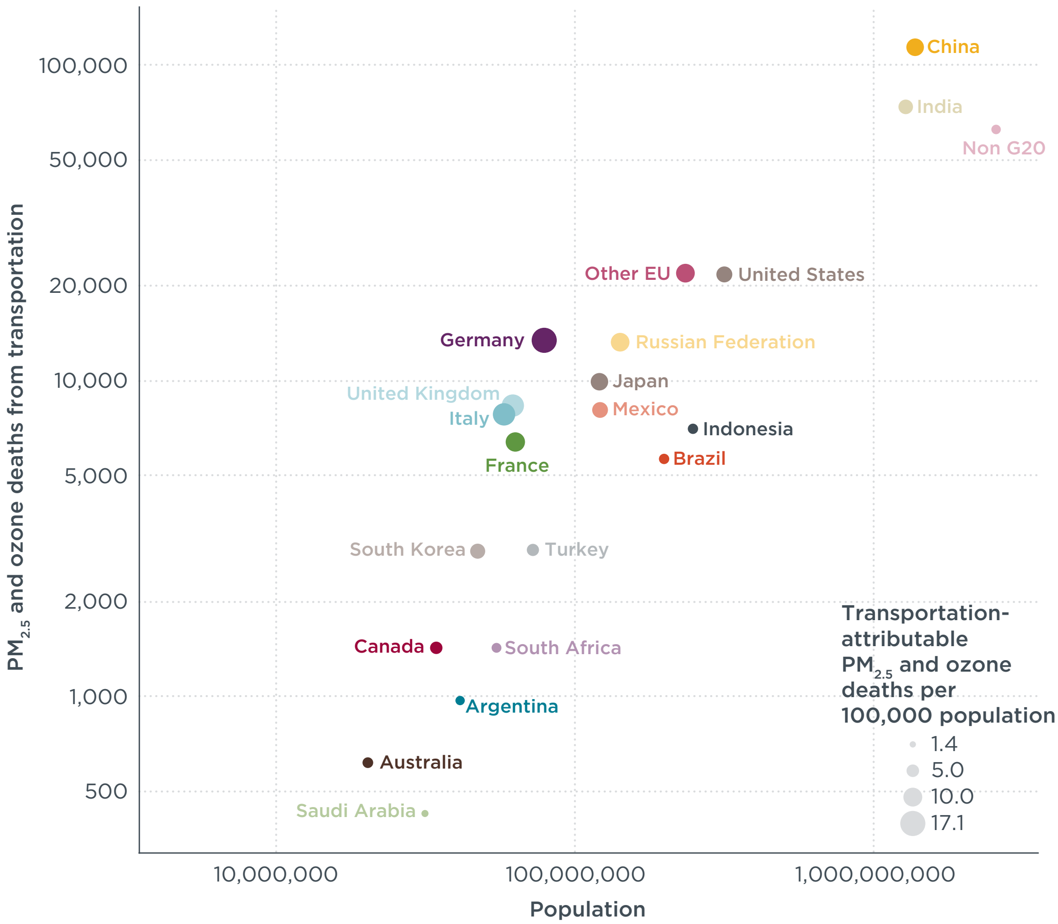

Figure 8. Transportation-attributable $PM_{2.5}$ and ozone deaths, associated mortality rates, and population in G20 economies in 2015.

3.5. IMPACTS BY TRADE BLOC

We further assessed this spatial heterogeneity in transportation emissions impacts by trade bloc, as these groups of countries with formal trade agreements have opportunities for coordinated action to harmonize vehicle emission standards, fuel quality standards, and other policies that affect vehicle emissions. Table A1 lists the identified trade bloc for each country (labels starting with "other" indicate the absence of an identified trade bloc). Considering $PM_{2.5}$ and ozone together, the trade bloc with the highest TAF consisted of the European Union and European Free Trade Association (EU and EFTA) which had 11.9 deaths per 100,000 people and a TAF of 25%. That was followed by the North American Free Trade Agreement (NAFTA), with 6.7 deaths per 100,000 people and a TAF of 20%, and Japan, with 8.3 deaths per 100,000 people and a TAF of 19%. $PM_{2.5}$ and ozone TAFs exhibited different spatial patterns globally (see Figure 9). $PM_{2.5}$ TAFs were highest in the EU and EFTA, at 27%, followed by 22%

for NAFTA and 21% for Japan. Ozone TAFs were highest in the Andean Community, at 27%, which was followed by the Association of Southeast Asian Nations (ASEAN) with 20%, the Gulf Cooperation Council (GCC) at 17%, the Caribbean Community (CARICOM) at 16%, the Southern Common Market (MERCOSUR) at 16%, and Central American Integration System (SICA) with an ozone TAF of 16%.

Figure 9. Globally and for each trade bloc, transportation-attributable fractions (TAF) of total (a) $PM_{2.5}$ and (b) ozone deaths in 2015, broken out by subsector. AMU = Arab Maghreb Union (North Africa); ASEAN = Association of Southeast Asian Nations; CARICOM = Caribbean Community; CEMAC = Central African Economic and Monetary Community; CIS = Commonwealth of Independent States; EAC = East African Community; ECOWAS = Economic Community of West African States; EU & EFTA = European Union and European Free Trade Association; GCC = Gulf Cooperation Council; MERCOSUR = Southern Common Market (South America); NAFTA = North American Free Trade Agreement; SAARC = South Asian Association for Regional Cooperation; SADC = Southern African Development Community; SICA = Central American Integration System.

Globally, and in each trade bloc, diesel emissions were the dominant contributor to total transportation-attributable health impacts in 2015 (see Figure 10). In total, emissions from on-road diesel vehicles, international shipping, and non-road mobile sources including agricultural and construction equipment contribute 82% to the total TAF globally. On-road diesels are the largest contributor to transportation-attributable $PM_{2.5}$ and ozone burdens in nearly all trade blocs, with a global subsector-specific TAF of 5% (of $PM_{2.5}$ and ozone burdens from all sources) in 2015. The on-road diesel vehicle TAF is by far the highest in the EU and EFTA trade bloc, representing 15% of the combined $PM_{2.5}$ and ozone burden from all sources. On-road non-diesel vehicles, international shipping, and non-road mobile sources each have global subsector-specific TAFs of 2%. On-road non-diesel vehicles, which are predominantly gasoline, contribute more to the total transportation-attributable ozone burden compared with $PM_{2.5}$ for all trade blocs, although this subsector also contributes substantially to the $PM_{2.5}$ TAF in the NAFTA trade bloc, reflecting the dominance of gasoline vehicles in the U.S. passenger vehicle fleet (see Figure 9). Shipping and non-road mobile sources are relatively minor contributors to ozone TAFs but make up more than half the total combined $PM_{2.5}$ and ozone TAF in Australia, Japan, and CARICOM region, and more than one-third of the total TAF globally. On-road diesels also contribute more than any other subsector to BC concentrations and TAFs in major vehicle markets (see Figure 7).

