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SPECIAL REPORT 20

BURDEN OF DISEASE ATTRIBUTABLE TO COAL-BURNING AND OTHER MAJOR SOURCES OF AIR POLLUTION IN CHINA

GBD MAPS Working Group

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GBD MAPS Working Group

Special Report 20

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- Competitively funds and oversees research projects;
- Provides intensive independent review of HEI-supported studies and related research;
- Integrates HEI's research results with those of other institutions into broader evaluations; and
- Communicates the results of HEI's research and analyses to public and private decision makers.

HEI typically receives balanced funding from the U.S. Environmental Protection Agency and the worldwide motor vehicle industry. Frequently, other public and private organizations in the United States and around the world also support major projects or research programs. GBD MAPS is funded by the William and Flora Hewlett Foundation and the Oak Foundation. HEI has funded more than 330 research projects in North America, Europe, Asia, and Latin America, the results of which have informed decisions regarding carbon monoxide, air toxics, nitrogen oxides, diesel exhaust, ozone, particulate matter, and other pollutants. These results have appeared in more than 260 comprehensive reports published by HEI, as well as in more than 1000 articles in the peer-reviewed literature.

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EXECUTIVE SUMMARY

Burden of Disease Attributable to Coal-Burning and Other Major Sources of Air Pollution in China

WHAT THIS STUDY ADDS

- This report provides the first comprehensive assessment of the current and predicted burdens of disease attributable to coal-burning and other major sources of air pollution in China at the national and provincial levels.
- **Coal-burning was the most important contributor to ambient $PM_{2.5}$**, responsible for 40% of population-weighted $PM_{2.5}$ in China. Given the large impact of coal combustion on ambient $PM_{2.5}$ concentrations, coal combustion was an important contributor to disease burden in China, causing an estimated 366,000 deaths in 2013.
- **Industrial sources**, from both coal (155,000 deaths) and noncoal (95,000 deaths) emissions, **were the largest sectoral contributor to disease burden in China**, responsible for 27% of the mortality attributable to ambient $PM_{2.5}$ in 2013.
- **Household solid fuel combustion, of both coal and biomass, is also an important source of disease burden in China.** Domestic biomass and coal combustion were together the next greatest contributor to ambient $PM_{2.5}$-attributable mortality in 2013 — with a combined impact (177,000 deaths; 19% of the mortality attributable to ambient $PM_{2.5}$ in 2013) larger than that of industrial coal (155,000 deaths), transportation (137,000 deaths), or coal combustion in power plants (86,500 deaths).
- **Under four different energy efficiency and air pollution control scenarios, population-weighted mean exposure to $PM_{2.5}$ is projected to decrease significantly** (from 54 $\mu g/m^3$ in 2013 to 50, 38, 38, and 27 $\mu g/m^3$ in 2030 for BAU1, BAU2, PC1, and PC2, respectively).
- **Despite these air pollution reductions, the overall health burden is expected to increase by 2030** as the population ages and becomes more susceptible to diseases most closely linked to air pollution.
- **Even under the most stringent energy use and pollution control future scenario, coal will remain the single largest source contributor to ambient $PM_{2.5}$ and health burden in 2030.** This finding highlights the urgent need for even more aggressive strategies to reduce emissions from coal combustion along with reductions in emissions from other sectors, strategies that are beginning to be incorporated in the Thirteenth Five-Year Plan.
- **The GBD MAPS estimates suggest that emissions reductions in the industrial and domestic sectors should be prioritized for future energy and air quality management strategies.** Because domestic combustion also leads to a large disease burden due to household air pollution exposure, reductions in domestic biomass and coal emissions would be particularly beneficial to public health.

INTRODUCTION AND BACKGROUND[1]

The Global Burden of Diseases, Injuries, and Risk Factors Study 2013 (GBD 2013[2]) estimated that exposure to ambient fine particulate air pollution ($PM_{2.5}$[3]) contributed to 2.9 million premature deaths in 2013 with 64% of those deaths occurring in China, India, and other developing countries of Asia, and also with large burdens of disease in Eastern Europe[4]. The GBD is the largest and most comprehensive effort to date to measure epidemiological levels and trends worldwide (*http://www.healthdata.org/gbd*). It currently involves more than 1000 collaborators from 108 countries. GBD 2013 estimated the burden of disease attributable to 79 risk factors including ambient and household air pollution for 1990–2013 in 188 countries and at subnational levels for China, the United Kingdom, and Mexico (Forouzanfar et al. 2015).

Estimating and communicating the burden of disease attributable to air pollution from coal combustion and other major sources is a critical next step to support the control of *both* air pollution and climate-forcing emissions. The GBD analytic framework is uniquely suited to develop such estimates for coal-burning and other key emission sources, including for example, transportation, industrial, and domestic combustion, which all contribute to high levels of air pollution. The GBD allows estimates at subnational, national, regional, and global scales of both current burden due to past exposure and predictions of future burden based on projected trends in mortality and air pollution emissions and concentrations.

GBD MAPS (Global Burden of Disease from Major Air Pollution Sources) is designed to estimate the current and future burdens of disease attributable to ambient air pollution from coal-burning and other major $PM_{2.5}$ sources in China, India, and Eastern Europe using the GBD framework, and to disseminate the estimates to inform planned policy decisions in these locales. The ultimate objective is to apply the methods developed as part of this project to multiple air pollution sources on a global scale, and to integrate them into the GBD framework so that they can be updated on a regular basis to track progress. GBD MAPS is a multiyear collaboration between the Health Effects Institute (HEI), the Institute for Health Metrics and Evaluation, Tsinghua University, the University of British Columbia and other leading academic centers.

This report describes the objectives, methods, and results of the GBD MAPS analysis for China and its provinces. Subsequent reports will report on similar analyses for India and Eastern Europe.

HEI Review Process

A draft final version of this report was reviewed for accuracy, quality, and appropriateness of interpretation by three independent external reviewers selected by HEI for their expertise in air quality, atmospheric chemistry and modeling, and health effects. The external reviewers were Tong

[1] This document was made possible, in part, through support provided by the William and Flora Hewlett Foundation and the Oak Foundation. The contents of this document have not been reviewed by these or other institutions, including those that support the Health Effects Institute; therefore, it may not reflect the views or policies of these parties, and no endorsement by them should be inferred.

[2] A list of abbreviations and the citation of the full report appear at the end of this Executive Summary.

[3] Particulate matter ≤ 2.5 µm in aerodynamic diameter.

[4] Cohen A, Brauer M, Burnett R, Anderson HR, Estep K, Frostad J, et al. The Global Burden of Disease attributable to ambient air pollution: estimates of current burden and 23-year trends from the GBD 2013 study. Submitted.

Zhu (Peking University), Markus Amman (IIASA Institute), and John Evans (Harvard University). The draft was also reviewed by experts on the GBD MAPS Steering Committee. The Working Group prepared the final report in response to the comments received. The report's major findings were also presented at the American Association for the Advancement of Science in February 2016 and HEI Annual Conference in Denver in May 2016.

AIR POLLUTION EMISSIONS AND EFFECTS

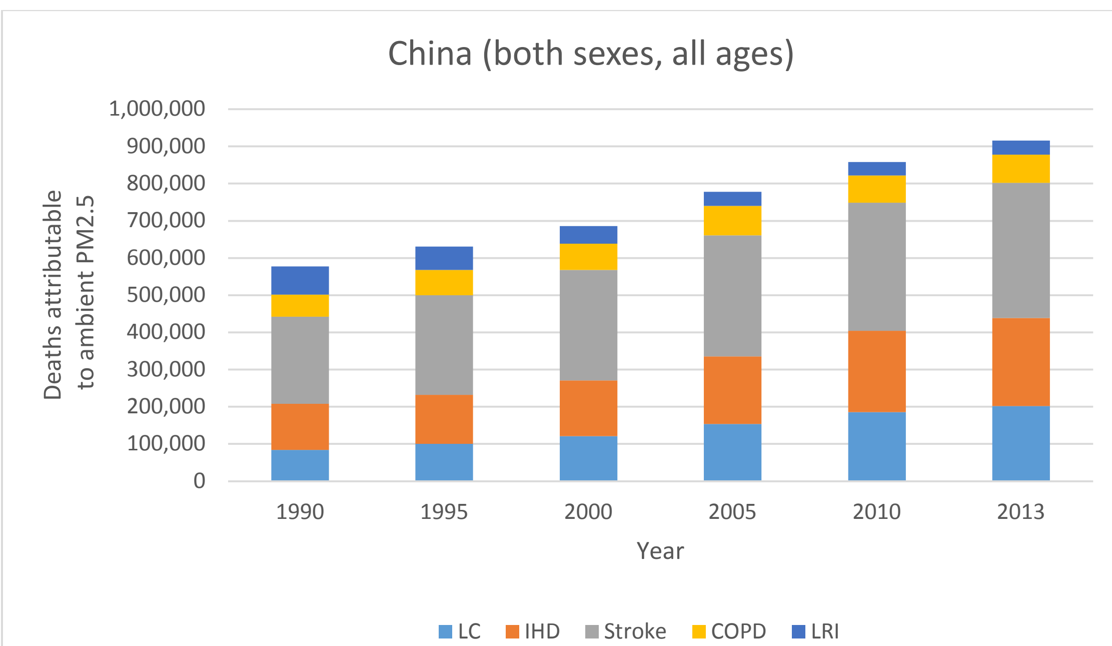
Chinese air pollution levels have increased over the last two decades and are now among the highest in the world. Estimates from the GBD indicate that in 2013 the population-weighted mean concentration of $PM_{2.5}$ for China as a whole was 54 μg/m^3, with 99.6% of the population estimated to live in areas where the World Health Organization (WHO) Air Quality Guideline of 10 μg/m^3 was exceeded (Brauer et al. 2016). At the provincial level, population-weighted mean concentrations ranged from 6.4 μg/m^3 (Tibet) to 83.5 μg/m^3 (Henan) in 2013. (Executive Summary Figure 1). Population-weighted $PM_{2.5}$ concentrations in China increased by 38% from 1990 to 2013, with increases of over 40% in some provinces, such as Tianjin where levels increased by 45%.

Executive Summary Figure 1. 2013 Chinese provincial-level population-weighted $PM_{2.5}$ concentrations.

These concentration increases reflect large increases in emissions of $PM_{2.5}$ and its precursors. With regard to coal, China's consumption increased from 1055.2 metric tons in 1990 to 3623 metric tons in 2013. China is the world's largest producer and consumer of coal, accounting for nearly half of the total global coal consumption, and coal combustion is widely recognized as an important contributor to ambient air pollution in China. In response, the State Council established a medium- and long-term national coal cap targets in the Air Pollution Prevention and Control Action Plan in September 2013, which were reinforced in the new Air Pollution Prevention and Control Law that took effect on January 1, 2016. The State Council Air Pollution Action Plan set regional coal consumption caps for key air pollution regions such as Beijing-Tianjin-Hebei, the Yangtze River Delta, and the Pearl River Delta, requiring them to reduce their coal consumption and achieve reductions in average $PM_{2.5}$ levels of 25%, 20%, and 15%, respectively, by 2017.

The health effects of exposure to particulate matter (PM) in ambient air are widespread and substantial, and they have been reviewed and summarized in detail (WHO 2005; U.S. Environmental Protection Agency [U.S. EPA] 2009). Long-term (months to years) exposure to $PM_{2.5}$ was determined by the U.S. EPA to be a cause of cardiovascular mortality, and adverse respiratory effects such as decreased lung function and development of asthma were likely to be causally linked to PM exposure. Further, the International Agency for Research on Cancer, concluded in 2014 that airborne PM was a cause of cancer in humans (Loomis et al. 2013). Chinese studies constitute an important and growing component of the international literature. In a comprehensive review published in 2010, HEI identified over 100 Chinese studies of the adverse effects of air pollution published as of 2007, including studies of mortality and morbidity from respiratory and cardiovascular disease due to short-term exposure to particles, lung cancer, and chronic respiratory disease (HEI 2010). Since then the Chinese epidemiologic literature on the adverse effects of air pollution has grown substantially, including new multicity studies of short-term exposure to air pollution and mortality and morbidity from cardiovascular and respiratory disease (Chen et al. 2012; Wong et al. 2008), and cohort studies of long-term exposure to air pollution and mortality (Cao et al. 2011; Zhou et al. 2014).

Of the 2.9 million premature deaths estimated globally in GBD 2013, 916,000 deaths took place in China. Cardiovascular disease, heart disease, and stroke account for the majority of these deaths, which have increased since 1990 (Executive Summary Figure 2) (Cohen et al. under review). In populated regions, a large fraction of $PM_{2.5}$ originates from combustion processes, and it includes both primary PM (direct emissions) and secondary PM (resulting from atmospheric transformations of gaseous precursors). Additionally the disease burden in China attributable to household air pollution from the combustion of solid fuels (e.g., coal, biomass) for cooking and heating is substantial, with an estimated 807,000 attributable deaths in 2013.

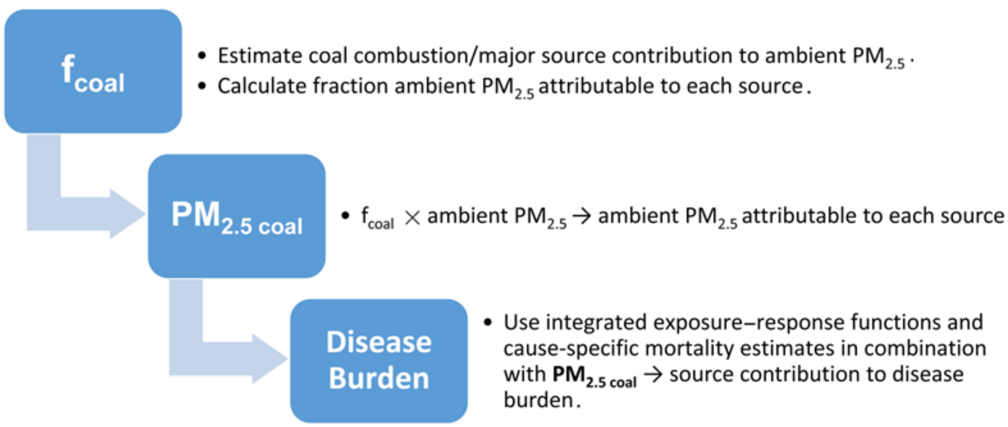


Executive Summary Figure 2. Deaths in China attributable to $PM_{2.5}$ 1990–2013 by year and cause. (LC = lung cancer, IHD = ischemic heart disease, COPD = chronic obstructive pulmonary disease, LRI = lower respiratory infection.)

Estimating the Burden of Disease Due to Coal-Burning and Other Major Air Pollution Sources

The overall analytic approach in GBD MAPS has three main components: (Executive Summary Figure 3)

1. Estimate the fractional contributions to ambient $PM_{2.5}$ from coal-combustion and other major emissions sources using the chemical transport model GEOS-Chem for the year 2013 and for 2030 under four different scenarios of energy use and pollution control.
2. Combine these fractional contributions with high-resolution ambient $PM_{2.5}$ concentration estimates developed for the GBD to estimate the fractional contributions to population exposure.
3. Estimate source contributions to disease burden for China at the national and provincial levels by combining the estimates of coal combustion and other source-specific ambient $PM_{2.5}$ with (a) cause-specific disease burden estimates from the GBD, and (b) integrated exposure–response (IER) functions describing air pollution mortality risk estimates for heart and lung diseases.



Executive Summary Figure 3. Schematic representation of GBD MAPS methodology for estimating the burden of disease attributable to coal-burning and other major sources for 2013 and for four scenarios for 2030.

Current Burden of Disease from Coal and Other Major Sources

The disease burden attributable to the contribution of coal combustion to ambient $PM_{2.5}$ in 2013 for China as a whole and for all provinces was estimated in a core analysis and in separate analyses for major sectoral contributions — specifically transportation, noncoal industrial, domestic biomass, open biofuel burning, and solvent use.