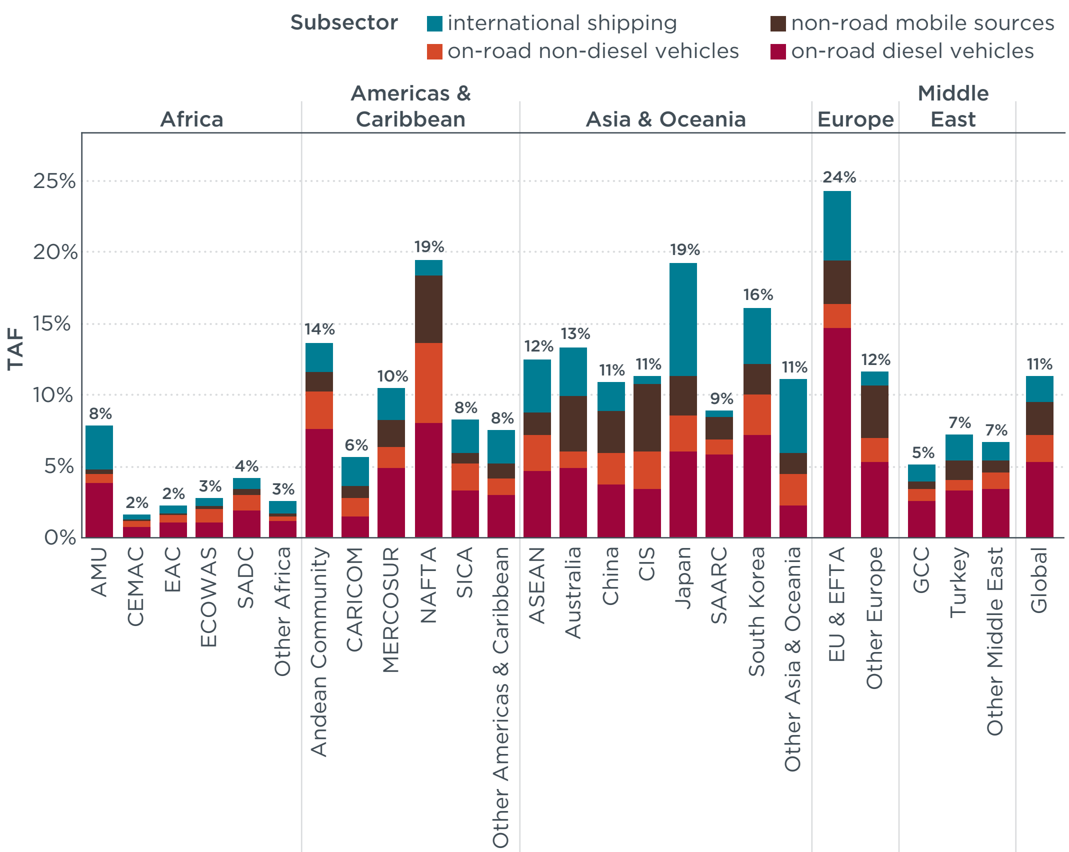

Figure 10. Globally and for each trade bloc, transportation-attributable fractions (TAF) of combined $PM_{2.5}$ and ozone deaths in 2015, broken out by subsector. AMU = Arab Maghreb Union (North Africa); ASEAN = Association of Southeast Asian Nations; CARICOM = Caribbean Community; CEMAC = Central African Economic and Monetary Community; CIS = Commonwealth of Independent States; EAC = East African Community; ECOWAS = Economic Community of West African States; EU & EFTA = European Union and European Free Trade Association; GCC = Gulf Cooperation Council; MERCOSUR = Southern Common Market (South America); NAFTA = North American Free Trade Agreement; SAARC = South Asian Association for Regional Cooperation; SADC = Southern African Development Community; SICA = Central American Integration System.

Consistent with our findings for the top four vehicle markets, more than three-quarters of the global transportation-attributable health impacts in 2015 are estimated to have occurred in four trade blocs as shown in Figure 11: China, including Hong Kong (117,000 premature deaths, 30% of the global total); South Asian Association for Regional Cooperation (SAARC), which includes India (86,000, 22%); EU and EFTA (58,000, 15%); and NAFTA (31,000, 8%). The large numbers of estimated transportation-attributable pollution deaths in these trade blocs reflect large populations and emissions in these regions. Normalizing by population, transportation-attributable deaths per 100,000 people were still in the mid- to high range of all trade blocs examined (8.4 for China, 5.0 for SAARC, and 6.7 for NAFTA). The importance of transportation-related health impacts in China and SAARC are masked by the TAF metric because large emissions from non-transportation emission sources result in relatively low TAFs in these regions. For SAARC, the global share of transportation-attributable air pollution deaths in this trade bloc is amplified compared to its share of transportation emissions, largely due to a combination of large population and high baseline disease rates, particularly for respiratory disease affected by ozone (see Figure 12).

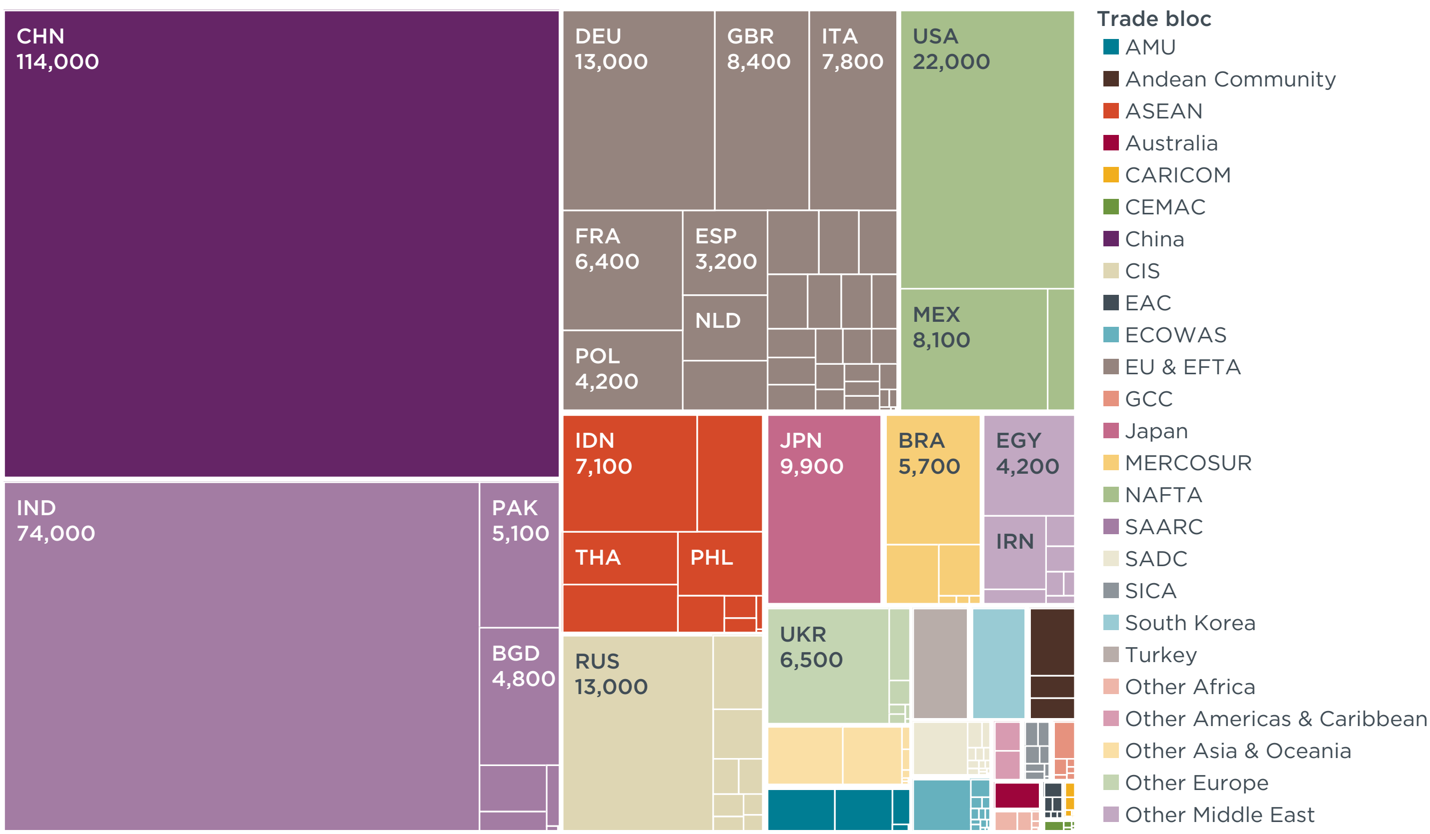

Figure 11. National total $PM_{2.5}$ and ozone mortality that is attributable to transportation emissions in 2015 in major trade blocs globally, using central relative risk estimates. AMU = Arab Maghreb Union (North Africa); ASEAN = Association of Southeast Asian Nations; CARICOM = Caribbean Community; CEMAC = Central African Economic and Monetary Community; CIS = Commonwealth of Independent States; EAC = East African Community; ECOWAS = Economic Community of West African States; EU & EFTA = European Union and European Free Trade Association; GCC = Gulf Cooperation Council; MERCOSUR = Southern Common Market (South America); NAFTA = North American Free Trade Agreement; SAARC = South Asian Association for Regional Cooperation; SADC = Southern African Development Community; SICA = Central American Integration System.

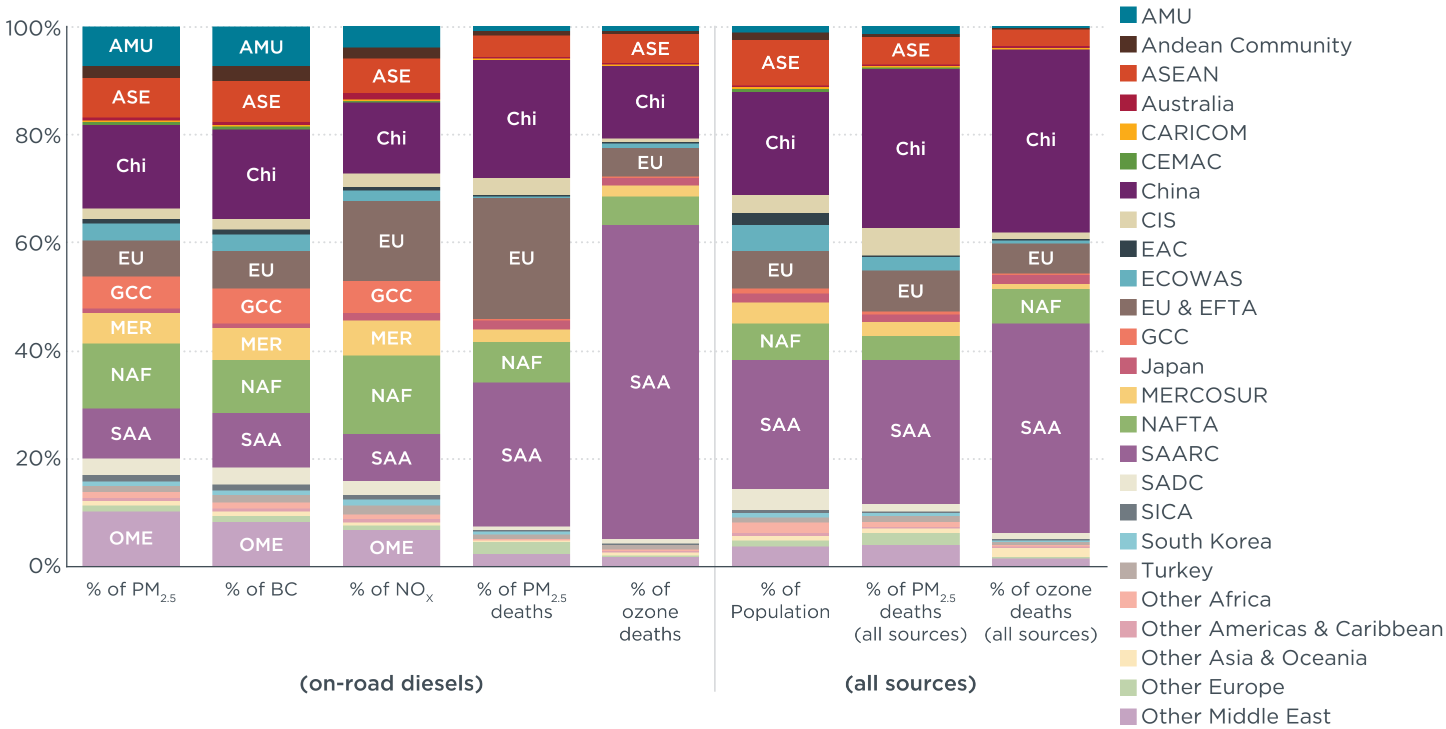

Figure 12. Share of global on-road diesel vehicle emissions and associated $PM_{2.5}$ and ozone deaths, compared with share of population and total $PM_{2.5}$ and ozone disease burdens, by trade bloc in 2015.

Figure 13 compares population size with the absolute number of transportation-attributable $PM_{2.5}$ and ozone deaths by trade bloc in 2015. These estimates are shown on a logarithmic scale to account for the substantial differences in population size among trade blocs and individual countries. The size of each point in Figure 13 corresponds to the transportation-attributable mortality rate per 100,000 population. In 2015, EU and EFTA countries along with those accounted for as "Other Europe" (which includes Ukraine) had the highest transportation-attributable mortality rates, at approximately 12 deaths per 100,000 population. These elevated rates are likely attributable to a combination of factors, including high levels of transportation activity; high levels of LDV dieselization; and relatively high baseline incidence rates for diseases that are affected by air pollution. In contrast, trade blocs in sub-Saharan Africa, South America, Central America, the Caribbean, and the Middle East had comparatively low transportation-attributable mortality rates. Considering the increases in motorization, freight activity, urbanization, total population, and baseline incidence rates of diseases affected by air pollution that are projected in many of these regions, coupled with the time it takes to turn over the entire existing vehicle fleet, we suggest that these regions take appropriate action to control emissions from new vehicles before health damages intensify (i.e., increased transportation-attributable mortality rates, absolute burden, and associated welfare loss).

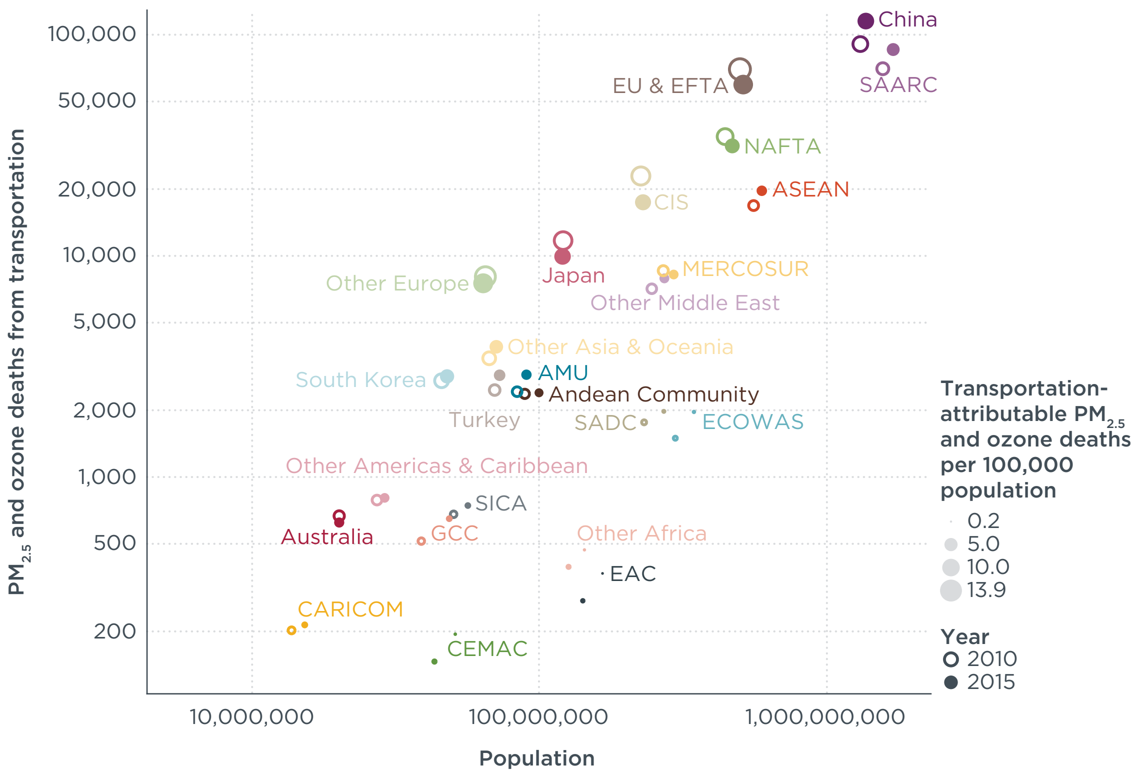

Figure 13. Transportation-attributable $PM_{2.5}$ and ozone deaths, associated mortality rates, and population by trade bloc in 2015.