First, an emissions inventory of sulfur dioxide (SO_2), nitrogen oxides (NO_x), PM_{10}, $PM_{2.5}$, black carbon (BC), organic carbon (OC), nonmethane volatile organic compound (NMVOC), and ammonia (NH_3) was developed for China for the year 2013, based upon the model intercomparison study (MICS) Asia III emissions inventory for 2010 and updated to 2013. Emissions from each sector, stratified by province, were calculated from activity data (energy consumption, industrial products, solvent use, etc.), technology-based emission factors, and penetrations of control technologies. The activity data and technology distribution for each sector were derived based on the Chinese statistics, a wide variety of Chinese technology reports, and an energy-demand modeling approach.

Simulations were conducted using the Goddard Earth Observing System chemical transport model GEOS-Chem (version v9-01-03) to estimate the coal-combustion (and other sector) contributions to current and future ambient $PM_{2.5}$. Standard simulations were first conducted using the emission inventories of the base year, 2013. For sensitivity simulations, the total coal-related emissions and sector-specific coal-related emissions were removed respectively from the inventory in each simulation. The global and nested grid model of GEOS-Chem was run in sequence using the new inventories. The simulation results therefore depict the ambient $PM_{2.5}$ concentrations with the coal-related emission sources shut off. The differences between the standard and sensitivity simulations are analyzed to produce contributions of the total coal combustion and coal combustion from each specific source to ambient $PM_{2.5}$ concentration. In this way, the complexity of atmospheric transformations and formation of $PM_{2.5}$ from precursors are accounted for in the analyses. Sensitivity simulations include model runs to estimate the contributions of total coal, power plant coal, industrial coal, and domestic coal emissions.

The spatially resolved fractional contributions of coal combustion and other sources estimated with GEOS-Chem simulations were then multiplied by the high-resolution (0.1° × 0.1°) ambient $PM_{2.5}$ concentration estimates developed for GBD 2013. These estimates combine: (a) satellite-based estimates + GEOS-Chem, (b) TM5–FASST (Fast Scenario Screening Tool) chemical transport model simulations, and (c) available annual average PM measurements to estimate the amount of ambient $PM_{2.5}$ resulting from coal combustion. This approach explicitly incorporates the available Chinese surface measurements of $PM_{2.5}$ (in 2013) including approximately 80 $PM_{2.5}$ measurements as well as ~300 additional locations with PM_{10} measurements that were used to estimate $PM_{2.5}$ based on $PM_{2.5}$:PM_{10} ratios.

The spatially resolved estimated levels of coal combustion and other source-specific ambient $PM_{2.5}$ concentrations were then used with IER functions (which estimate the relative risk of $PM_{2.5}$ and adult ischemic heart disease, stroke, chronic obstructive pulmonary disease and lung cancer; and childhood and adult acute lower respiratory infections) to estimate sector contributions to disease burden for China as a whole and for each province.

Future Burden of Disease from Coal-Burning

Four different energy scenarios and air pollution control scenarios were developed that reflect different future emission pathways and incorporate changes in energy use and in emissions control (Executive Summary Table). These were used for analysis of mortality and disease burden projections in the year 2030. We estimated both coal and other major source contributions to ambient $PM_{2.5}$ and their associated disease burdens under each of the future scenarios for the year 2030 — considering both future mortality projections and future emissions scenarios.

Executive Summary Table. Definitions of Four Future Scenarios for 2030

Energy Policy	End-of-Pipe Emission Control Policy	
	[1] Twelfth Five-Year Plan for Environmental Protection	[2] Maximum Feasible Emission Controls Regardless of Cost
BAU: Current Legislation and Implementation Status as of end of 2012.	BAU[1]	BAU[2]
PC: Additional energy saving policies will be implemented, including life style changes, structural adjustments and energy efficiency improvements.	PC[1]	PC[2]

To estimate future (year 2030) mortality and disease burden for each of the four future scenarios, future population-weighted $PM_{2.5}$ concentrations and future mortality were estimated by scaling GBD 2013 estimates of exposure by the ratios of simulated ambient $PM_{2.5}$ in 2030 for each scenario with the simulated 2013 levels, and for mortality, according to observed temporal trends mortality rates and attributable fractions from 1990–2013, applied to 2030.

FINDINGS AND CONCLUSIONS

This report provides the first comprehensive assessment of the current and predicted burdens of disease attributable to coal-burning and other major sources in China at the national and provincial levels.

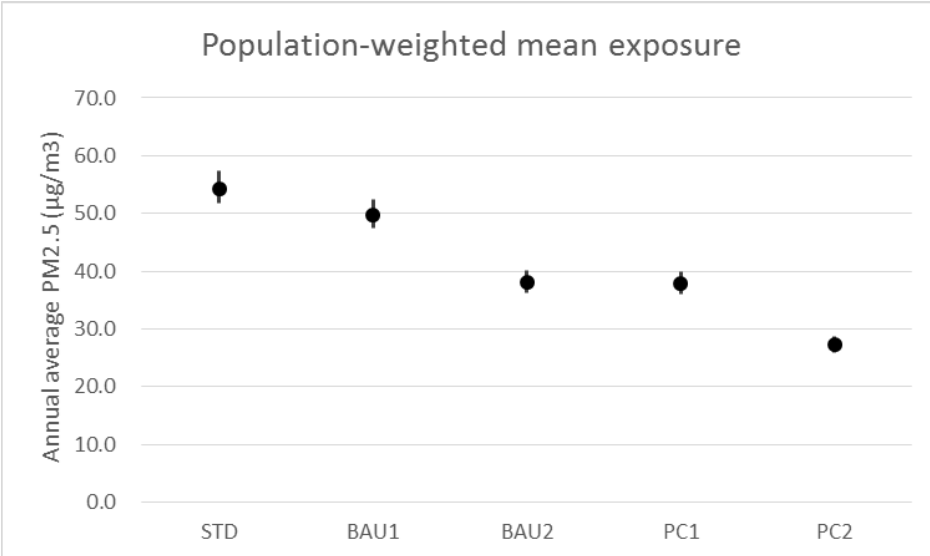
Coal-burning was the most important single source contributor to ambient $PM_{2.5}$, responsible for 40% of population-weighted $PM_{2.5}$ in China. In specific provinces (Chongqing, Guizhou, Sichuan) the contribution was nearly 50%. We compared our estimates with limited available source-apportionment analyses for three major Chinese cities and with model-based estimates in selected Chinese cities. Our estimates of the contribution of coal combustion to ambient $PM_{2.5}$ were remarkably similar to the source-apportionment study, although we estimated slightly higher contributions (3%–4%). Our estimates were somewhat lower than the model-based estimates.

Ambient $PM_{2.5}$ is a major contributor to mortality and disease burden in China, estimated to be responsible for 916,000 deaths in 2013, the 5th leading risk factor for mortality. Given the large impact of coal combustion on ambient $PM_{2.5}$ concentrations, coal combustion was an important contributor to disease burden in China, causing an estimated 366,000 deaths in 2013 (Executive Summary Figure 3). $PM_{2.5}$ from coal combustion specifically was the 12th leading risk factor for mortality in China in 2013 and accounted for more deaths than high cholesterol, drug use, or second-hand smoking.

Household solid fuel combustion, of both coal and biomass, is an important source of disease burden in China. Domestic biomass and coal combustion were together the next largest contributor to ambient $PM_{2.5}$ attributable mortality in 2013 — with a combined impact (177,000 deaths) larger than that of industrial coal (155,000 deaths), transportation (137,000 deaths), or coal combustion in power plants (86,500 deaths). Mortality attributable to ambient air pollution from domestic biomass combustion (136,000 deaths) was at the same level as that from industrial coal and transportation. Given the large contributions of domestic combustion to disease burden via household air pollution exposure and its important contribution to the burden attributable to ambient $PM_{2.5}$, reductions in domestic biomass and coal emissions would be expected to lead to large reductions in disease burden and should be prioritized for future energy and air quality management strategies.

Future Scenarios

Population-weighted mean exposure to $PM_{2.5}$ is projected to decrease under all scenarios (from 54 µg/m^3 in 2013 to 50, 38, 38, and 27 µg/m^3 in 2030 for BAU1, BAU2, PC1 and PC2, respectively) (Executive Summary Figure 4).



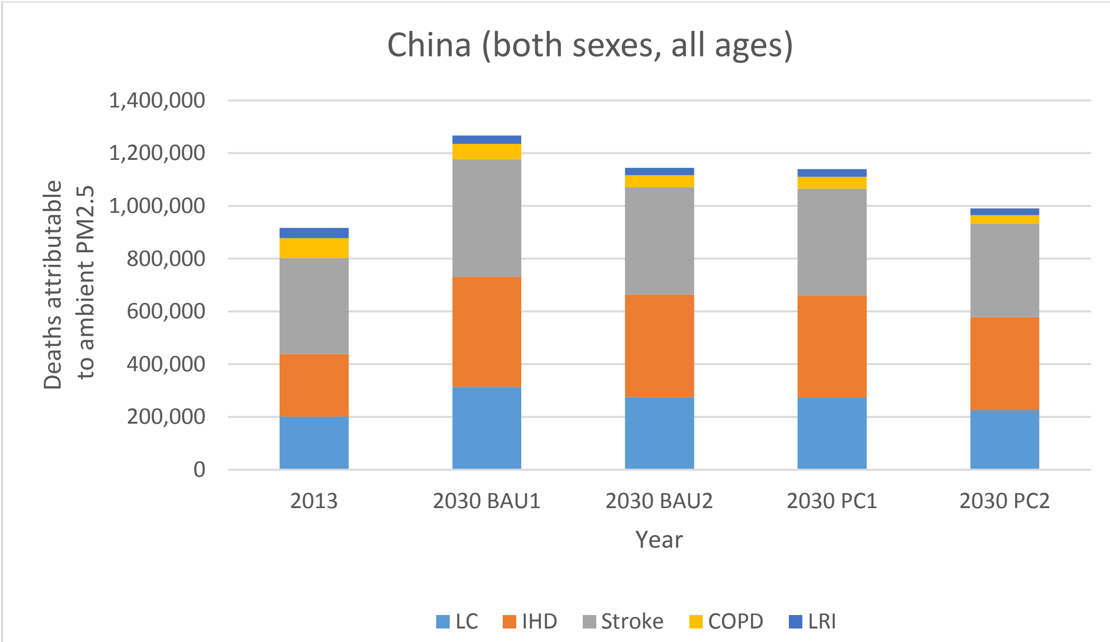
Executive Summary Figure 4. Population-weighted mean exposure to $PM_{2.5}$ under four future scenarios.

Despite reductions in $PM_{2.5}$ levels in all future scenarios, all of the future scenarios are predicted to lead to increases in future deaths attributable to ambient $PM_{2.5}$ with increases of 38%, 25%, 24%, and 8% for BAU1, BAU2, PC1, and PC2, respectively, compared to attributable mortality in 2013 (916,000). Specifically, ambient $PM_{2.5}$ was projected to be responsible for 1.3, 1.1, 1.1, and 0.99 million deaths in 2030, for the BAU1, BAU2, PC1, and PC2 scenarios, respectively.

The projected increase in mortality is due to the aging population and increased prevalence of ischemic heart disease, stroke, chronic obstructive pulmonary disease, and lung cancer, leading to increases in the number of deaths attributable to exposure to ambient $PM_{2.5}$ (Executive Summary Figure 5). These projections illustrate the importance of population dynamics in determining temporal trends in mortality attributable to ambient $PM_{2.5}$. GBD 2013 estimated increases in exposure, the numbers of deaths, and the death rate attributable to $PM_{2.5}$ between 1990 and 2013, but future projections predict increased attributable mortality, even with declining exposures.

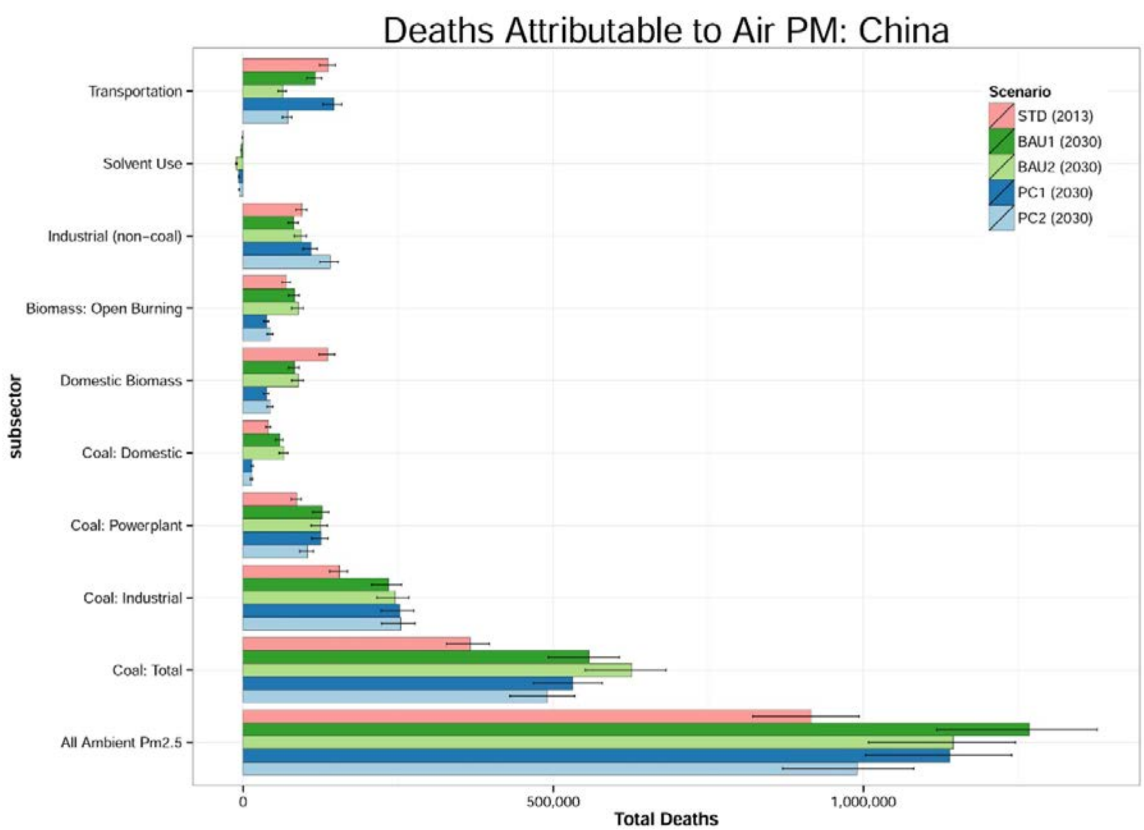
Differences between 2013 and 2030 largely reflect the impact of population aging and changes in the prevalence of diseases affected by exposure to air pollution. Differences between the different 2030 scenarios reflect the impact of energy policies and pollution control.

Importantly however, if normalized for age and population size, death rates from air pollution exposure would be predicted to decrease in 2030 in all scenarios, with the greatest reductions for the most stringent scenarios. Strict control of PM levels is therefore critical to stabilizing or reducing burden in the face of changing demographics.



Executive Summary Figure 5. Deaths in China attributable to $PM_{2.5}$ 2013-2030 by year and cause. Deaths in 2030 are predicted under four scenarios of pollution reduction developed for GBD MAPS.

The importance of coal combustion as a contributor to deaths attributable to $PM_{2.5}$ is projected to increase under all future scenarios, such that coal combustion will remain the single largest source contributor to ambient $PM_{2.5}$ (Executive Summary Figure 6).



Executive Summary Figure 6. Deaths attributable to $PM_{2.5}$ from major air pollution sources in 2013 and in 2030 under four alternative scenarios.

Indeed, even under the most stringent energy use and pollution control future scenario, coal will remain the single largest source contributor to ambient $PM_{2.5}$ and health burden in 2030. Whereas coal combustion was responsible for 40% of the mortality attributable to $PM_{2.5}$ in 2013, it is estimated to contribute to 44%, 55%, 47%, and 49% for BAU1, BAU2, PC1, and PC2, respectively. Further, the reductions in ambient $PM_{2.5}$ attributable to coal are lower than the reductions projected for the other sectors, resulting in an expected increase in the relative contribution of coal combustion to attributable disease burden. This finding highlights the urgent need for even more aggressive strategies to reduce emissions from coal combustion along with reductions in emissions from other sectors, strategies which are beginning to be incorporated in the Thirteenth Five-Year Plan.

REFERENCES

Brauer M, Freedman G, Frostad J, van Donkelaar A, Martin RV, Dentener F, et al. 2016. Ambient air pollution exposure estimation for the global burden of disease 2013. Environ Sci Technol 50:79–88; doi:10.1021/acs.est.5b03709.