3.6. IMPACTS IN URBAN AREAS

Because transportation emissions and exposure tend to be co-located in urban areas, we estimated the transportation-attributable air pollution deaths and TAFs for 100 major urban areas worldwide (see Figure 14). The urban areas with the highest number of transportation-attributable air pollution deaths are a combination of those with the largest populations and transportation emissions. The top 10 for 2015, in order from the highest number, are Guangzhou, Tokyo, Shanghai, Mexico City, Cairo, New Delhi, Moscow, Beijing, London, and Los Angeles. In contrast, when normalized by population, the urban areas with the highest number of transportation-attributable air pollution deaths per 100,000 people in 2015 are mainly in Europe. The top 10 urban areas for 2015 when normalized by population, again in order beginning with the highest, are Milan, Turin, Stuttgart, Kiev, Cologne, Haarlem, Berlin, Rotterdam, London, and Leeds. The urban areas with the highest TAFs are also mainly in Europe. Beginning with the highest, they are, for 2015, Milan, Rotterdam, Turin, Stuttgart, Mexico City, Leeds, Manchester, London, Paris, and Cologne. Although urban areas in South and East Asia frequently have large numbers of transportation-attributable deaths, they tend to have lower TAFs because of the high overall pollution concentrations from other emission sources, such as coal-fired electric generating units and residential solid fuel combustion for cooking and heating. The only urban areas ranking in the top 20 for both transportation-attributable deaths and TAF in 2015 are Mexico City, London, Paris, and Cologne. In some cases, these cities contribute a large share of the national or regional transportation-attributable deaths (see Table A2).

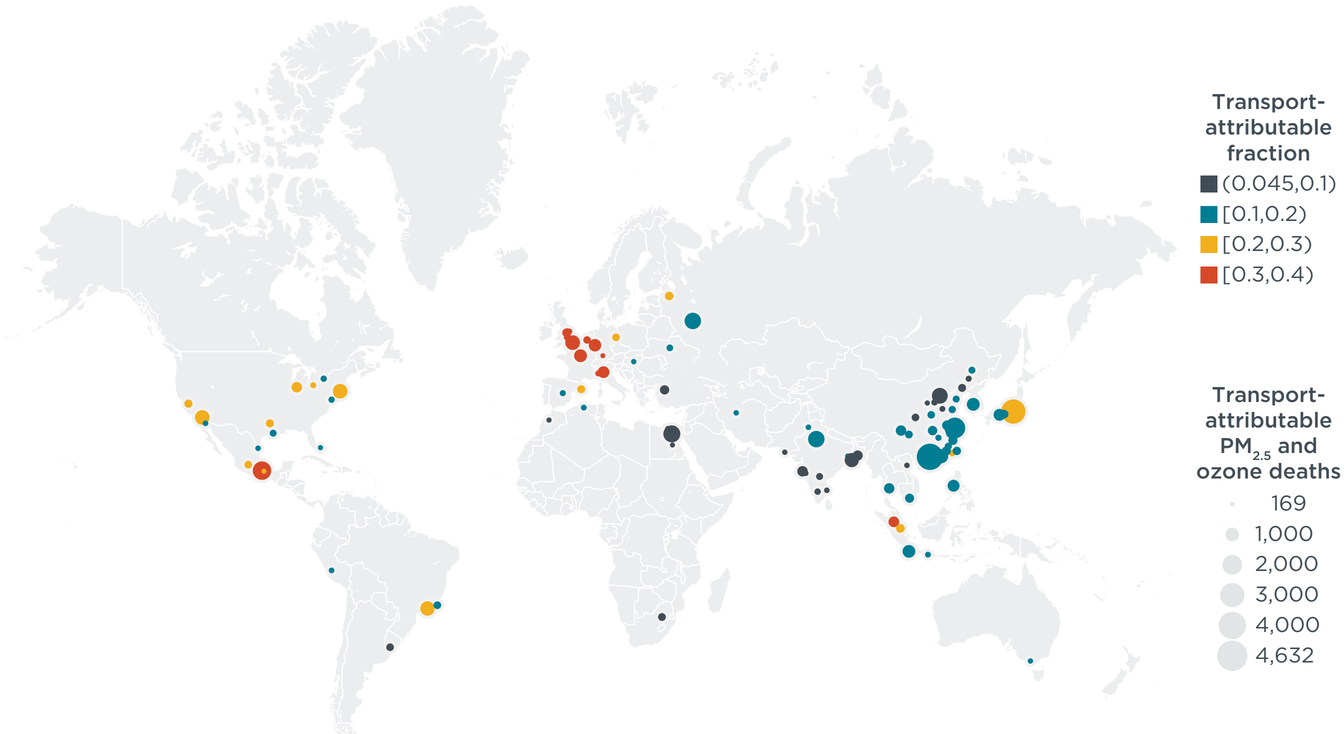

Figure 14. Total number of transportation-attributable $PM_{2.5}$ and ozone deaths in 2015 by urban area. Bubble size indicates total number of transportation-attributable $PM_{2.5}$ and ozone deaths using central relative risk estimates. Bubble color indicates transportation-attributable fraction (TAF) of total $PM_{2.5}$ and ozone deaths.

4. DISCUSSION

We estimated that tailpipe emissions from the transportation sector contributed 11.7% of global $PM_{2.5}$ and ozone deaths in 2010 and 11.4% in 2015. Vehicle tailpipe emissions were associated with 361,000 (95% CI, 258,000–462,000) and 385,000 (95% CI, 274,000–493,000) premature deaths from $PM_{2.5}$ and ozone globally in 2010 and 2015, translating into 5.43 deaths per 100,000 people in 2010 and 5.38 deaths per 100,000 people in 2015. The global health damages from transportation tailpipe emissions are valued at approximately $1 trillion in 2015 (2015 US$), corresponding to 17% of the total damages from ambient $PM_{2.5}$- and ozone-related deaths. We found substantial heterogeneity in transportation-attributable health impacts around the world. For example, although transportation-attributable deaths were greatest in China and SAARC (which includes India) trade blocs, the transportation-attributable deaths per 100,000 people were highest in the EU, EFTA, and other European countries. In addition, the transportation-attributable fraction of total $PM_{2.5}$- and ozone-related deaths was highest in the EU and EFTA and NAFTA trade blocs. Among transportation subsectors, on-road diesels contributed most to the health burden from transportation tailpipe emissions in nearly all trade blocs, for both $PM_{2.5}$ and ozone, although other subsectors also contributed substantially. Specifically, on-road non-diesel vehicles substantially increased ozone mortality, whereas shipping and non-road mobile sources mainly increased $PM_{2.5}$ mortality.

4.1. COMPARISON WITH OTHER STUDIES

Our estimate of 361,000 (95% CI, 258,000–462,000) $PM_{2.5}$ and ozone deaths in 2010 and 385,000 (95% CI, 274,000–493,000) in 2015 from transportation sector emissions are within the range of estimates reported by previous studies, although toward the upper end (see Table A3). Chambliss et al. (2014) estimated 242,000 $PM_{2.5}$ deaths in 2005; Silva et al. (2016) estimated 376,000 $PM_{2.5}$ and ozone deaths from transportation in 2005; and Lelieveld et al. (2015) estimated 165,000 $PM_{2.5}$ and ozone deaths in 2010. Our estimated global TAF (11.4%) is also within the range of previous studies—Chambliss et al. (2014) estimated 8.5% in 2005, Lelieveld et al. (2015) estimated 5% for land transportation in 2010, and Silva et al. (2016) estimated 13.8% (of $PM_{2.5}$ and ozone mortality from anthropogenic emissions only) in 2005. In addition, Weagle et al. (2018) estimated that transportation contributed 8.6% of total $PM_{2.5}$ concentrations in 2014. Our results also are consistent with these previous studies in showing that TAFs in Europe and North America are substantially higher than the global average. Our TAFs for $PM_{2.5}$ are similar to previous estimates examining contributions of major emission sources in China and India, although slightly lower for China (12% vs. 15%) and higher for India (8% vs. 6%) (GBD MAPS Working Group 2016, 2018).

Differences in results are driven by several factors: different emissions inventories from which to derive transportation-related emissions; different subcategories of the transportation sector, which are not always clearly specified in the methods descriptions of previous studies; different sources of baseline disease rates, most often using IHME GBD estimates but from different GBD iterations; different concentration-response functions, again most often using IHME GBD IER curves but from different GBD iterations; different spatial resolutions; and different approaches for isolating the influence of transportation emissions, although all but one assumed, as we did, that the transportation-attributable burden is proportional to the transportation-attributable fraction of concentrations. The multiple methodological differences occurring in concert, combined with nonspecific information about some (e.g., which subcategories of the transportation sector are included) limit our ability to tease out which driving factors may contribute most to influencing mismatches between our results and those in the literature. We note that we use a finer spatial resolution of 0.1°x0.1°, which also was used by Weagle et al. (2018), and the most up-to-date disease rates and concentration-response functions from the GBD 2017 study. Thus, our results may be viewed as an updated and refined estimate compared with these previous studies. However, uncertainties remain, and our estimates should continue to be updated as input data and methods advance, and as transportation policies and emissions evolve.

4.2. TRANSPORTATION-ATTRIBUTABLE HEALTH IMPACTS ARE LIKELY UNDERESTIMATED

These estimated health impacts associated with the transportation sector are likely underestimated for several reasons. First, air pollution from transportation tailpipe emissions is just one component of the public health impacts of the transportation sector; we have excluded other important health impacts of the sector, including from noise, physical activity effects, road injuries, resuspension of road dust, release of particles from brake and tire wear, evaporative emissions, and fuel life-cycle emissions. Health impacts from tailpipe emissions specifically may also be underestimated because we considered only the health impacts from $PM_{2.5}$ and ozone, and excluded other transportation-related pollutants, such as nitrogen dioxide (NO_2), which is associated with asthma incidence among children worldwide (Achakulwisut, Brauer, Hystad, Anenberg, 2019; Anenberg et al., 2018; Khreis et al., 2017) and asthma emergency department visits (Anenberg et al., 2018; Orellano, Quaranta, Reynoso, Balbi, & Vasquez, 2017; Zhang, Li, Tian, Guo, & Pan, 2016; Zheng et al., 2015). Transportation is the largest source of NO_2 concentrations, and health effects may be particularly pronounced in cities, which can have very high NO_2 concentrations (Achakulwisut et al., 2019; Khreis, de Hoogh, & Nieuwenhuijsen, 2018). Finally, our $PM_{2.5}$ health risk modeling may underestimate impacts because recent evidence indicates that the health response to air pollution could continue relatively linearly at extremely high concentrations rather than flattening out (Burnett et al., 2018). For example, our estimates of global transportation-attributable mortality from $PM_{2.5}$ would approximately double if we used an updated $PM_{2.5}$ concentration-response curve for mortality that includes only epidemiological studies from ambient air pollution (as opposed to ambient air pollution, household air pollution, environmental tobacco smoke, and active smoking) and all non-accidental mortality (as opposed to only stroke, ischemic heart disease, lung cancer, chronic obstructive pulmonary disease, lower respiratory infections, and diabetes) (Burnett et al., 2018). In addition, recent evidence indicates that considering only the six diseases currently included in the GBD 2017 study excludes other diseases that also may be associated with air pollution, including asthma (Anenberg et al., 2018), chronic kidney disease (Bragg-Gresham et al., 2018), preterm birth and other birth outcomes (Malley, Kuylenstierna et al., 2017), and cognitive decline (Zhang, Chen, & Zhang, 2018).

Although we used established methods that are commonly applied to estimate health impacts of air pollution on a global scale, our results are subject to several uncertainties that could influence our results. In any assessment of air pollution health impacts, there are uncertainties at each analytical step, including characterizing emissions, pollutant concentrations, and associated health impacts. The most influential uncertainty is likely from the choice of the health impact function. There are also important uncertainties in the magnitude and spatial distribution of transportation emissions, the ability of chemical transport modeling to capture atmospheric chemistry processes occurring at urban scales, and the representativeness of the epidemiological concentration-response functions for all pollution mixtures and all populations globally, among others. The direction in which these uncertainties would influence results is unknown.

4.3. POLICY RECOMMENDATIONS

Despite recent progress in countries adopting more stringent vehicle emission standards, transportation emissions remain a major contributor to ambient air pollution and its associated health impacts at the global, national, and urban scales. At the global scale, comparison of estimates for 2005, 2010, and 2015 indicates that the health impacts of transportation emissions are increasing rather than decreasing. Transportation emissions in the leading markets—California, the United States, and the EU—have been regulated for nearly 50 years. However, it is only relatively recently that world-class emissions standards have been applied to nearly all types of vehicles, equipment, and fuels and have achieved reductions in harmful pollutants to such an extent, on the order of 99% in the best cases (Miller & Façanha 2014). DPF-forcing standards were first applied to all new HDVs in the United States in 2007 and in the EU in 2014. U.S. 2010 and Euro VI standards limited real-world NO_x emissions from new HDVs by 80% to 90% compared with previous stages (Anenberg et al., 2017). Our analysis shows that although transportation-attributable mortality in the EU and the United States declined 14% and 16%, respectively, from 2010 to 2015, the EU and United States still had the third and fourth largest health burdens from transportation emissions in 2015.

The dominant contribution of diesel vehicles and engines, including non-road mobile sources, to transportation-attributable health impacts highlights the importance of further developing emissions standards (e.g., low-NO_x standards in the United States); expanding the global uptake of world-class standards; strengthening compliance and enforcement practices; and accelerating fleet turnover to remove vehicles and equipment with older technology. The substantial and growing global burden of transportation-attributable health impacts highlights the importance of addressing each of these objectives in short order. In recognition of varying regional circumstances, achieving these objectives will require collaboration among governments. Of particular importance is the continued collaboration of G20 economies, which collectively accounted for 84% of global transportation-attributable mortality in 2015. G20 economies and other regional leaders likewise have an important role in promoting regional cooperation, for example, by promoting the alignment of vehicle and fuel standards among countries within trade blocs. ASEAN and ECOWAS are two examples of such ongoing collaborations.

In Europe, the emissions reductions expected from progressively tightening NO_x emission limits for diesel LDVs failed to materialize due to gaps in compliance and enforcement and widespread cheating. If the affected vehicles remain on the roads—whether in Europe or after being exported overseas—the associated health impacts are likely to persist. Other markets can learn from the experiences of the EU and India, which after initially promoting dieselization of their LDV fleets have since reversed those policies in an effort to stem the attendant health impacts. Even with declining LDV diesel market shares, it will take several years for the emissions reductions to materialize for the in-use fleets in these regions. We recommend that vehicle-importing countries—particularly those that allow second-hand vehicle imports—update their regulatory framework and taxation policies to ensure they do not become a dumping ground for diesel LDVs, whether older or new, with high tailpipe $PM_{2.5}$ without DPFs and NO_x emissions, which is common even among Euro 6 diesel LDVs.