Cao J, Yang C, Li J, Chen R, Chen B, Gu D, et al. 2011. Association between long-term exposure to outdoor air pollution and mortality in China: a cohort study. J Hazard Mater 186:1594–1600.

Chen R, Kan H, Chen B, Huang W, Bai Z, Song G, et al. 2012. Association of particulate air pollution with daily mortality: the China air pollution and health effects study. Am J Epidemiol 175:1173–1181.

Forouzanfar MH, Alexander L, Anderson HR, Bachman VF, Biryukov S, Brauer M, et al. 2015. Global, regional, and national comparative risk assessment of 79 behavioural, environmental and occupational, and metabolic risks or clusters of risks in 188 countries, 1990–2013: a systematic analysis for the Global Burden of Disease Study 2013. Lancet 386:2287–2323.

Health Effects Institute. 2010. Outdoor Air Pollution and Health in the Developing Countries of Asia: A Comprehensive Review. Special Report 18. Boston, MA:Health Effects Institute.

Loomis D, Grosse Y, Lauby-Secretan B, El Ghissassi F, Bouvard V, Benbrahim-Tallaa L, et al. 2013. The carcinogenicity of outdoor air pollution. Lancet Oncol 14:1262–1263.

U.S. Environmental Protection Agency. 2009. Final Report: Integrated Science Assessment for Particulate Matter. EPA/600/R-08/139F. Washington, DC:U.S. EPA.

Wong CM, Vichit-Vadakan N, Kan H, Qian Z. 2008. Public health and air pollution in Asia (PAPA): a multicity study of short-term effects of air pollution on mortality. Environ Health Perspect 116:1195–1202.

World Health Organization (WHO). 2005. WHO Air quality guidelines for particulate matter, ozone, nitrogen dioxide and sulfur dioxide. Global Update 2005. Available: *http://www.euro.who.int/__data/assets/pdf_file/0005/78638/E90038.pdf?ua=1.*

Zhou M, Liu Y, Wang L, Kuang X, Xu X, Kan H. 2014. Particulate air pollution and mortality in a cohort of Chinese men. Environ Pollut 186:1–6.

ABBREVIATIONS AND OTHER TERMS

BAU	business as usual scenario
BC	black carbon
GBD MAPS	global burden of disease from major air pollution sources
GEOS-Chem	Goddard Earth Observing System global chemical transport model
HEI	Health Effects Institute
IER	integrated exposure–response
NH_3	ammonia
NMVOC	nonmethane volatile organic compound
NO_x	nitrogen oxides
OC	organic carbon
PC	(alternative) policy scenario
PM	particulate matter
PM_{10}	particulate matter $\leq$ 10 µm in aerodynamic diameter
$PM_{2.5}$	particulate matter $\leq$ 2.5 µm in aerodynamic diameter
SO_2	sulfur dioxide
U.S. EPA	U.S. Environmental Protection Agency
WHO	World Health Organization

HEI SPECIAL REPORT 20

Burden of Disease Attributable to Coal-Burning and Other Major Sources of Air Pollution in China

INTRODUCTION[5,6]

PROJECT RATIONALE AND OVERVIEW

The Global Burden of Disease (GBD[7]) 2013 study estimated that exposure to ambient fine particulate air pollution ($PM_{2.5}$)[8] contributed to 2.9 million premature deaths in 2013 with 64% of those deaths occurring in China[9], India[10], and other developing countries of Asia[11], and also with large burdens of disease in Eastern Europe[12]. Estimating and communicating the burden of disease attributable to ambient air pollution from major specific sources is a critical next step to support the control of *both* air pollution and climate-forcing emissions from coal-burning and other major sources of air pollution in China and other developing countries of Asia.

The GBD analytic framework is uniquely suited to develop such estimates for coal-burning and other key emission sources, including predictions at subnational, national, regional, and global scales of future sector-specific impacts based on projected trends in mortality and air pollution emissions and concentrations. Major near-term opportunities currently exist in China, India, and Eastern Europe, where coal use is high and may not decrease in the near future absent major changes in energy policy. Other sources that contribute to high levels of air pollution include transportation, industrial sources, agriculture, and household biomass combustion.

The project, GBD MAPS (Global Burden of Disease from Major Air Pollution Sources), comprises a multi-year collaboration between the Health Effects Institute (HEI), the Institute for Health Metrics and

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[7] A list of abbreviations and other terms appears at the end of this report.

[8] The mass concentration of particles with aerodynamic diameter smaller than or equal to 2.5 micrometers. $PM_{2.5}$ comprises particles directly emitted from major sources, primary particles, and particles resulting from atmospheric transformation of gaseous emissions from major sources, secondary particles.

[9] China: 916,102 attributable deaths in 2013.

[10] India: 586,788 attributable deaths in 2013.

[11] South Asia excluding India (Afghanistan, Bangladesh, Bhutan, Nepal, Pakistan): 174,247 attributable deaths in 2013; Southeast Asia (Cambodia, Indonesia, Laos, Malaysia, Maldives, Myanmar, Philippines, Sri Lanka, Thailand, Timor-Leste, Vietnam): 184,676 attributable deaths in 2013.

[12] Eastern Europe (Belarus, Estonia, Latvia, Lithuania, Moldova, Russia, Ukraine): 168,396 attributable deaths in 2013.

Evaluation (IHME), Tsinghua University, the University of British Columbia, and other leading academic centers. The project goal is to estimate the burden of disease attributable to ambient air pollution from coal-burning and other major sources in China, India, and Eastern Europe using the GBD framework and to disseminate the estimates to inform planned policy decisions in these locales. The ultimate objective is to apply the methods developed as part of this project to multiple air pollution sources on a global scale and to integrate them into the GBD framework so that they can be updated on a regular basis to track progress.

This report describes the objectives, methods, and results of the GBD MAPS analysis for China and its provinces. Subsequent reports will report on similar analyses for India and Eastern Europe.

SPECIFIC OBJECTIVES

1. To estimate ambient concentrations and the associated disease burden attributable to coal combustion from all sectors and for major subsectors, including stationary (power generation, industrial) and domestic (household) sources for the year 2013.
2. To compare the above estimates of ambient $PM_{2.5}$ concentrations attributable to coal combustion derived from chemical transport model simulations with available receptor modeling estimates[13] of the coal combustion contribution for a number of cities in China.
3. To estimate the ambient concentrations and disease burden attributable to other sectors, specifically transportation, noncoal industrial, and other sources (domestic biomass, open biomass burning, solvent use).
4. To estimate ambient concentrations and disease burden attributable to coal combustion and other major source sectors for the year 2030 — considering both future mortality projections and future emissions scenarios. Scenarios including different energy sources and air pollution control strategies are evaluated to reflect different future emissions possibilities.

PROCESS

The GBD MAPS Project was begun by HEI in 2014. HEI selected the Working Group that designed and conducted the analyses and drafted this report, and an International Steering Committee who advised the Working Group and reviewed its work. The Working Group initially developed a detailed analytic plan in consultation with the Steering Committee. In April 2015 the Working Group prepared a mid-project progress report that was reviewed by the Steering Committee. A draft final report was reviewed by the Steering Committee and three external reviewers selected by HEI. The external reviewers were Tong Zhu (Peking University), Markus Amman (IIASA Institute), and John Evans (Harvard University). The Working Group prepared the final draft report in response to the comments received. The report's major findings were presented at the American Association for the Advancement of Science in February 2016 and the HEI Annual Conference in Denver in May 2016.

[13] That is, source apportionment, in which particle chemical speciation monitoring at specific receptor sites is used to quantify contributions to ambient concentrations from different source sectors.

BACKGROUND

AIR QUALITY IN CHINA

GBD estimates of ambient $PM_{2.5}$, which incorporate ground measurements, chemical transport model estimates and satellite-based estimates indicate that in 2013 the population-weighted mean concentration for China as a whole was 54 μg/m^3, with 99.6% of the population estimated to live in areas where the World Health Organization (WHO) Air Quality Guideline of 10 μg/m^3 was exceeded (Brauer et al. 2016). At the provincial level, population-weighted mean concentrations ranged from 6.4 μg/m^3 (Tibet) to 83.5 μg/m^3 (Henan) in 2013. Further, there was an estimated 38% increase in population-weighted $PM_{2.5}$ concentrations in China between 1990 and 2013. At the provincial level, increases in population-weighted concentrations over 40% were estimated in Guanxi (46%), Hunan (41%), Jiangxi (45%), and Tianjin (45%).

These concentration increases reflect large increases in emissions of $PM_{2.5}$ and its precursors. China's coal consumption increased from 1055.2 metric tons (Mt) in 1990 to 3623 Mt in 2013. China is the world's largest producer and consumer of coal, accounting for nearly half of the total global coal consumption. Coal is responsible for 50%–60% of China's $PM_{2.5}$ emissions. The State Council established a medium- and long-term national coal cap target in the Air Pollution Prevention and Control Action Plan in September 2013 — a goal that was reinforced in the latest draft of the Air Pollution Prevention and Control Law released this past September. The State Council Air Pollution Action Plan set regional coal consumption caps for key air pollution regions such as Beijing–Tianjin–Hebei, the Yangtze River Delta, and the Pearl River Delta, requiring them to reduce their coal consumption and achieve reductions in average $PM_{2.5}$ levels of 25%, 20%, and 15%, respectively, by 2017.

During the years 2005–2010, the nitrogen oxides (NO_x), sulfur dioxide (SO_2), $PM_{2.5}$, nonmethane volatile organic compound (NMVOC), and ammonia (NH_3) emissions in China changed by +34%, −15%, −12%, +21%, and +10%, respectively, as an integrated effect of economic development and recent control policies (Zhao et al. 2013a). There have been a number of studies on future emission trends in China, and those reported after 2005 are briefly reviewed (Amann et al. 2008; Cofala et al. 2012; Ohara et al. 2007; S.X. Wang et al. 2014; Wei et al. 2011; Xing et al. 2011; Zhao et al. 2013c). China's NO_x emissions could increase by 16%–36% during 2010–2030 under current legislation and current implementation status. With the implementation of the 12th Five Year Plan and continuously strengthened end-of-pipe control policies after 2015, China's NO_x emissions could be significantly reduced. The maximum feasible control measures could reduce China's NO_x emissions to less than 1/3 of the 2010 levels (Amann et al. 2008; Cofala et al. 2012; S.X. Wang et al. 2014; Xing et al. 2011; Zhao et al. 2013c). SO_2 and $PM_{2.5}$ emissions in China are projected to remain relatively stable up to 2030 under the current policies. If the best available technologies are fully applied, the SO_2 and $PM_{2.5}$ emissions would be reduced to about 1/4–1/3 of the levels of the base year (Amann et al. 2008; Cofala et al. 2012; S.X. Wang et al. 2014; Xing et al. 2011). China's NMVOC emissions are estimated to increase gradually from 2010 to 2030 under current policy and current implementation status. Given that China is still in the initial stage of NMVOC emission controls, and new policies could only emerge slowly in the next five years, the emission trends should not deviate greatly from the baseline until 2020. However, control measures at different stringency levels might result in dramatically different emissions by 2030 (Ohara et al. 2007; S.X. Wang et al. 2014; Wei et al. 2011; Xing et al. 2011).

$PM_{2.5}$ mass concentrations and speciation compositions varied substantially over geographical regions in China. Near six-fold variations in average $PM_{2.5}$ concentrations (34.0–193.4 μg/m^3) across China were found, with high $PM_{2.5}$ levels (> 100 μg/m^3) appearing in the cities in the northern and western regions and low levels (< 40 μg/m^3) in the remote forest area (Changbai Mountain) and in Hong Kong. The percentages of the sum of sulfate, nitrate, and ammonium; organic matter; crustal material; and elemental carbon in $PM_{2.5}$ mass ranged from 7.1%–57%, 17.7%–53%, 7.1%–43%, and 1.3%–12.8%, respectively. At both urban and rural sites in the eastern region, the sum of sulfate, nitrate, and ammonium typically constituted much higher fractions (40%–57%) of $PM_{2.5}$ mass. Organic matter made a significant contribution to $PM_{2.5}$ over all the sites. Organic matter plus sulfate, nitrate, and ammonium accounted for 53%–90% of $PM_{2.5}$ mass across China. $PM_{2.5}$ speciation across China was also characterized by high content of crustal material, which was usually at a level of more than ~10 $\mu g/m^3$ or shared ~10% of $PM_{2.5}$ mass in urban areas, due to transported desert dust and locally induced dust (Yang et al. 2011).

A number of studies have used atmospheric models to study the source contributions of energy use to air pollutant concentration. Early studies mainly focused on traditional air pollutants, including SO_2, NO_x, CO, and ozone, using relatively simple models such as CALMET/CALPUFF (Hao et al. 2007). Later on, more studies (Bi et al. 2007; Chen et al. 2007; Cheng et al. 2007; Li et al. 2008; Wang et al. 2008; Wu et al. 2009, 2012) shifted their emphasis to particulate matter (PM), but mainly PM_{10}. Among these studies, most of them took advantage of 3-D chemical transport models such as the Community Multiscale Air Quality Model (CMAQ). Recently, due to frequent air pollution episodes characterized by extremely high $PM_{2.5}$ concentrations in China, research has increasingly focused on $PM_{2.5}$. For example, Zhang and colleagues (2012) studied source contributions to sulfate and nitrate in $PM_{2.5}$ and reported that while the power sector was the largest contributor to inorganic components, industry and traffic were also important sources. Yang and colleagues (2013), in their study of the city of Jinan in northern China, found that secondary particle sources accounted for 55% of the total $PM_{2.5}$ concentration with coal combustion contributing an additional 21%. L.T. Wang and colleagues (2014) studied a severe $PM_{2.5}$ pollution episode in January 2013 in Hebei province in northern China and concluded that industrial and domestic sources were the most significant local contributors. D.X. Wang and colleagues (2014) studied the same pollution episode in a different city, Xi'an in northwestern China, and also reported that industrial and domestic activities were the two predominant contributing sources, accounting for 58% and 16%, respectively, during this high $PM_{2.5}$ episode. Zhang and colleagues (2015) evaluated source contributions to $PM_{2.5}$ in northern China using the adjoint to the Goddard Earth Observing System global chemical transport model GEOS-Chem (*www.geos-chem.org*). For wintertime periods in Beijing, residential (49.8%) and industrial (26.5%) sources were the most important contributors to ambient $PM_{2.5}$. Most of the studies conducted with regional models focused on developed metropolises or specific time periods, such as wintertime air pollution episodes. Analyses at national to global spatial scales and at seasonal to annual temporal scales often employed global chemical transport models such as GEOS-Chem. These studies similarly began with an emphasis on traditional pollutants such as carbon monoxide (e.g., Heald et al. 2004; Wang et al. 2004). Subsequent $PM_{2.5}$ analyses have investigated sources of primary and secondary carbonaceous aerosols (e.g., Fu et al. 2012) as well as mechanisms for enhanced sulfate production during air pollution episodes (e.g., Y. Wang et al. 2014). A number of studies have applied the GEOS-Chem model to interpret satellite observations of aerosol optical depth to estimate $PM_{2.5}$ at both the global scale (e.g., van Donkelaar et al. 2010; 2015) and the national scale (e.g., Geng et al. 2015). Several studies have investigated sensitivity to $PM_{2.5}$ sources (e.g., Jiang et al. 2015; Kharol et al. 2013; Zhang et al. 2015). Nonetheless, while many regional or national scale studies have examined contributions from major emission sources including power plants, industry, traffic, and domestic sectors, none of them has studied coal-burning in each sector separately. However, the utilization of coal and end-of-pipe control policies is quite different in different sectors, which deserves further investigation.