China and India have made major advances in regulating emissions from transportation, with both markets adopting world-class standards for LDVs and HDVs and advancing similar policies for non-road mobile sources. Other major markets such as Mexico and Brazil have likewise adopted world-class standards for HDVs. However, most of these recently adopted standards will apply to new vehicles entering these markets between 2020 and 2023. The experiences of the United States and the EU demonstrate the substantial time lag between implementation of new vehicle standards and the realization of their full benefits for the in-use vehicle fleet, due to the long lifetimes of vehicles and equipment, and public health due to the contribution of air pollution to chronic diseases including cardiovascular and respiratory diseases. State- or national-level vehicle retrofit or scrappage programs have been applied in some regions (e.g., California's in-use HDV program and China's yellow-label vehicle program); however, such programs have yet to be implemented consistently across regions as the natural complement to world-class standards.

As concentrated centers of transportation activity, emissions, and associated health impacts, major urban areas are well-positioned to adopt policies that accelerate the introduction of low- and zero-emission vehicles, encourage fleet turnover, and restrict activity of vehicles and equipment with comparatively high emissions. However, because urban areas often depend on national governments to set and enforce standards for new vehicles and equipment and transportation fuels, there remains a need for constructive dialogue among local, state/provincial, and national governments to limit the health impacts of transportation emissions.

International shipping emissions contributed 15.4% of global transportation-attributable mortalities in 2015, reflecting an 8.5% increase in premature deaths from shipping emissions between 2010 and 2015. BC emissions from shipping are emitted primarily near coastal areas and in 2015, 89% of BC emissions from international shipping were from vessels whose main fuel type is residual fuel (Comer et al., 2017). Comer et al. (2017) recommended three policies to reduce BC emissions from global shipping: banning the use of residual fuels, establishing a BC emissions standard for ships, and incorporating BC emissions reductions into greenhouse gas reduction strategies for the global shipping sector. Our estimates of the health impacts associated with global shipping emissions—valued at $150 billion (2015 US$) in 2015—highlight the importance of considering such new policies to control emissions of BC from shipping in addition to further reducing NO_x and sulfur oxide (SO_x) emissions.

Because the time frame of our analysis is limited to 2010 and 2015, it does not capture the projected changes in transportation emissions impacts from factors in more recent years, including the adoption and continued implementation of world-class standards, declining diesel market shares among LDVs in Europe and India, and growing uptake of vehicles with zero-tailpipe emissions. These developments will take time to manifest into changes in the vehicle fleet, emissions, pollutant concentrations, and health impacts. Future work therefore can continue to monitor on an ongoing basis the changes in health impacts associated with transportation-sector emissions associated with these policies. Future work also may seek to quantify the future health benefits of recently adopted policies, as well as the health damages from expected growth in transportation activity in India, Africa, and other rapidly developing parts of the world. Future work may also address the additional co-benefits, such as from more physical activity and fewer greenhouse gas emissions, from increased access to active transportation, and from public transportation. Also of interest are the projected impacts of growing and aging populations, urbanization, and other factors related to the global and regional incidence of transportation-attributable mortalities in conjunction with realistic policy scenarios. Our results point to the need for reducing emissions from the transportation sector to be a central element of management plans aimed at reducing ambient air pollution and its burden on public health.

APPENDIX

EMISSIONS UNCERTAINTIES

The true value of transport sector emissions is highly uncertain. Uncertainty for sectoral emissions tends to be substantially higher than for national emissions totals (Saikawa et al., 2017). In an investigation of emissions in Asia, Streets et al. (2003) found the highest uncertainty in BC and OC emissions (>300%), moderate uncertainty in NO_x emissions (±37%), and the lowest uncertainty in SO_x emissions (±16%) (Streets et al., 2003). In a Monte Carlo analysis of emissions in India, Saikawa et al. (2017) found high uncertainty in transport of PM_{10} and NO_X (±120%), and moderate uncertainty in transport of SO_x (±66%). Emission factors are typically positively skewed unless uncertainties are low (less than or equal to 30%) (Frey 2007), such that the true confidence intervals are not symmetric around the mean. For example, Streets et al. suggests interpreting an uncertainty margin of ±400% as within a factor of five (-80%, +400%) (Streets et al. 2003).

We attempt to account for uncertainty in country-specific diesel road transport emissions by comparing fleet average emission factors of countries at similar stages of vehicle emission control deployment. Figure A1 shows average estimated fuel-specific BC and NO_x emission factors for diesel road transport in 2015. Countries are grouped according to their latest emissions standard implemented for new heavy-duty trucks as of 2015. Variation among countries in the same grouping depends on the distribution of activity and energy use among vehicle types (e.g., diesel cars, light commercial vehicles, trucks, and buses), the share of the in-use vehicle fleet by age and technology (e.g., Euro I, Euro II, etc), and the assumed share of high emitting vehicles, which is to say those with malfunctioning emission controls due to poor design, inadequate maintenance, or tampering. We estimate uncertainty in diesel road transport emissions by assuming the true fleet average emission factor of each country falls somewhere within the distribution of other countries at a similar stage of new truck emissions controls. For example, the fleet average BC emission factor of Afghanistan has a central estimate of 75.5 mg per megajoule and a range of (-50%, +80%). These uncertainty estimates do not attempt to account for potential errors in fuel sulfur input data, which could affect SO_x emissions by an order of magnitude or greater. Nor do we attempt to account for uncertainty regarding total energy consumption allocated to diesel road transport (Garg et al., 2006), although the likely range of uncertainty is much smaller than the uncertainty in fleet average emission factors.

Considering diesel road transport accounts for roughly two-thirds of global transport BC emissions (Klimont et al. 2017), propagating the estimated uncertainty in diesel road transport emissions through to health impacts could provide an instructive example of the sensitivity of health impacts to uncertainty in transport emissions. It should be noted, however, that we do not attempt to account for all potential sources of uncertainty in transport emissions, nor do we estimate emissions uncertainty for all transport subsectors.

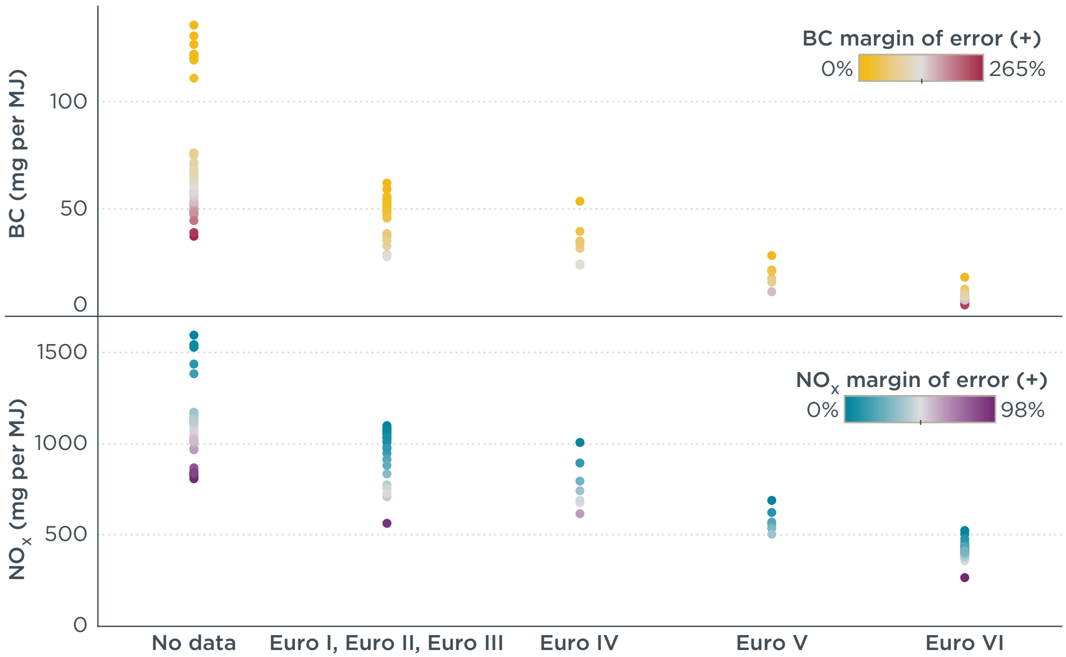

Figure A1. Average estimated fuel-specific BC and NO_x emission factors for diesel road transport in 2015. Countries are grouped according to their latest emissions standard implemented for new heavy-duty trucks as of 2015.

EMISSIONS SENSITIVITY ANALYSIS

As a sensitivity analysis, we assessed the influence of sensitivity of concentrations to on-road diesel emissions globally. Specifically, we examined the effect of 10% higher and lower estimates of on-road diesel BC and NO_x emissions around the 2010 base case. The grid cell maximum change in annual average $PM_{2.5}$ concentration between the +10% sensitivity case and the 2010 base case is 3.8 $\mu g/m^3$, whereas the grid cell maximum annual average $PM_{2.5}$ concentration from on-road diesels is 9.8 $\mu g/m^3$. Using the 10% lower and upper sensitivity GEOS-Chem simulations results in transportation-attributable estimates that are 9% lower and 16% higher than the base case for total transportation-attributable health impacts, and 20% lower and 37% higher for health impacts from on-road diesels specifically. The magnitude of these uncertainties in on-road diesel emissions on simulated concentrations and transportation-attributable health impacts is non-negligible, but less than the magnitude of overall base case transportation and on-road diesel concentrations and health impacts.

Table A1. Trade bloc assignment for each country.

Country	Trade Bloc
Afghanistan	SAARC
Albania	Other Europe
Algeria	AMU
American Samoa	Other
Angola	SADC
Antigua and Barbuda	CARICOM
Argentina	MERCOSUR
Armenia	CIS
Aruba	Other
Australia	Australia
Austria	EU & EFTA
Azerbaijan	CIS
Bahamas	CARICOM
Bahrain	GCC
Bangladesh	SAARC
Barbados	CARICOM
Belarus	CIS
Belgium	EU & EFTA
Belize	SICA
Benin	ECOWAS
Bermuda	Other
Bhutan	SAARC
Bolivia	MERCOSUR
Bosnia and Herzegovina	EU & EFTA
Botswana	SADC
Brazil	MERCOSUR
Brunei	ASEAN
Bulgaria	EU & EFTA
Burkina Faso	ECOWAS
Burundi	EAC
Côte d'Ivoire	ECOWAS
Cabo Verde	ECOWAS
Cambodia	ASEAN
Cameroon	CEMAC
Canada	NAFTA
Cayman Islands	Other
Central African Republic	CEMAC
Chad	CEMAC
Chile	Other Latin America
China	China
Colombia	Andean Community
Comoros	Other
Congo	CEMAC
Costa Rica	SICA
Croatia	EU & EFTA
Cuba	Other Latin America
Cyprus	EU & EFTA
Czech Republic	EU & EFTA

Country	Trade Bloc
Democratic People's Republic of Korea	Other Asia Pacific
Democratic Republic of the Congo	SADC
Denmark	EU & EFTA
Djibouti	Other
Dominica	CARICOM
Dominican Republic	SICA
Ecuador	Andean Community
Egypt	Other Middle East
El Salvador	SICA
Equatorial Guinea	CEMAC
Eritrea	Other Africa
Estonia	EU & EFTA
Ethiopia	Other Africa
Falkland Islands	Other
Fiji	Other
Finland	EU & EFTA
France	EU & EFTA
French Guiana	Other
French Polynesia	Other
Gabon	CEMAC
Gambia	ECOWAS
Georgia	Other Europe
Germany	EU & EFTA
Ghana	ECOWAS
Greece	EU & EFTA
Grenada	CARICOM
Guadeloupe	Other
Guam	Other
Guatemala	SICA
Guinea	ECOWAS
Guinea-Bissau	ECOWAS
Guyana	CARICOM
Haiti	CARICOM
Honduras	SICA
Hungary	EU & EFTA
Iceland	EU & EFTA
India	SAARC
Indonesia	ASEAN
Iran	Other Middle East
Iraq	Other Middle East
Ireland	EU & EFTA
Israel	Other Middle East
Italy	EU & EFTA
Jamaica	CARICOM
Japan	Japan
Jordan	Other Middle East
Kazakhstan	CIS
Kenya	EAC
Kiribati	Other

Country	Trade Bloc
Kuwait	GCC
Kyrgyzstan	CIS
Lao People's Democratic Republic	ASEAN
Latvia	EU & EFTA
Lebanon	Other Middle East
Lesotho	SADC
Liberia	ECOWAS
Libya	AMU
Liechtenstein	EU & EFTA
Lithuania	EU & EFTA
Luxembourg	EU & EFTA
Macedonia (FYROM)	Other Europe
Madagascar	SADC
Malawi	SADC
Malaysia	ASEAN
Maldives	SAARC
Mali	ECOWAS
Malta	EU & EFTA
Martinique	Other
Mauritania	AMU
Mauritius	SADC
Mayotte	Other
Mexico	NAFTA
Micronesia	Other
Mongolia	Other Asia Pacific
Montenegro	Other Europe
Montserrat	CARICOM
Morocco	AMU
Mozambique	SADC
Myanmar	ASEAN
Namibia	SADC
Nepal	SAARC
Netherlands	EU & EFTA
Netherlands	EU & EFTA
New Caledonia	Other
New Zealand	Other Asia Pacific
Nicaragua	SICA
Niger	ECOWAS
Nigeria	ECOWAS
Norway	EU & EFTA
Occupied Palestinian Territory	Other
Oman	GCC
Pakistan	SAARC
Palau	Other
Panama	SICA
Papua New Guinea	Other
Paraguay	MERCOSUR
Peru	Andean Community
Philippines	ASEAN
Poland	EU & EFTA
Portugal	EU & EFTA

Country	Trade Bloc
Qatar	GCC
Republic of Korea	Republic of Korea
Republic of Moldova	CIS
Reunion	Other
Romania	Other
Russian Federation	CIS
Rwanda	EAC
Saint Kitts and Nevis	CARICOM
Saint Lucia	CARICOM
Saint Vincent and Grenadines	CARICOM
Samoa	Other
Sao Tome and Principe	Other
Saudi Arabia	GCC
Senegal	ECOWAS
Serbia	Other
Seychelles	SADC
Sierra Leone	ECOWAS
Singapore	ASEAN
Slovakia	EU & EFTA
Slovenia	EU & EFTA
Solomon Islands	Other
Somalia	Other
South Africa	SADC
South Sudan	EAC
Spain	EU & EFTA
Sri Lanka	SAARC
Sudan	Other Africa
Suriname	CARICOM
Swaziland	SADC
Sweden	EU & EFTA
Switzerland	EU & EFTA
Syria	Other Middle East
Tajikistan	CIS
Thailand	ASEAN
Timor-Leste	Other
Togo	ECOWAS
Tonga	Other
Trinidad and Tobago	CARICOM
Tunisia	AMU
Turkey	Turkey
Turkmenistan	Other Asia Pacific
Uganda	EAC
Ukraine	Other Europe
United Arab Emirates	GCC
United Kingdom	EU & EFTA
United Republic of Tanzania	EAC
United States of America	NAFTA
Uruguay	MERCOSUR
Uzbekistan	CIS

Country	Trade Bloc
Vanuatu	Other
Venezuela	MERCOSUR
Viet Nam	ASEAN
Virgin Islands (British)	Other
Western Sahara	Other
Yemen	Other Middle East
Zambia	SADC
Zimbabwe	SADC

Table A2. Transportation-attributable (TA) deaths for 100 major urban areas in 2015, including the share of the region TA deaths, and the transport-attributable fraction of $PM_{2.5}$ and ozone deaths in 2015. The share of region TA deaths is likely highly dependent on the apportioning of population to the urban versus rural areas of the country. See Center for International Earth Science Information Network (2017) for more details.