BURDEN OF DISEASE IN CHINA ATTRIBUTABLE TO AIR POLLUTION

Air Pollution and Health

The air pollution to which individuals are exposed is a complex mixture including hundreds of individual gaseous compounds and particles of complex composition that varies in composition both spatially and temporally. Therefore indicator pollutants are typically used to estimate exposures for epidemiologic analysis and disease burden assessment. Within the GBD framework, disease burden attributable to both ozone and $PM_{2.5}$ is considered, based upon evidence of independent adverse health impacts and distinctions between the spatial and temporal patterns of concentrations of these two species. Given that the burden attributable to $PM_{2.5}$ vastly exceeds that attributable to ozone (in the case of GBD 2013 — 2.9 million [916,000] vs 216,000 [67,000] deaths globally [in China] attributable to $PM_{2.5}$ and ozone, respectively), here we focus our assessment only on $PM_{2.5}$. In populated regions, a large fraction of $PM_{2.5}$ originates from combustion processes, and it includes both primary PM (direct emissions) and secondary PM (resulting from atmospheric transformations).

Additionally the disease burden in China attributable to household air pollution from the combustion of solid fuels (e.g., coal, biomass) for cooking and heating is substantial, with an estimated 807,000 attributable deaths in 2013. While we do estimate the contribution of such domestic pollution to concentrations of ambient $PM_{2.5}$ and the resulting disease burden, we do not estimate the proportion of disease burden from such sources due to household exposure. As such, our estimates of the disease burden attributable to such domestic sources substantially underestimate the full burden. The estimation of the combined impact of household air pollution via its contributions to ambient air pollution and as a specific risk factor is an area of uncertainty; at present there are no studies that estimate the joint effects of exposure to ambient and household air pollution. At present, in the GBD project the effects of these two risk factors are assumed to be independent[14].

The health effects of exposure to PM in ambient air are widespread and substantial, and they have been reviewed and summarized in detail (WHO 2005; U.S. Environmental Protection Agency [U.S. EPA] 2009). The epidemiologic observations of adverse health impacts associated with elevated ambient $PM_{2.5}$ concentrations is supported by toxicological experiments, epidemiologic analyses of acute exposures, and controlled exposure studies. For example, in its most recent comprehensive Integrated Science Assessment (ISA) document, the U.S. EPA determined that short-term exposure to fine particles ($PM_{2.5}$) was a cause of mortality and cardiovascular effects, such as hospitalization and emergency department visits, and was likely to be a cause of respiratory effects such as chronic obstructive pulmonary disease (COPD), respiratory infection hospitalizations, and emergency department visits for asthma. Long term (months to years) exposure to $PM_{2.5}$ was determined to be a cause of cardiovascular mortality, while respiratory effects such as decreased lung function, increased respiratory symptoms, and development of new asthma were likely to be causally linked to PM exposure. At the time of this assessment, reproductive and developmental effects, as well as cancer, were characterized by the U.S. EPA as suggestive of being caused by long-term exposure to fine PM. Of note, a number of systematic reviews have since concluded that $PM_{2.5}$ exposure is associated with low birthweight and preterm birth (Shah et al. 2011; Stieb et al. 2012). Detailed evaluations by the American Heart Association (Brook et al. 2010) and the European Society of Cardiology (Newby et al. 2015) have also concluded that $PM_{2.5}$ exposure is a cause of cardiovascular morbidity and mortality, and that exposure for a few hours to weeks can trigger cardiovascular disease-related mortality and nonfatal events. Evidence on the effects of both short-term and long-term exposure to PM on respiratory disease has also strengthened in recent years. For example, evidence now suggests that exposure to PM is associated with hospitalization for asthma as well as for COPD (Sava et al. 2012). Further, the International Agency for Research on Cancer (IARC) concluded in 2014 that airborne PM was a cause of cancer in humans (Loomis et al. 2013).

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[14] Following standard approaches, the combined effect of multiple independent risk factors is estimated by: Combined PAF = $1 - (1-PAF_1)(1-PAF_2)(1-PAF_3)\ldots$where PAF is the Population Attributable Fraction for risk factors 1, 2, and 3. The Population Attributable Fraction is the proportional reduction in burden of disease that would occur if exposure to the risk factor was reduced to a theoretical minimum risk level.

Chinese studies constitute an important and growing component of the international literature summarized above. Recent studies based on data from GBD 2013 indicate that Chinese life expectancy increased between 1990 and 2013 on national and provincial levels, and that cancer and noncommunicable cardiovascular and respiratory diseases are now the leading causes of death in China, albeit with considerable variation in their relative importance at the provincial level (Zhou et al. 2015). In a comprehensive review published in 2010, HEI identified over 100 Chinese studies of the adverse effects of air pollution in China published as of 2007, including studies of mortality and morbidity from respiratory and cardiovascular disease due to short-term exposure to particles, lung cancer (LC), and chronic respiratory disease (HEI 2010). Since then the Chinese epidemiologic literature on the adverse effects of air pollution has grown substantially, including new multicity studies of short-term exposure to air pollution and mortality and morbidity from cardiovascular and respiratory disease (Chen et al. 2012; Wong et al. 2008) and cohort studies of long-term exposure to air pollution and mortality (Cao et al. 2011; Dong et al. 2012; Zhang et al. 2011, 2014; Zhou et al. 2014). For example, Cao and colleagues (2011) reported an increased risk of mortality from cardiorespiratory disease and LC associated with long-term exposure to total suspended particulates (TSPs) in 70,947 residents of 31 Chinese cities in the China National Hypertension Survey and its follow-up study. These new studies are consistent with the large international literature that identifies exposure to particulate air pollution as a cause of mortality and morbidity from cardiovascular and respiratory disease, and they may contribute to the scientific literature on long-term effects of air pollution for high-exposure settings typical in developing countries. This literature is summarized in the text box: Summary of Studies on Air Pollution and Mortality in China.

There has been no direct epidemiologic evidence on mortality risk from long-term exposure to $PM_{2.5}$ in China. Interestingly, after conversion from TSP/PM_{10} to $PM_{2.5}$, the Chinese air pollution cohort studies generally reported lower exposure–response coefficients of $PM_{2.5}$ compared to studies in developed countries. For example, each 10 $\mu g/m^3$ $PM_{2.5}$ was associated with a 0.9% and a 2.5% increased risk of mortality in the China National Hypertension Survey and its follow-up study (Cao et al. 2011) and Chinese men’s cohort (Zhou et al. 2014). In contrast, a 10 $\mu g/m^3$ $PM_{2.5}$ was associated with a 4% and a 16% increase of total mortality in the ACS cohort (Pope et al. 2002) and Harvard Six Cities’ study (Laden et al. 2006). The heterogeneity might be explained by various characteristics of the study contexts, such as exposure–assessment methodology, $PM_{2.5}$ levels and components, and population sensitivity to $PM_{2.5}$. It may also reflect a nonlinear relationship between long-term exposure to $PM_{2.5}$ and mortality (Pope et al. 2011). The findings of the Chinese cohorts support the Integrated Exposure-Response (IER) Function, which indicated that the relationship between air pollutants’ concentration and health risk is likely to be nonlinear and tends to become flat at the higher end (Burnett et al. 2014). The IER Function provides a reasonable way to inform the risk estimates across the full range of $PM_{2.5}$ concentrations worldwide.

Within the GBD and based upon systematic reviews of the literature in which all evaluated risk factors were held to a comparable evidence standard, $PM_{2.5}$ exposure was determined to be a cause of ischemic heart disease (IHD), stroke, LC, and COPD in adults and of lower respiratory infections (LRI) in both adults and children under the age of five years (Forouzanfar et al. 2015; Lim et al. 2012). The determinations of adult health impacts were based primarily on evidence derived from long-term cohort studies where $PM_{2.5}$ exposure was associated with shortening of life. In epidemiologic cohort studies of long-term exposure (which form the basis of the exposure–response functions used in a health impact assessment) $PM_{2.5}$ is the most robust indicator of adverse (mortality) impacts related to air pollution (Chen et al. 2008). Here, as in most epidemiologic analyses, ambient concentrations are used as proxies for personal exposure to air pollution of outdoor origin.

Text Box: Summary of Studies on Air Pollution and Mortality in China

- ✓ Observational studies
  - ■ Short-term exposure studies (time-series or case–crossover analyses)
    - ◆ Single-city analysis: Beijing (Chang et al. 2003), Shanghai (Kan et al. 2008), Wuhan (Qian et al. 2007), etc.
    - ◆ Multiple-city analysis: PAPA (Wong et al. 2008), CAPES (Chen et al. 2012)
  - ■ Long-term exposure studies:
    - ◆ Cohort study: China National Hypertension Survey and its follow-up study (Cao et al. 2011), Chinese Men Cohort (Zhou et al. 2014), Four Northern Chinese Cities Cohort (Zhang et al. 2014), Shenyang Cohort (Dong et al. 2012; Zhang et al. 2011)
    - ◆ Ecological study: Beijing (Zhang et al. 2000), Shenyang (Xu et al. 1996), Guangzhou (Tie et al. 2009)
- ✓ Intervention studies: Hong Kong Fuel Intervention study (Hedley et al. 2002)

References:

Cao J, Yang C, Li J, Chen R, Chen B, Gu D, et al. 2011. Association between long-term exposure to outdoor air pollution and mortality in China: a cohort study. J Hazard Mater 186:1594–1600.

Chang G, Pan X, Xie X, Gao Y. 2003. [Time-series analysis on the relationship between air pollution and daily mortality in Beijing]. Wei Sheng Yan Jiu 32:565–568. [In Chinese.]

Chen R, Kan H, Chen B, Huang W, Bai Z, Song G, et al. 2012. Association of particulate air pollution with daily mortality: The China Air Pollution and Health Effects Study. Am J Epidemiol 175:1173–1181.

Dong GH, Zhang P, Sun B, Zhang L, Chen X, Ma N, et al. 2012. Long-term exposure to ambient air pollution and respiratory disease mortality in Shenyang, China: a 12-year population-based retrospective cohort study. Respiration 84:360–368.

Hedley AJ, Wong CM, Thach TQ, Ma S, Lam TH, Anderson HR. 2002. Cardiorespiratory and all-cause mortality after restrictions on sulphur content of fuel in Hong Kong: an intervention study. Lancet 360:1646–1652.

Kan H, London SJ, Chen G, Zhang Y, Song G, Zhao N, et al. 2008. Season, sex, age, and education as modifiers of the effects of outdoor air pollution on daily mortality in Shanghai, China: the Public Health and Air Pollution in Asia (PAPA) study. Environ Health Perspect 116:1183–1188.

Qian Z, He Q, Lin HM, Kong L, Liao D, Dan J, et al. 2007. Association of daily cause-specific mortality with ambient particle air pollution in Wuhan, China. Environ Res 105:380–389.

Tie XX, Wu D, Brasseur G. 2009. Lung cancer mortality and exposure to atmospheric aerosol particles in Guangzhou, China. Atmos Environ 43:2375–2377.

Wong CM, Vichit-Vadakan N, Kan H, Qian Z. 2008. Public Health and Air Pollution in Asia (PAPA): a multicity study of short-term effects of air pollution on mortality. Environ Health Perspect 116:1195–1202.

Xu ZY, Liu Y, Yu D, Chen B, Xu X, Jin L, et al. 1996. [Effect of air pollution on mortalities in Shenyang city]. Chin J Public Health 15:61–64. [In Chinese.]

Zhang J, Song H, Tong S, Li L, Liu B, Wan L. 2000. Ambient sulfate concentration and chronic disease mortality in Beijing. Sci Total Environ 262:63–71.

Zhang LW, Chen X, Xue XD, Sun M, Han B, Li CP, et al. 2014. Long-term exposure to high particulate matter pollution and cardiovascular mortality: a 12-year cohort study in four cities in northern China. Environ Int 62:41–47.

Zhang P, Dong G, Sun B, Zhang L, Chen X, Ma N, et al. 2011. Long-term exposure to ambient air pollution and mortality due to cardiovascular disease and cerebrovascular disease in Shenyang, China. PLoS One 6:e20827; doi:10.1371/journal.pone.0020827.

Zhou M, Liu Y, Wang L, Kuang X, Xu X, Kan H. 2014. Particulate air pollution and mortality in a cohort of Chinese men. Environ Pollut 186:1–6.

The above evidence has not identified consistently different levels of impacts for different PM components or sources, though the differential toxicity of PM of varying composition and from diverse sources remains an area of active research. The most recent U.S. EPA synthesis (U.S. EPA 2009) concluded that "many constituents of PM can be linked with differing health effects and the evidence is not yet sufficient to allow differentiation of those constituents or sources that are more closely related to specific health outcomes." Further, with respect to sources of PM, the WHO REVIHAAP assessment (WHO 2013) concluded that evidence developed since the 2009 U.S. EPA ISA did not lead to any changes in conclusions regarding the inability to differentiate PM constituents or sources that are more closely linked to specific health outcomes. IARC's recent evaluation of the carcinogenicity of particulate air pollution was assumed to be independent of source or composition (Loomis et al. 2013). This conclusion has most recently been corroborated in large-scale epidemiologic and toxicologic studies (HEI NPACT Review Panel 2013). Based on the current evidence, the GBD assessments of $PM_{2.5}$ have followed this same assumption, which we also employ in this analysis.

Integrated Exposure–Response Functions

In the GBD framework the relative risks of mortality from IHD, stroke, COPD, LC, and LRI and pneumonia in children and adults were estimated using cause-specific IER (Burnett et al. 2014). The IER integrates published relative risk estimates for $PM_{2.5}$ from different sources of exposure (outdoor air pollution, second-hand smoke, household air pollution, and active smoking) to estimate the relative risk of mortality from exposure to $PM_{2.5}$ over the entire global range. The application of these functions has particular importance for China, as much of the population exposure distribution lies above the concentrations for which epidemiologic estimates of effect have been characterized. In order to estimate air pollution effects at these higher concentrations the IERs assume that risk is a function of $PM_{2.5}$ inhaled dose, regardless of exposure source, consistent with the current evidence summarized above. The functions reflect the change in risk observed in cohort studies at low concentrations of ambient $PM_{2.5}$ and are near-linear at low concentrations. The predicted risk for the highest $PM_{2.5}$ concentrations is consistent with the risks from smoking (Pope et al. 2011). In GBD 2013 the IERs were fit using a Bayesian Markov Chain Monte Carlo approach and a modified power function (A. Cohen et al. Submitted[15]; Forouzanfar et al. 2015). More detail is provided in the text box, IER Model for $PM_{2.5}$.

[15] Cohen A, Brauer M, Burnett R, Anderson HR, Estep K, Frostad J, et al. The Global Burden of Disease attributable to ambient air pollution: estimates of current burden and 23-year trends from the GBD 2013 study. Submitted.

Text Box: IER Model for $PM_{2.5}$

Risk estimates attributable to exposure to outdoor concentrations of fine particulate matter ($PM_{2.5}$) of mortality from IHD, stroke, COPD, LC, and acute LRI in both children and adults is constructed by integrating information on risk from multiple sources of $PM_{2.5}$ exposure such as ambient air pollution (AAP), second-hand smoke (SHS), household air pollution from burning of biofuels for heating and cooking (HAP), and active smoking (AS). This approach was taken since no direct evidence of risk over the entire global concentration range was available. Exposure to each of these sources was integrated on an equivalent 24-hour average ambient concentration in μg/m^3. We developed the IER assuming that the effect of each type of $PM_{2.5}$ exposure (AAP, SHS, HAP, and AS) was independent of the other. That is, exposure to AAP is based solely on the ambient air exposure estimates of the specific cohort studies examined. The same assumption was made for the other types of $PM_{2.5}$ exposures. We used this type of risk information as a tool to extend the concentration–response curve beyond that observed in cohort studies in North American and Western Europe.

The IER has the mathematical form

$$IER(z) = 1 + \alpha \times (1 - e^{-\beta (z - z_{cf})_+^{\gamma}})$$

for counterfactual concentration z_{cf}, below which no additional risk is assumed since $(z - z_{cf})_+ = (z - z_{cf})$ if $z > z_{cf}$ and zero otherwise. The uncertainty distributions of the IER parameters (α, β, γ) were determined within a Bayesian framework assuming that the logarithm of the each study's observed relative risk was normally distributed, with mean defined by the IER function and variance given by the square of the observed standard error of the study-specific log-relative risk estimate. The parameters (α, β, γ) were each assumed to have a noninformative gamma distribution independent of each other, each with mean 100 and variance 10,000.

The counterfactual concentration, z_{cf}, or Theoretical Minimum Risk Exposure Level (TMREL) was assigned a uniform distribution with lower/upper bounds given by the average of the minimum and 5th percentiles of outdoor air pollution cohort studies exposure distributions, with the assumption that it is unpractical to characterize the shape of the concentration–response function below the 5th percentile of any exposure distribution. The specific outdoor air pollution cohort studies selected for this averaging were based on the criteria that their 5th percentiles were less than that of the American Cancer Society Cancer Prevention II (CPS II) cohort's 5th percentile of 8.8 μg/m^3. This criteria was selected since GBD 2010 (Lim et al. 2012) used the minimum, 5.8 μg/m^3, and 5th percentile solely from the CPS II cohort. The resulting lower/upper bounds for GBD 2013 were 5.9 μg/m^3 and 8.7 μg/m^3.

One thousand sets of IER parameter estimates were generated by the estimated uncertainty distributions of (α, β, γ) and the prespecified uniform uncertainty distribution of z_{cf}. One thousand predicted values of the IER curve were calculated for each $PM_{2.5}$ concentration based on these 1000 sets of parameter estimates and thus characterize the uncertainty in the estimates of the IER function. The average of the 1000 IER predictions at each concentration were used as the central estimate and the upper/lower bounds were defined by the 0.975/0.025 percentiles.

Text box continues on the next page.

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The figure below displays the mean IER function over the 1000 simulations for cardiovascular IHD, LC and COPD, cardiovascular stroke, and acute LRI over the global range in $PM_{2.5}$ concentrations (< 125 µg/m^3) (A. Cohen et al. Submitted). The upper and lower uncertainty bounds are also presented. The IER curves are all supralinear with a greater change in relative risk for lower concentrations compared to higher values. IHD, stroke, and LRI display greater curvature than COPD and LC. For both IHD and stroke, we present IER functions for three age groups (25–20, 50–55, and 80+) with decreasing relative risk for increasing age.

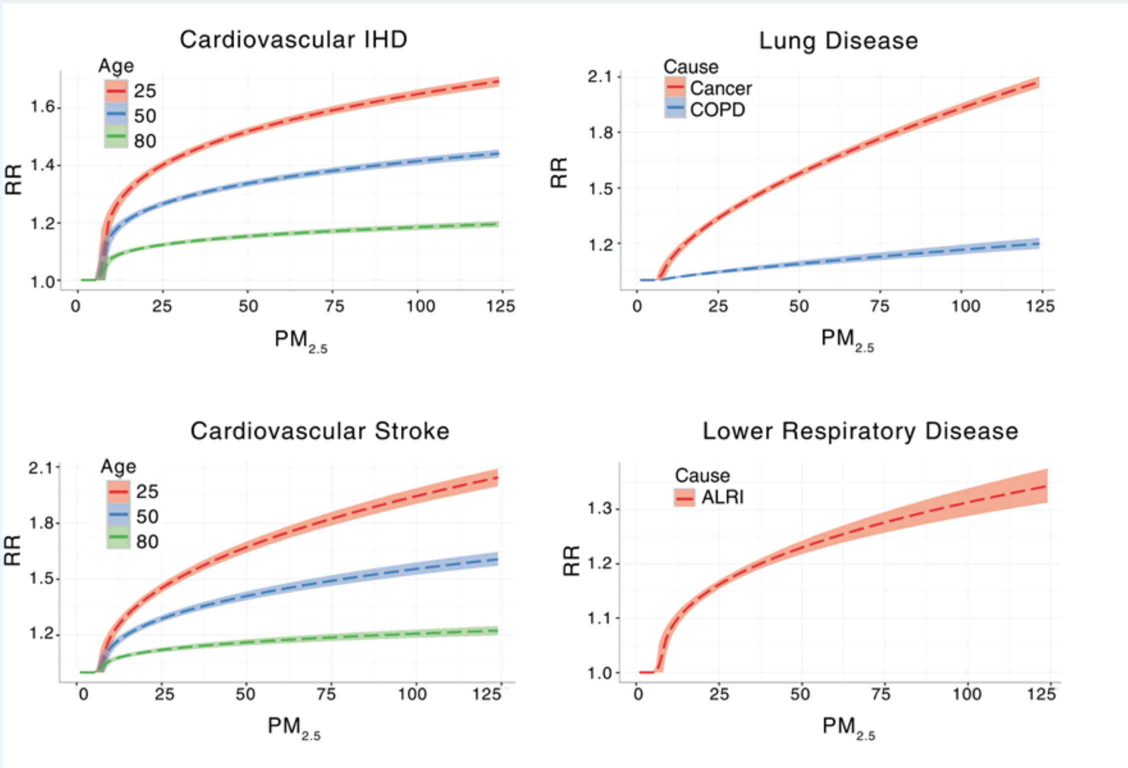


Figure. IER functions for cardiovascular IHD, LC and COPD, cardiovascular stroke, and acute LRI. Curves depict the central estimate of the IER (dashed line) and their uncertainty (shaded area). Note the relative risk = 1 for PM2.5 concentrations from 0 to 5.9 µg/m3 (lower bound of the TMREL uncertainty distribution) (A. Cohen et al. Submitted).

STATUS AND TRENDS OF AIR QUALITY IMPACTS ON HEALTH IN CHINA

Exposure to $PM_{2.5}$ contributed to 916,000 deaths in 2013 (10% of total deaths in China), a 59% increase from 1990. 571,000 male deaths and 345,000 female deaths were attributed to exposure to $PM_{2.5}$ in 2013. In China in 2013, $PM_{2.5}$ was responsible for 17% of IHD deaths, 19% of stroke deaths, 37% of LC deaths, 8% of COPD deaths, and 18% of acute LRI deaths.

Exposure to $PM_{2.5}$ was the risk with the eighth highest disease burden for disability-adjusted life years (DALYs) and the fourth highest ranking risk factor for deaths in China in 2013 (Figure 1a,b). More than 5% (5.4%) of total DALYs in 2013 were attributable to exposure to $PM_{2.5}$. This represents an increase in the disease burden attributable to ambient $PM_{2.5}$ since 1990 when ambient $PM_{2.5}$ accounted for 3.8% of total DALYs and ranked 8th. The main diseases affected by $PM_{2.5}$ in China are cardiovascular diseases, including IHD and stroke, which together account for 64% of the DALYs attributable to $PM_{2.5}$ exposure in China, followed by LC which was responsible for 24%. In 1990, cardiovascular diseases were also the main diseases affected by $PM_{2.5}$ in China with LRIs being were the second largest contributor.

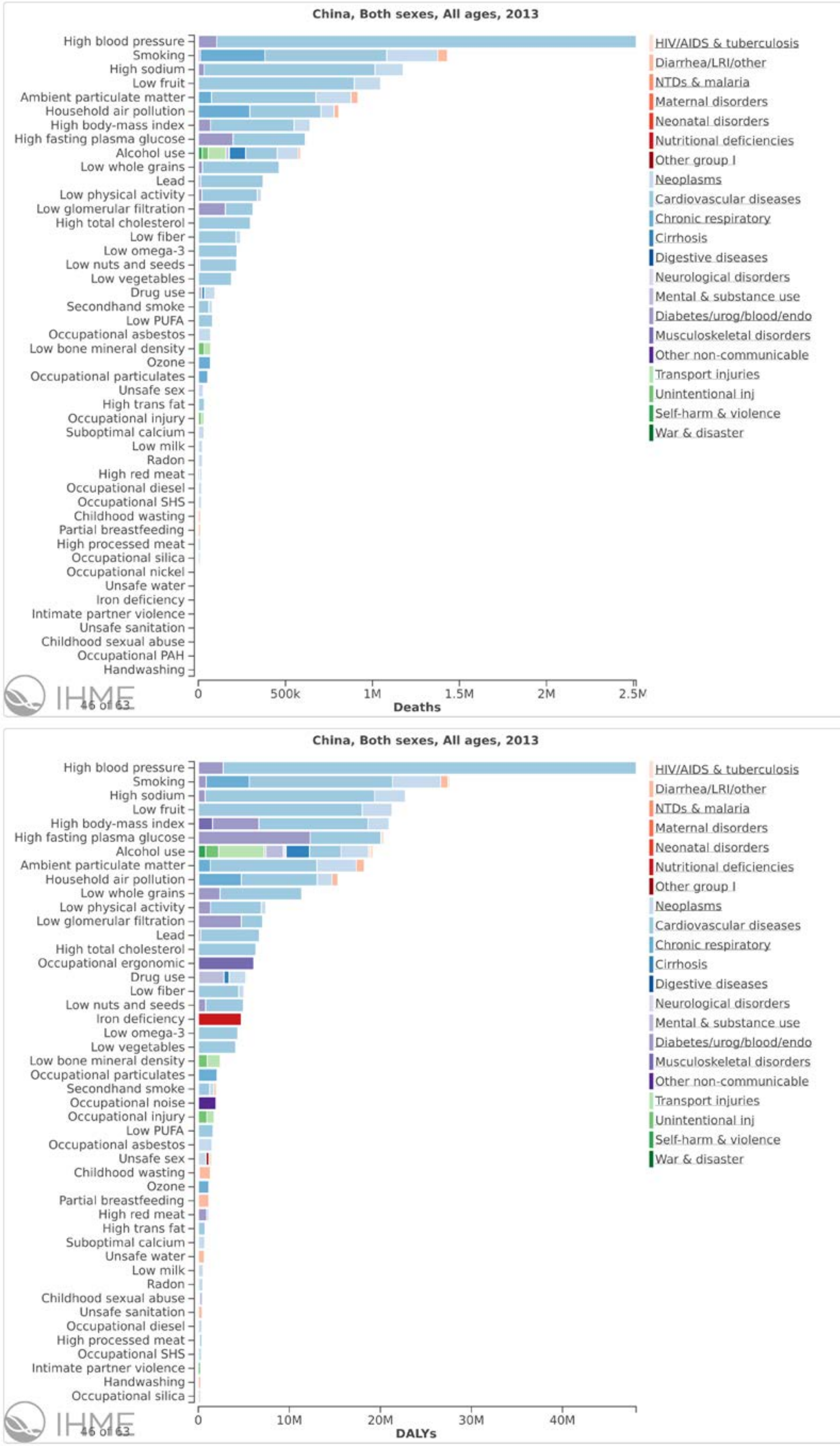


Figure 1 (a) Top Chinese mortality risk factors 2013 and (b) Top Chinese DALY risk factors 2013.

At a provincial level, differences in total deaths or DALYs are driven mainly by population size; however, comparison of DALY rates attributable to $PM_{2.5}$ indicate higher levels in Xinjiang, Guizhou, Henan, Hebei, and Shandong, compared with other provinces (Figure 2).

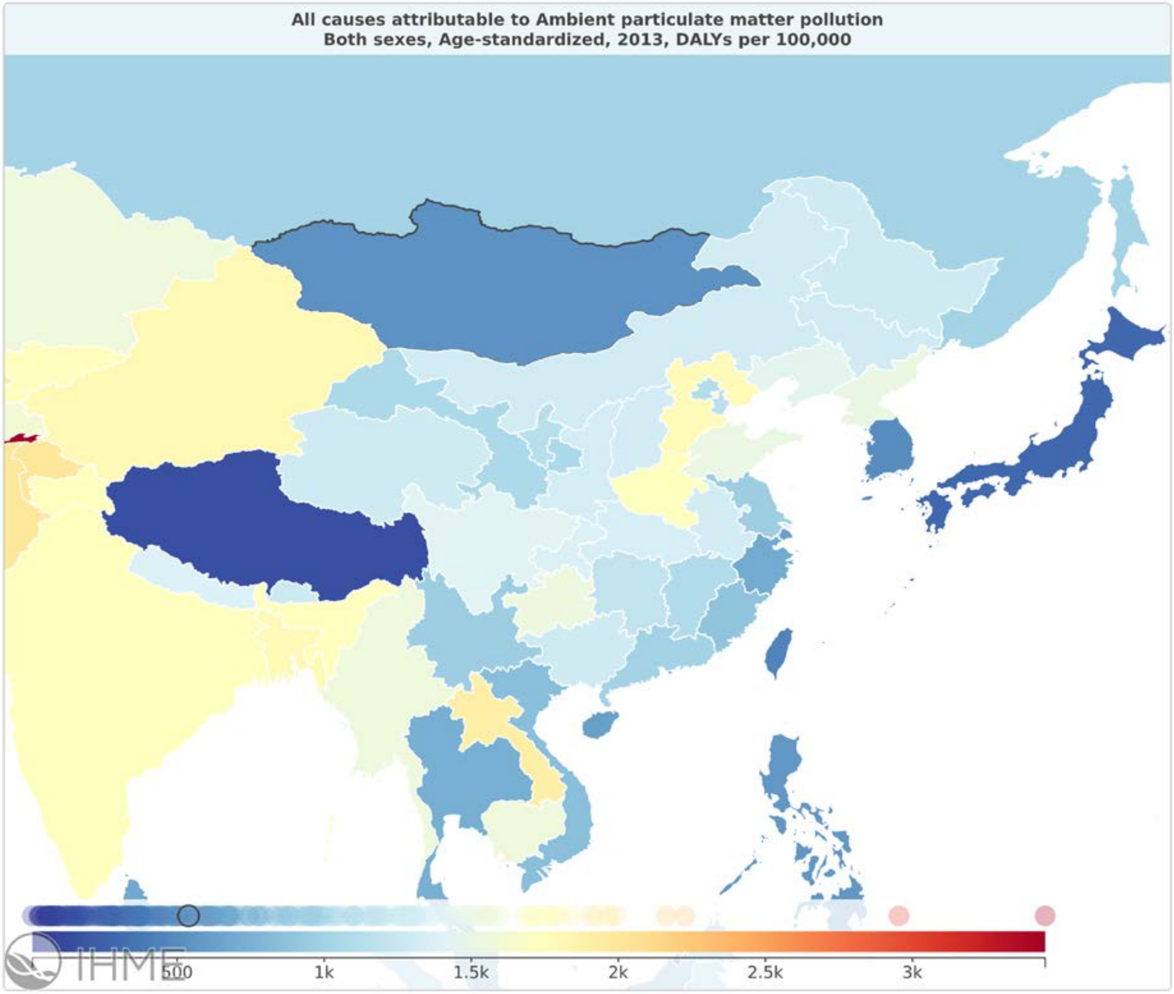


Figure 2. Chinese age-standardized DALY rates attributable to $PM_{2.5}$ exposure by province 2013.

A strong increasing trend in total burden and death accelerated after 2000 (Figure 3a,b). In 1990 there were 577,000 $PM_{2.5}$-associated deaths, increasing (19%) to 686,000 in 2000, followed by a 34% increase to 916,000 deaths in 2013. The annual rate of change for $PM_{2.5}$-attributable deaths between 2000 and 2013 was approximately 2.6%.