Rank order of TA Deaths in 2015	City, Region	Share of Region TA Deaths (%)	Transport-Attributable Fraction (%)
1	Guangzhou, China	4.0%	17.6%
2	Tokyo, Japan	39.7%	25.2%
3	Shanghai, China	2.5%	15.0%
4	Mexico City, Mexico	28.1%	34.8%
5	Cairo, Egypt	45.8%	8.1%
6	New Delhi, India	2.5%	10.8%
7	Moscow, Russian Federation	13.7%	18.0%
8	Beijing, China	1.5%	8.9%
9	London, United Kingdom	17.4%	32.7%
10	Los Angeles, United States	6.6%	22.0%
11	New York, United States	6.5%	24.4%
12	Sao Paulo, Brazil	24.3%	22.6%
13	Shantou, China	1.2%	19.9%
14	Kolkata, India	1.8%	8.3%
15	Wuxi, China	1.0%	12.7%
16	Seoul, South Korea	38.3%	14.8%
17	Paris, France	17.2%	32.5%
18	Jakarta, Indonesia	15.4%	13.5%
19	Cologne, Germany	7.7%	32.1%
20	Osaka, Japan	9.6%	16.2%
21	Manila, Philippines	39.0%	15.1%
22	Milan, Italy	11.8%	39.4%
23	Hangzhou, China	0.8%	14.9%
24	Kuala Lumpur, Malaysia	30.3%	31.5%
25	Chicago, United States	3.4%	22.6%
26	Chengdu, China	0.6%	10.2%
27	Mumbai, India	1.0%	5.9%
28	Bangkok, Thailand	26.7%	14.7%
29	Tianjin, China	0.6%	9.3%
30	Dhaka, Bangladesh	14.3%	9.0%
31	Nanjing, China	0.6%	12.1%
32	Wuhan, China	0.5%	10.4%
33	Istanbul, Turkey	20.0%	8.8%
34	Wenzhou, China	0.5%	14.8%

Rank order of TA Deaths in 2015	City, Region	Share of Region TA Deaths (%)	Transport-Attributable Fraction (%)
35	Ho Chi Minh City, Vietnam	16.0%	16.1%
36	Quanzhou, China	0.5%	19.7%
37	Manchester, United Kingdom	6.4%	34.5%
38	Singapore, Singapore	100.0%	24.2%
39	Nagoya, Japan	5.0%	16.3%
40	St. Petersburg, Russian Federation	3.6%	21.0%
42	Taipei, Taiwan	22.8%	11.5%
43	San Francisco, United States	2.0%	28.6%
44	Dallas, United States	2.0%	20.3%
45	Shenyeng, China	0.4%	9.6%
46	Barcelona, Spain	13.3%	26.3%
47	Chongqing, China	0.4%	11.0%
48	Johannesburg, South Africa	29.1%	8.5%
49	Philadelphia, United States	1.9%	26.4%
50	Rio de Janeiro, Brazil	6.9%	19.3%
51	Guadalajara, Mexico	4.8%	27.5%
52	Buenos Aires, Argentina	40.4%	9.6%
53	Xian, China	0.3%	8.4%
54	Berlin, Germany	2.9%	26.1%
55	Zhengzhou, China	0.3%	10.1%
56	Rotterdam, Netherlands	15.0%	37.9%
57	Qingdao, China	0.3%	12.6%
58	Fuzhou, China	0.3%	18.3%
59	Dalian, China	0.3%	13.8%
60	Birmingham, United Kingdom	3.9%	31.4%
61	Harbin, China	0.3%	10.2%
62	Hyderabad, India	0.4%	7.1%
63	Houston, United States	1.5%	15.6%
64	Ningbo, China	0.3%	17.7%
65	Taichung, Taiwan	15.0%	21.9%
66	Kiev, Ukraine	4.7%	16.8%
67	Bangalore, India	0.4%	7.9%
68	Shijianzhuang, China	0.2%	8.6%
69	Toronto, Canada	19.0%	18.6%
70	Washington, D.C., United States	1.2%	19.2%
71	Nanchang, China	0.2%	11.8%
72	Changchun, China	0.2%	9.7%
73	Madrid, Spain	7.7%	16.4%
74	Detroit, United States	1.1%	20.1%
75	Turin, Italy	3.0%	37.5%
76	Jinan, China	0.2%	8.3%
77	Surabaya, Indonesia	3.3%	15.0%
78	Leeds, United Kingdom	2.8%	34.7%
79	Algiers, Algeria	16.5%	14.8%
80	Lima, Peru	47.4%	11.7%
81	San Diego, United States	1.0%	14.9%
82	Monterrey, Mexico	2.6%	15.9%
83	Chennai, India	0.3%	5.1%

Rank order of TA Deaths in 2015	City, Region	Share of Region TA Deaths (%)	Transport-Attributable Fraction (%)
84	Lahore, Pakistan	4.1%	10.6%
85	Hanoi, Vietnam	5.9%	9.5%
86	Pune, India	0.3%	6.8%
87	Asansol, India	0.3%	10.9%
88	Tehran, Iran	9.5%	10.8%
89	Karachi, Pakistan	3.9%	5.1%
90	Melbourne, Australia	31.2%	17.6%
91	Budapest, Hungary	17.4%	16.7%
92	Alexandria, Egypt	4.5%	5.9%
93	Miami, United States	0.8%	13.1%
94	Bandung, Indonesia	2.6%	13.3%
95	Casablanca, Morocco	15.7%	7.8%
96	Taiyuan, China	0.2%	6.5%
97	Sohag, Egypt	4.1%	4.5%
98	Jiaojing, China	0.2%	13.0%
99	Puebla, Mexico	2.1%	24.6%
100	Stuttgart, Germany	1.3%	35.8%

Table A3. Comparison of global results from this study with other estimates in the literature.

Study	Analysis year	Sector description	Methods	Result
This study	2010	Tailpipe emissions from on-road diesel, other on-road, shipping, non-road mobile sources	$PM_{2.5}$ RR: GBD 2017 IER Ozone RR: GBD 2017 Resolution: 0.1°x0.1° Emissions: ICCT (Miller & Jin, 2018), ECLIPSE (Klimont et al., 2017; Stohl et al., 2015)	Deaths: 361,000 (258,000–462,000) TAF: 11.7%
This study	2015	Tailpipe emissions from on-road diesel, other on-road, shipping, non-road mobile sources	Same as row 1	Deaths: 385,000 (274,000–493,000) TAF: 11.4%
Chambliss et al. (2014)	2005	all mobile equipment powered by gasoline and diesel engines such as on-road passenger vehicles and commercial trucks, rail transportation, off-road agricultural and construction equipment	$PM_{2.5}$ only: GBD2010 IER Resolution: 0.5° x 0.67° Emissions: Representative Concentration Pathway 8.5 (van Vuuren et al., 2011)	Deaths: 242,000 TAF: 8.5%
Lelieveld et al. (2015)	2010	Road and non-road transport on land	$PM_{2.5}$ RR: GBD2010 IER Ozone RR: Ostro (2004) Resolution: 1.1° x 1.1° Emissions: Emissions Database for Global Atmospheric Research (EDGAR)	Deaths: 165,000 TAF: 5%
Silva et al. (2016)	2005	Land transportation, shipping, and aviation	$PM_{2.5}$ RR: GBD2010 Ozone RR: Jerrett et al. (2009) Resolution: 0.5° x 0.67° Emissions: Representative Concentration Pathway 8.5 (van Vuuren et al., 2011)	Deaths: 376,000 TAF: 13.8% of anthropogenic $PM_{2.5}$- and ozone-related deaths
Weagle et al. (2018)	2014	Transportation	Concentration only Resolution: 0.1° x 0.1° Emissions: EDGAR v4.3 (Crippa et al., 2016), MIX (Li et al., 2017)	TAF: 8.6%

Note: RR = relative risk; IER = Integrated Exposure Response curve; ICCT = International Council on Clean Transportation; GBD = Global Burden of Disease; TAF = Transportation-attributable fraction.

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