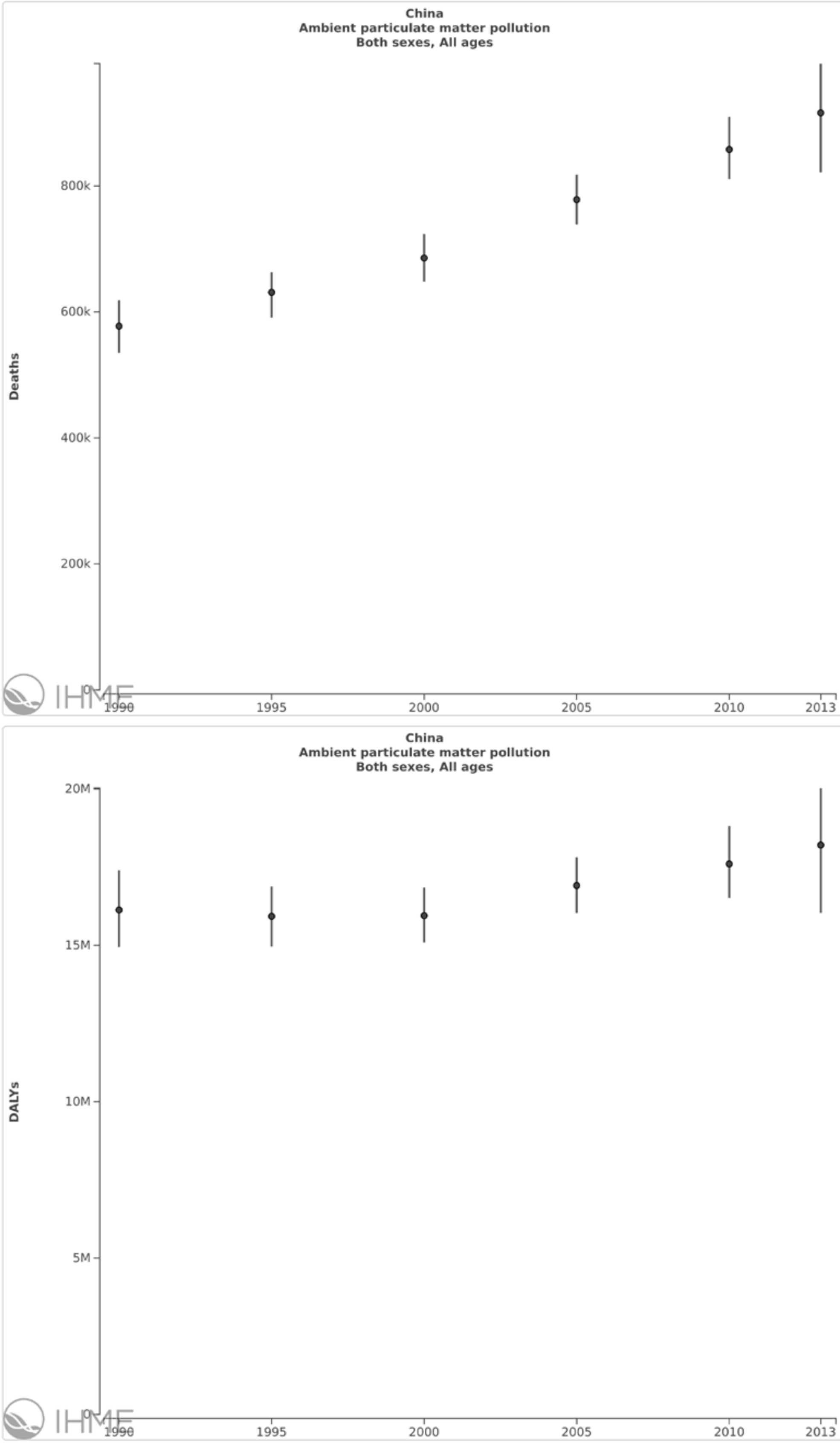


Figure 3 (a) Chinese deaths and (b) Chinese DALYs attributable to $PM_{2.5}$, 1990–2013.

Four factors play a role in the upward trend in $PM_{2.5}$-attributable deaths, including increases in $PM_{2.5}$ exposures, population growth, population aging, and changes in the prevalence of diseases affected by air pollution.

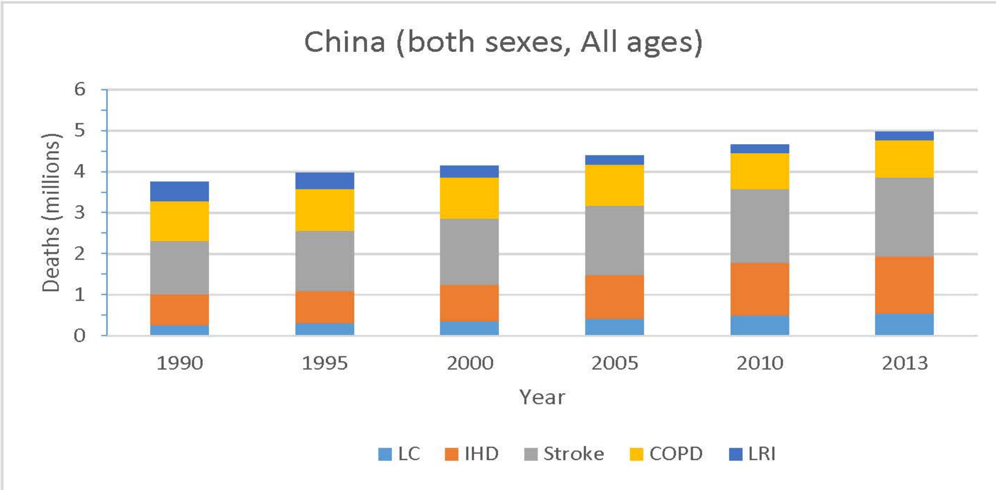


Figure 4. Total deaths in China from diseases caused by exposure to $PM_{2.5}$. Data downloaded from GBD Compare (October 28, 2015).

The total numbers of deaths from diseases caused by exposure to $PM_{2.5}$ increased from 1990 to 2013 (Figure 4). Deaths per capita for all ages increased slightly between 1990 and 2000, from 50 deaths per 100,000 to 53 per 100,000 in 2000, but rose to 66 deaths per 100,000 in 2013. However, the age-standardized death rates show a different picture. Since 1990 there has been a modest decrease, from 80 deaths per 100,000 to 71 per 100,000 in 2013, with an annual decline of about 0.5% (Figure 5). A more substantial decrease in the rate of attributable DALYs has occurred over this same period, driven largely by reductions in the incidence of LRIs.

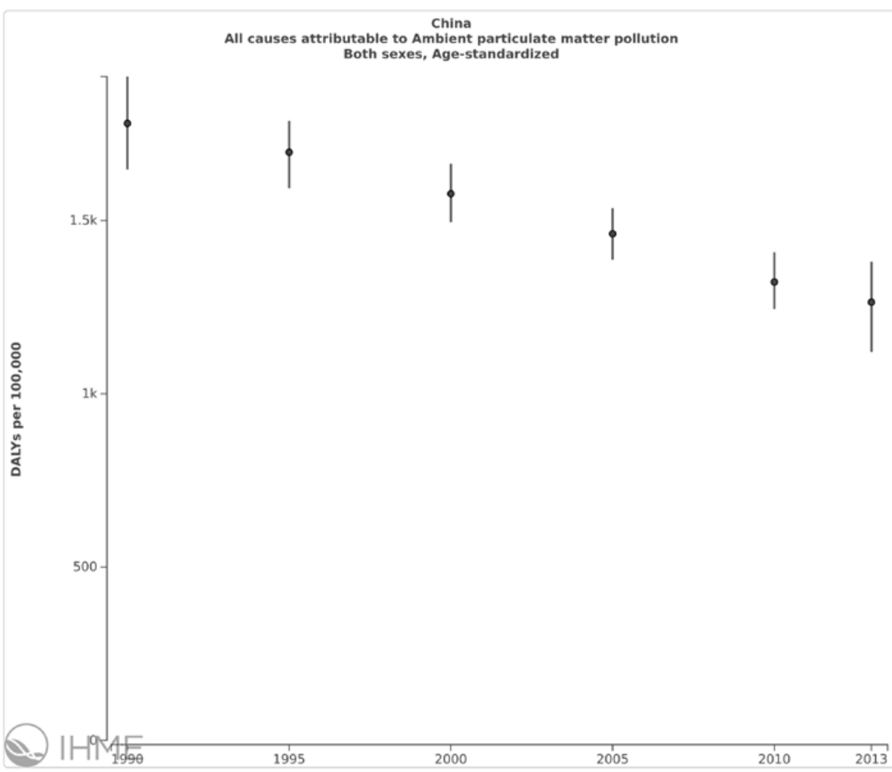


Figure 5. Chinese DALYs per 100,000 persons attributable to $PM_{2.5}$ 1990–2013.

Overall, the fraction of cause-specific mortality attributable to $PM_{2.5}$ in China has not improved during the past 23 years. For example, the population attributable fraction[16] for IHD increased from 16.6% in 1990 to 17.0% in 2013 with increases also for stroke, COPD, LC, and LRI. This trend mirrors the increasing levels of annual average $PM_{2.5}$ over this interval discussed above.

AIR QUALITY MANAGEMENT IN CHINA

China has recently instituted policies to address growing levels of air pollution. The Chinese government has set a target to reduce CO_2 emissions per unit GDP by 40%–45% in 2020 compared with 2005 levels (Wang and Hao 2012). In the Eleventh Five-Year Plan for Environmental Protection, China set a target to reduce energy use per unit GDP and national SO_2 emissions by 20% and 10%, respectively (The State Council of the People's Republic of China 2006). In the Twelfth Five-Year Plan for Environmental Protection, China aimed to reduce energy use per unit GDP by 16%, and national SO_2 and NO_x emissions by 8% and 10%, respectively (The State Council of the People's Republic of China 2012). In September 2013, China announced the Clean Air Action Plan, promising to significantly reduce PM concentrations across the country by 2017, with more aggressive targets in metropolitan regions (The State Council of the People's Republic of China 2013). In order to achieve the targets above, China has enforced a series of control measures in various economic sectors. For power plants, small units ($\leq$ 100 MW) have almost been phased out (NDRC 2011; The State Council of the People's Republic of China 2012). The vast majority of coal-fired power plants have been equipped with high efficiency flue gas desulfurization (FGD) facilities, and flue gas denitrification facilities are penetrating with a dramatic speed (S.X. Wang et al. 2014). However, satellite observations of SO_2, which were used to model emissions from 26 isolated coal-fired power plants in China, indicated that the average lag time between installation and effective operation of FGD facilities at these power plants was around two years, with no indication of operational FGD facilities before the year 2008 (Wang et al. 2015). For the industrial sector, air quality management in China focused efforts on increasing the energy efficiency of out-of-date production technologies (Zhao et al. 2013c,d). China has been controlling PM emissions from industrial sources since the late 1980s. Most industrial boilers were historically equipped with wet scrubbers and cyclone dust collectors, while high efficiency fabric filters began to penetrate the market recently (Lei et al. 2011; Zhao et al. 2013a). During 2010–2014, a series of new industrial emission standards have been released, and most of them are comparable to the most stringent ones in the world (S.X. Wang et al. 2014). For the residential sector, by the end of 2006, 96% of China's newly-built buildings have complied with the energy-saving standard for the design of buildings released in 1996 (Tsinghua University Building Energy Research Center [THUBERC] 2009). A more stringent standard was released in 2010 (The State Council of the People's Republic of China 2012). Policies have also been planned to reduce emissions from domestic burning of biomass and coal (Appendix IV). For the transportation sector, China has issued a series of emission standards for new vehicles and engines based on European Union standards since 2000. Most recently, the Euro IV standards for light-duty vehicles and heavy-duty diesel vehicles were implemented in 2011 and 2013, respectively (S.X. Wang et al. 2014; Wu et al. 2012).

[16] The Population Attributable Fraction is the proportional reduction in burden of disease that would occur if exposure to the risk factor was reduced to a theoretical minimum risk level. The burden of disease attributable to a specific risk factor is the product of the PAF for that risk factor and the total burden of diseases linked to that risk factor (i.e., DALYs for IHD, stroke, COPD, LC, and acute LRI in the case of ambient $PM_{2.5}$).

HEI REVIEW PROCESS

A draft final version of this report was reviewed for accuracy, quality, and appropriateness of interpretation by three independent external reviewers selected by HEI for their expertise in air quality, atmospheric chemistry and modeling, and health effects. The external reviewers were: Tong Zhu (Peking University), Markus Amman (IIASA Institute), and John Evans (Harvard University). The draft was also reviewed by experts on the GBD MAPS Steering Committee. The Working Group prepared the final report in response to the comments received. The report's major findings were also presented at the American Association for the Advancement of Science in February 2016 and HEI Annual Conference in Denver in May, 2016.

ESTIMATING THE BURDEN OF DISEASE DUE TO COAL-BURNING AND OTHER MAJOR AIR POLLUTION SOURCES

The overall analytic approach in GBD MAPS has three main components that were conducted sequentially:

1. Estimation of the fractional contributions to ambient $PM_{2.5}$ from coal combustion and other major emissions sources using the nested version of the global chemical transport model GEOS-Chem (see GEOS-Chem text box).
2. Partition GBD estimates by sector using fractional contributions calculated with GEOS-Chem.
3. Combine the coal combustion and other source-specific ambient $PM_{2.5}$ from (2) together with: (a) cause-specific disease burden estimates from the GBD, and (b) IER functions describing air pollution risk estimates for adult IHD, stroke, COPD and LC; and childhood and adult LRIs, to **estimate sector contributions to disease burden** for China at the country and provincial levels.

These steps are described in more detail in the following sections.

ESTIMATION OF CONTRIBUTIONS TO AMBIENT $PM_{2.5}$ FROM COAL COMBUSTION AND OTHER MAJOR EMISSIONS SOURCES

We simulated the contribution to ambient $PM_{2.5}$ from multiple source sectors using the chemical transport model GEOS-Chem. This allowed us to quantify the sensitivity of ambient $PM_{2.5}$ levels to perturbations in source contributions. GEOS-Chem was selected for sensitivity simulations given the large user community in China and its prior use in the GBD and similar analyses. Simulation was based on emissions for a single base year (2013) with meteorology from a single year (2012) that best represented the mean for the 2010–2012 period. Four future scenarios for the year 2030 were evaluated (Table 1). For source sectors other than coal combustion, we assume that future trends in emissions reflect business-as-usual (BAU) assumptions.

Sensitivity simulations include:

- Analysis of variation across 2010–2012 meteorology with the standard simulation (STD).
- Impact of 20% NH_3 emissions reduction.
- Contribution of total coal combustion, industrial coal combustion, coal-fired power plant generation, and domestic coal combustion.
- Contribution of other source sectors (transportation, noncoal industrial, domestic biomass, open biomass, solvent use).

For sensitivity simulations, the total coal-related emissions, sector-specific coal-related emissions, and emissions from other major sectors are removed respectively from the inventory in each scenario. The global and nested grid models of GEOS-Chem were then run in sequence using the new inventories. These sensitivity simulation results therefore depict the ambient $PM_{2.5}$ concentrations with all the coal-related emission sources shut off. The differences of the standard and sensitivity simulations were analyzed to produce contributions of the coal combustion to ambient $PM_{2.5}$ concentration. The same process was repeated for other major source sectors. By comparing the difference in simulated ambient concentrations between the standard and sensitivity simulations, we therefore consider in our analyses the complex nonlinear relationships between emissions and ambient concentrations and the nonlinear atmospheric chemistry affecting particle formation.

Text Box: GEOS-Chem and the High-Resolution Simulation for China

GEOS-Chem (*www.geos-chem.org*) is a chemical transport model that is used by about one hundred research groups around the world. The model solves for the temporal and spatial evolution of aerosols and gaseous compounds using meteorological data sets, emission inventories, and equations that represent the physics and chemistry of the atmosphere. The simulation of $PM_{2.5}$ includes the sulfate-nitrate-ammonium-water system (Park et al. 2004), primary (Park et al. 2003) and secondary (Henze et al. 2006, 2008; Liao et al. 2007; Pye et al. 2010) carbonaceous aerosols, mineral dust (Fairlie et al. 2007), and sea salt (Alexander et al. 2005). The boundary fields are provided by the global GEOS-Chem simulation with a resolution of 4° latitude and 5° longitude (~445 × 553 km at equator), which are updated every three hours. The East Asia nested version of GEOS-Chem used here covers the area from 70°E to 150°E and from 10°S to 55°N, and was developed by Chen et al. (2009) to resolve the domain of East Asia at a resolution of 0.5° × 0.67° (~56 × 74 km at equator) with dynamic boundary conditions using meteorological fields from the NASA Goddard Earth Observation System (GEOS-5). Although a high-resolution (0.25° × 0.3125°, ~28 × 35 km at equator) domain has been developed for East Asia, it was still in the early stages of implementation and therefore was not used. Ambient ozone also contributes to the disease burden attributable to air pollution in the GBD. Although ozone precursors are emitted in coal combustion and also simulated in GEOS-Chem, we did not estimate the contribution of coal combustion to disease burden via ozone production. For GBD 2013, the impact of ozone on mortality in China was 7% of that from $PM_{2.5}$ and therefore the relative importance of coal combustion to disease burden via ozone vs that via $PM_{2.5}$ is also likely to be small. In addition, coal combustion emits additional pollutants (e.g., NO_2), which are associated with adverse health impacts, but which were not included in the GBD or in this analysis. The GEOS-Chem model has been widely applied to study $PM_{2.5}$ over China (e.g., Jiang et al. 2015; Kharol et al. 2013; Wang et al. 2014; Zhang et al. 2015; Xu et al. 2015) including to relate satellite observations of aerosol optical depth to ground-level $PM_{2.5}$ for the GBD assessment (Brauer et al. 2012, 2016; van Donkelaar et al. 2010, 2015).

The GEOS-Chem model has fully coupled ozone-NO_x-hydrocarbon chemistry and aerosols including sulfate (SO_4^{2-}), nitrate (NO_3^-), ammonium (NH_4^+) (Park et al. 2004; Pye et al. 2009), organic carbon (OC) and black carbon (BC) (Park et al. 2003), sea salt (Alexander et al. 2005), and mineral dust (Fairlie et al. 2007). For these simulations we also included the SO_4^{2-} module introduced by Wang et al. (2014). Partitioning of nitric acid (HNO_3) and ammonia between the gas and aerosol phases is calculated by ISORROPIA II (Fountoukis and Nenes 2007). We have corrected the too shallow nighttime mixing depths and overproduction of HNO_3 in the model following Heald et al. (2012) and Walker et al. (2012). Secondary organic aerosol formation considers the oxidation of isoprene (Henze and Seinfeld 2006), monoterpenes and other reactive volatile organic compounds (ORVOCs) (Liao et al. 2007), and aromatics (Henze et al. 2008). We assume that the organic mass:organic carbon ratio is 1:8 and relative humidity is 50% for $PM_{2.5}$ in China.

Figure 6 depicts an overview of the process to generate coal combustion contributions.

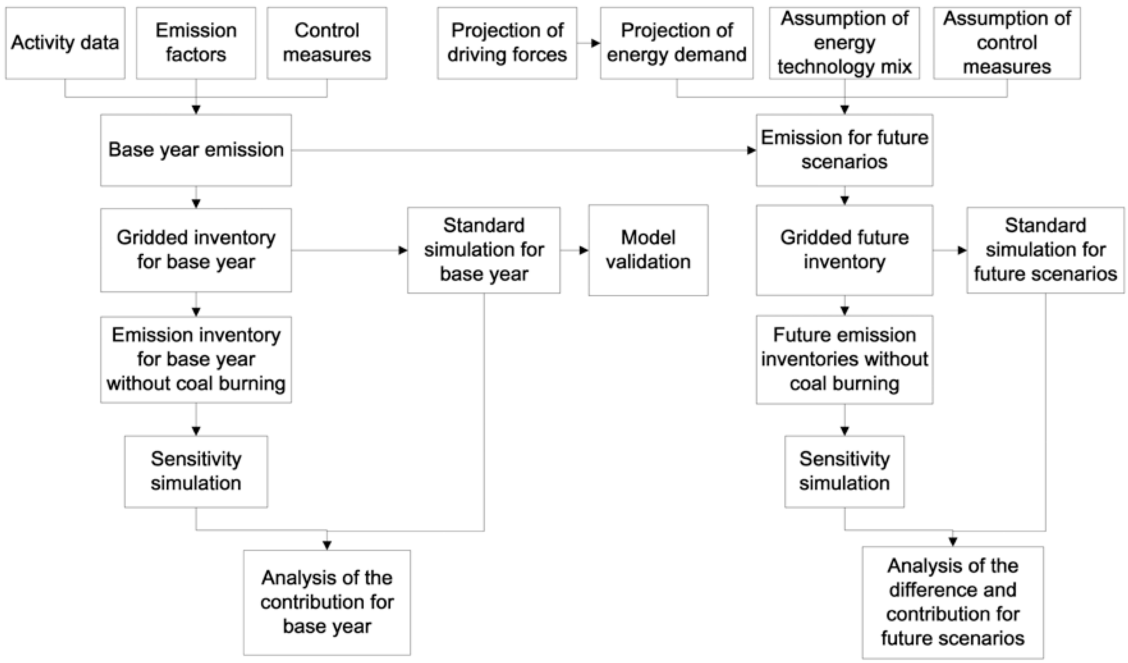


Figure 6. Technology roadmap of contribution estimate from coal combustion.

Standard and sensitivity simulations were conducted using the chemical transport model GEOS-Chem (version v9-01-03) to estimate the coal-combustion (and other sector) contributions to current and future ambient $PM_{2.5}$.

The meteorology for the year 2012 and emissions for the year 2013 were used as the baseline, and four future scenarios of the year 2030 are evaluated as well, comprising BAU[1], BAU[2], PC[1], and PC[2] (Table 1). Of specific relevance to this project, a unit-based method is used to estimate emissions from large point sources including coal-fired power plants, iron and steel plants, and cement plants. The emissions of these large point sources are based on the information about each facility, which significantly improves the accuracy and spatial resolution of emissions (Zhao et al. 2008).

Table 1. Definitions of Future Scenarios

	End-of-Pipe Emission Control Policy	
Energy policy	[1] Twelfth Five-Year Plan for Environmental Protection	[2] Maximum Feasible Emission Controls Regardless of Cost
BAU: Current Legislation and Implementation Status as of end of 2012	BAU[1]	BAU[2]
PC: Additional energy-saving policies will be implemented, including life style changes, structural adjustments and energy efficiency improvements.	PC[1]	PC[2]

Standard simulations were first conducted using the emission inventories of the base year and future scenarios. In order to obtain the boundary fields, the global model of GEOS-Chem is first conducted at a resolution of 4° × 5° on a global scale. Using the acquired boundary fields, the nested grid capability of GEOS-Chem is then used to conduct the simulations over East Asia. Results of the nested grid model of GEOS-Chem show the ambient $PM_{2.5}$ concentrations in current and future scenarios in China.

We developed an emission inventory of SO_2, NO_x, PM_{10}, $PM_{2.5}$, BC, OC, NMVOC, and NH_3 for China for the year 2013. This was based upon the model intercomparison study (MICS) Asia III[17] emissions inventory for 2010 and updated to 2013. Specifically, an emission factor method was used to calculate air pollutant emissions, as described in detail in previous work (Wang et al. 2014; Zhao et al. 2013b,c,d). The emissions from each sector in each province were calculated from the activity data (energy consumption, industrial products, solvent use, etc.), technology-based emission factors, and penetrations of control technologies. The activity data and technology distribution for each sector were derived based on Chinese statistics (National Bureau of Statistics [NBS] 2014a,b,c), a wide variety of Chinese technology reports (China Electricity Council 2011; Energy Research Institute in China 2009, 2010; THUBERC 2009), and an energy-demand modeling approach. The emission factors were described in Zhao and colleagues (2013b). The penetrations of removal technologies were updated to 2013 according to the governmental bulletin (Ministry of Environmental Protection of China [MEP] 2014), the evolution of emission standards, and a variety of technical reports. The method and parameters for spatiotemporal distribution of anthropogenic emissions, speciation of NMVOC emissions, and calculation of biogenic NMVOC emissions are described by Wang and colleagues (2011). A Monte Carlo uncertainty analysis was performed on the emission inventory, as described in Zhao and colleagues (2013c) and Wang and colleagues (2014). Table 2 shows the uncertainty analysis of the emissions in China.

Table 2. Results of the Uncertainty Analysis of the Emissions in China. The last line shows the average 90% confidence intervals of the total emissions.

	NO_x	SO_2	$PM_{2.5}$	NMVOC
Power plants	±34%	±30%	±31%	—
Industrial sector	±41%	±49%	±53%	±63%
Residential sector	±55%	±51%	±68%	±65%
Transportation sector	±66%	±48%	±52%	±57%
Solvent use sector	—	—	—	±78%
Other sectors (mainly biomass open burning)	±177%	±179%	±216%	±184%
Total emissions	[−31%, 44%]	[−29%, 45%]	[−39%, 49%]	[−42%, 67%]

Meteorology Sensitivity

Given the emphasis on this analysis on current (year 2013) and future emissions rather than year-to-year variation in ambient pollution a typical meteorologic year was selected for all simulations. Sensitivity analyses comparing ambient $PM_{2.5}$ concentrations simulated with 2010, 2011, and 2012 meteorology

17 The 2010 inventory has already been incorporated into the HTAP (Hemispheric Transport of Air Pollution assessment). Future emissions scenarios are similar to those used in the GAINS (Greenhouse Gas and Air Pollution Interactions and Synergies model).

were compared with a simulation in which the mean meteorology of the same three years was used. Comparisons of simulated ambient $PM_{2.5}$ using meteorology from 2012 indicated the smallest difference in comparison with those based upon the 2010–2012 mean for the entire country of China, and they were significantly better for eastern China, the area with the highest overall $PM_{2.5}$ concentrations. Mean monthly differences in simulated $PM_{2.5}$ concentrations ranged from −0.23 to −6.6 $\mu g/m^3$ when comparing meteorology from 2012 with the 3-year mean. In eastern China differences were larger, ranging from +34.8 to −35.4 $\mu g/m^3$ in October and November, respectively. Further, analyses of mean annual population-weighted $PM_{2.5}$ concentrations for each province indicated that year 2012 meteorology resulted in simulated $PM_{2.5}$ exposures that were closest to those simulated with the 2010–2012 mean meteorology for the most heavily populated provinces. Accordingly, we selected year 2012 meteorology for all simulations. Similar analyses were conducted for India with little variability in deviation from the 2010–2012 mean seen for the different years and deviations for 2012 found to be reasonably low (ranging from +9.3 to −1.6 $\mu g/m^3$).

ESTIMATION OF AMBIENT CONCENTRATIONS ATTRIBUTABLE TO COAL COMBUSTION AND OTHER SOURCES

The spatially resolved fractional contributions of coal combustion and other sources estimated with GEOS-Chem simulations were then multiplied by the high-resolution ambient $PM_{2.5}$ concentration estimates developed for the GBD (2013) to estimate ambient $PM_{2.5}$ concentrations attributable to coal combustion and other sources. The GBD concentration estimates: (a) combine satellite-based estimates using GEOS-Chem to provide information on the relationship between aerosol optical depth with surface $PM_{2.5}$, (b) TM5–FASST (Fast Scenario Screening Tool) simulations, (c) available annual average PM measurements (for 2010–2013) at 0.1° × 0.1° resolution.

Using these estimates explicitly incorporates available surface measurements of $PM_{2.5}$ for the locations of interest. For China, this includes approximately 80 $PM_{2.5}$ measurements as well as ~300 additional locations with PM_{10} measurements that were used to estimate $PM_{2.5}$ based on $PM_{2.5}$:PM_{10} ratios (for example, most of the 80 $PM_{2.5}$ measurement locations also have PM_{10} measurements to derive local ratios). The incorporation of $PM_{2.5}$:PM_{10} ratios vs direct $PM_{2.5}$ measurements is directly included as a variable in the above model (Brauer et al. 2016) and more generally represents a tradeoff between reduced absolute accuracy vs improved spatial representativeness.

ESTIMATION OF COAL AND OTHER SOURCE SECTOR CONTRIBUTIONS TO DISEASE BURDEN

The spatially resolved estimated levels of coal combustion and other source-specific ambient $PM_{2.5}$ concentrations were then used with IER functions (Burnett et al. 2014) (see IER text box) describing air pollution risk estimates for adult IHD, stroke, COPD and LC; and childhood and adult LRIs, to **estimate** sector contributions to disease burden for China — for the country as a whole and for each province. Based on prior analyses conducted for GBD 2010 for household cooking and heating as well as road transportation, the nominal assumption is that the proportional contribution of a source sector, p, to the air pollution disease burden is a simple proportion of that sector's contribution, represented by the population attributable fraction, PAF(p), to ambient $PM_{2.5}$. That is, PAF(p) = PAF × p, where PAF is the population attributable fraction associated with $PM_{2.5}$. It has been shown (see Appendix II) that this assumption is mathematically equivalent to averaging the PAF over all possible changes in concentration of size $p \times z$ within the concentration interval from the counterfactual to z. We have employed this strategy for estimating the contribution for overall burden associated with $PM_{2.5}$ exposure attributable to any single, or multiple sources, by decomposing the attributable risk of $PM_{2.5}$, as measured by the IER, into specific sources proportional to their contribution to total $PM_{2.5}$ concentration. This method of source attribution is based on the average derivative of the IER evaluated at concentration *z*, over the concentration range from the counterfactual to *z*. We note that we are not linearizing the IER itself, only calculating the average derivative. We take this approach since we do not know where in the exposure distribution such a change of $p \times z$ occurs as we are exposed to fine PM from all sources simultaneously. We use the proportion source PAF approach when evaluating both current and future scenarios.

We suggest that this approach provides more robust findings in that it insensitive to the order in which each source is removed from total concentration. It thus provides a more direct method of communicating results compared to removing the source contribution over the range *z* to $(1-p) \times z$. Decision makers could be confused when multiple sources are considered simultaneously with a myriad of burden estimates presented, each depending on the order of source control. The proportional source approach also has the advantage that the sum of burden estimates from all sources equals the burden from ambient $PM_{2.5}$ exposure. Removing exposure from a single source at a time does not have this highly desirable property. We suggest that our proportional source approach provides additional clarity when communicating with policy and decision makers. These methods are discussed in greater detail in Appendices I and II.

UNCERTAINTY

Although estimates of attributable disease burden are presented with corresponding uncertainty intervals, we did not conduct a formal evaluation of uncertainty in this analysis. The overall uncertainty in the burden estimates that we calculated is a function of uncertainty in the $PM_{2.5}$ estimates, the IER, and baseline health rates. For example, in the results we present the population-weighted mean concentrations for ambient $PM_{2.5}$ and $PM_{2.5}$ attributable to specific source sectors with 95% uncertainty intervals around their mean values. This uncertainty is estimated by sampling 1000 draws of each grid cell value of the mean of the chemical transport model CTM and satellite-based concentration estimates in combination with the calibration parameters and the standard error of the calibration function. This uncertainty is propagated along with uncertainty in the IER and uncertainty in the baseline disease rates to arrive at uncertainty in attributable burden. No uncertainty in the attribution of ambient $PM_{2.5}$ to the various source sectors was included, as this would be a major undertaking beyond the scope of this analysis. Uncertainty in the emissions estimates based on Monte Carlo simulation ranged from −42% to +67% depending on species, as described above. We have attempted to provide some context to the source attribution uncertainty via limited comparisons with source-apportionment analyses, where available, and have described it in more detail below. As in the GBD, all estimates are conducted by building up from the highest resolution inputs to limit spatial misalignment. Specifically, exposure estimates and population are at the $0.1° \times 0.1°$ grid cell level and are used with the IER to estimate the PAF. Source contributions are at the level of the simulations; ~56 × 74 km. Baseline disease burden is at the provincial level, which is then aggregated to national-level estimates.

CURRENT BURDEN OF DISEASE FROM COAL AND OTHER MAJOR SOURCES

EMISSIONS ESTIMATES FOR CURRENT COAL AND OTHER MAJOR SOURCES

In 2013, the anthropogenic emissions of SO_2, NO_x, PM_{10}, $PM_{2.5}$, BC, OC, NMVOC, and NH_3 in China were estimated to be 23.1 Mt, 25.6 Mt, 16.5 Mt, 12.2 Mt, 1.95 Mt, 3.42 Mt, 24.2 Mt, and 9.62 Mt, respectively. Provincial-level emissions estimates of $PM_{2.5}$, SO_2, NO_x, NH_3, and volatile organic compounds (VOCs) by sector[18] are given in Figures 7–11. Detailed tabulations are provided in Appendix III.

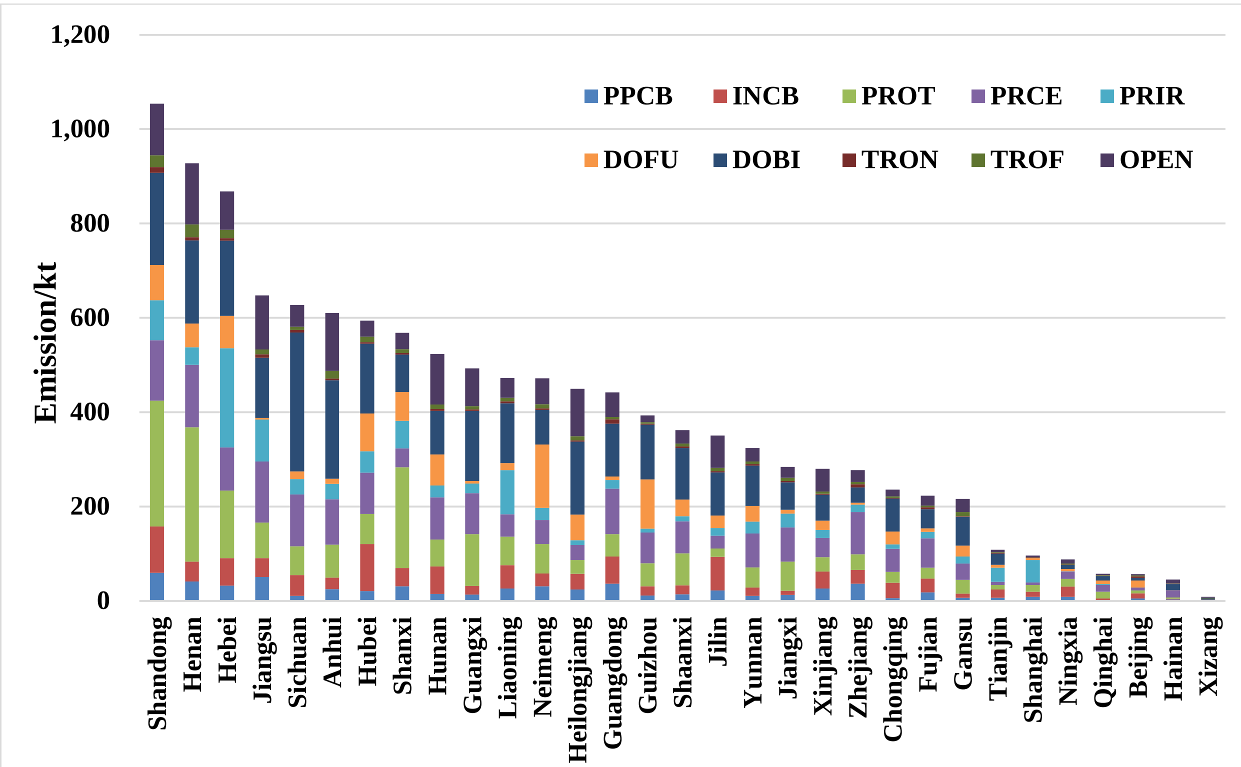


Figure 7. $PM_{2.5}$ emission in kilotons (kt) by sector for each province (2013).
(Neimeng = Inner Mongolia, Xizang = Tibet)

[18] Emissions sectors included in Figures 7–11: AGRL = agricultural livestock, AGRF = agricultural fertilizer, DOFU = domestic fuel use (nonbiomass), DOBI = domestic biomass, DOCB = domestic combustion, DOSO = domestic solvent use, DOTH = domestic and other, INCB = industrial combustion, OPEN = open burning, PPCB = power plant combustion, PRCE = cement emissions, PRIR = iron and steel emissions, PROT = industrial processes, PRSO = industrial solvent use, TRON = transport emissions on-road, TROF = transport emissions off-road.

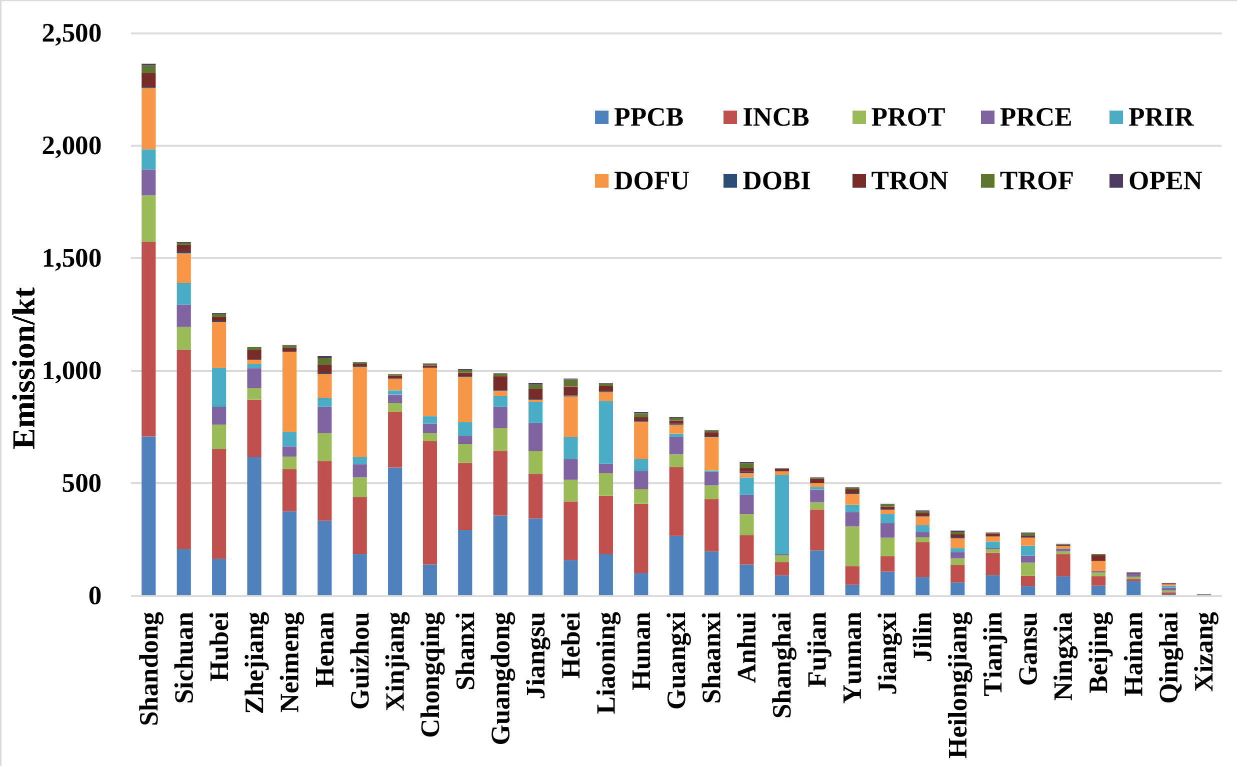


Figure 8. SO_2 emission in kilotons (kt) by sector for each province (2013).
(Neimeng = Inner Mongolia, Xizang = Tibet)

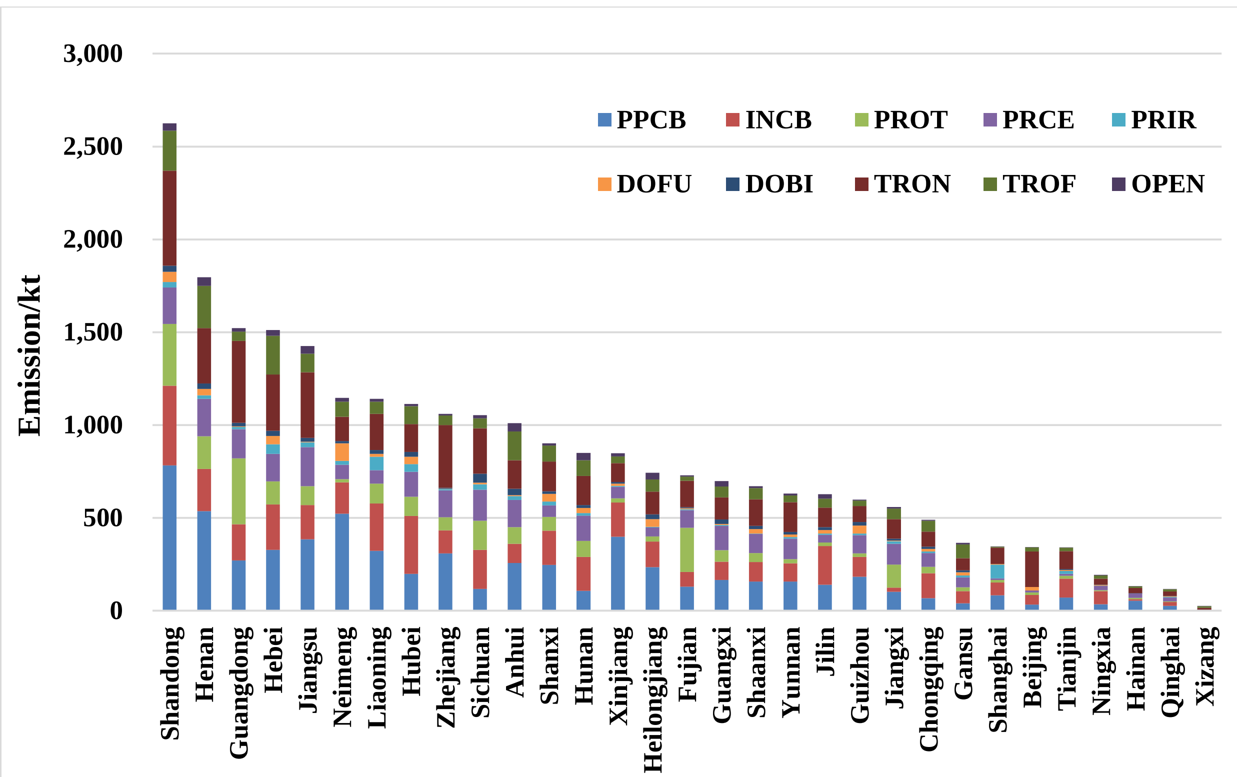


Figure 9. NO_x emission in kilotons (kt) by sector for each province (2013).
(Neimeng = Inner Mongolia, Xizang = Tibet)

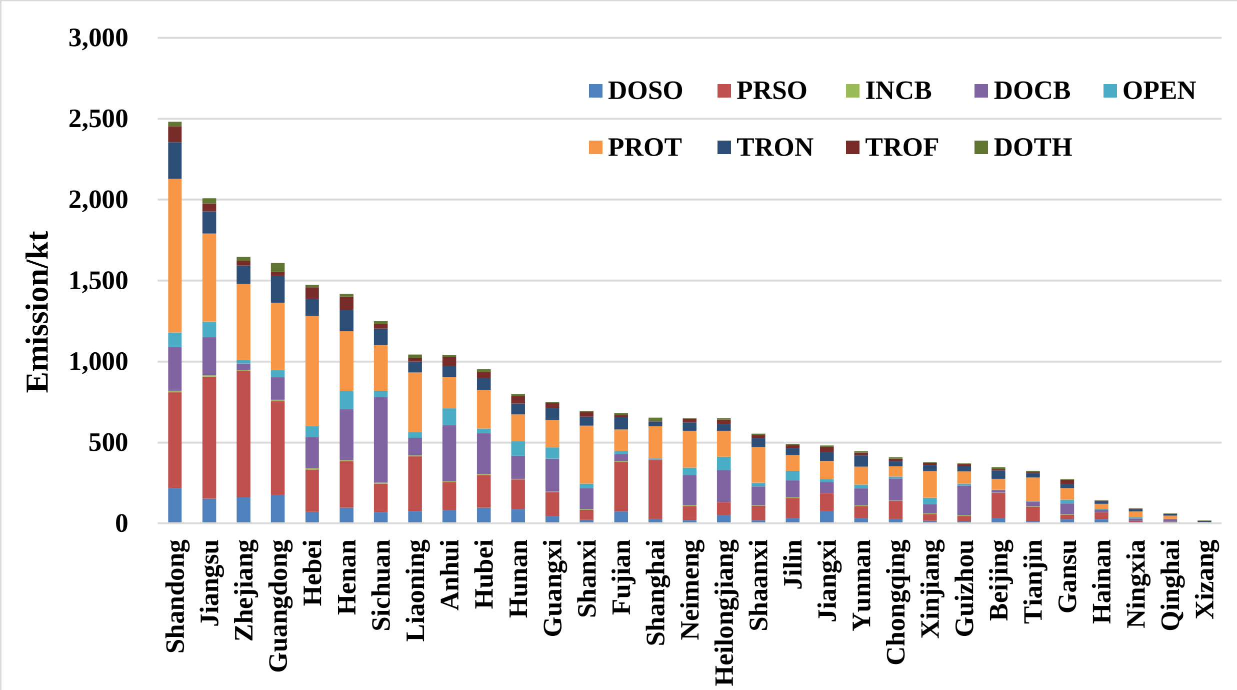


Figure 10. VOC emission in kilotons (kt) by sector for each province (2013).
(Neimeng = Inner Mongolia, Xizang = Tibet)

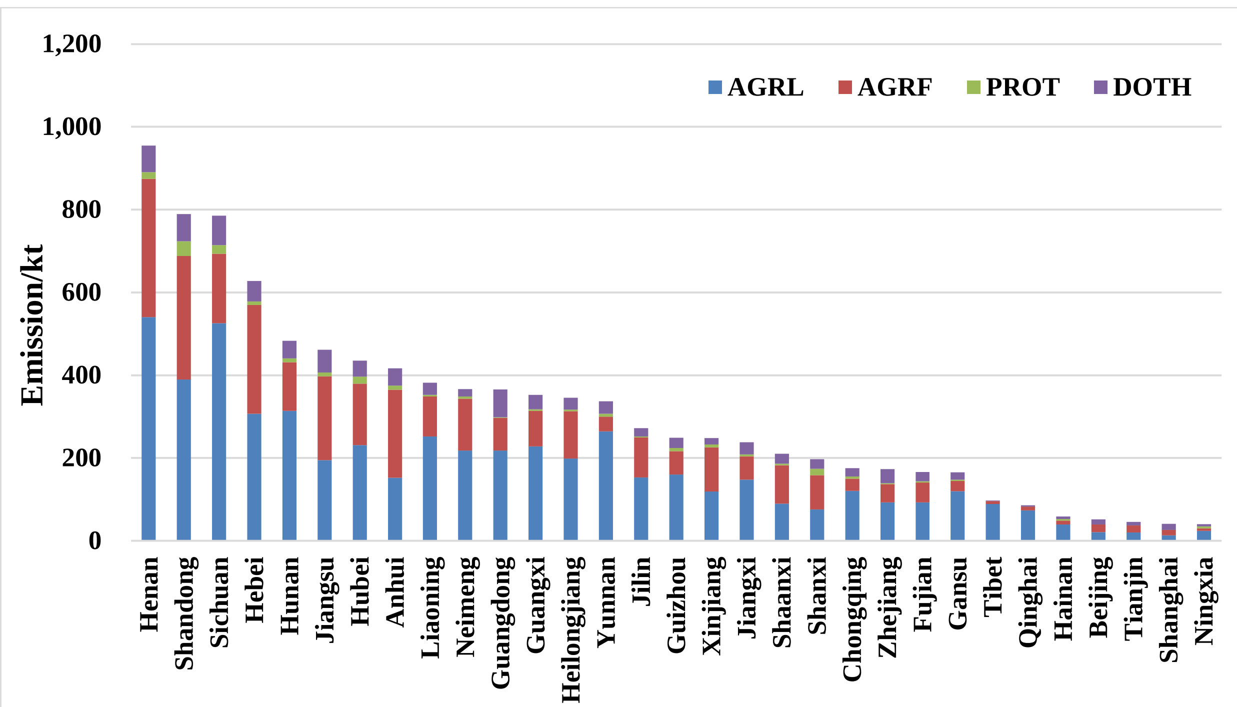


Figure 11. NH_3 emission in kilotons (kt) by sector for each province (2013).
(Neimeng = Inner Mongolia)

Evaluation of Model Performance

Our evaluations were limited given that the GEOS-Chem simulations have been extensively evaluated in East Asia against observations for $PM_{2.5}$ mass (Xu et al. 2015) and composition (Philip et al. 2014; Xu et al. 2015), and that our aim was to produce attributable disease burden estimates in a timely manner for consideration in policy development. Further, we scaled the GEOS-Chem simulations to ambient $PM_{2.5}$ estimates that incorporated information from ground measurements and remote sensing observations. In this study, we evaluated our models with respect to ground-level $PM_{2.5}$ measurements from the network of the China National Environmental Monitoring Center (CNEMC). This monitoring program was initiated in January, 2013, and initially covered 74 major cities in China; it currently comprises 338 cities at the prefecture level. The $PM_{2.5}$ monitoring followed the quality assurance/quality control required by CNEMC (constant temperature control as well as relative humidity at 50 ± 5%; details can be found at *http://www.cnemc.cn/*).

Although the simulations incorporated year 2012 meteorology and measurements were from 2013, we were specifically interested in spatial biases in the simulations. Figure 12 compares simulated and observed annual mean $PM_{2.5}$ concentrations from multiple monitors in each of 74 cities in China for the year 2013. The model reproduced the spatial distribution with a 16.3% low bias. The correlation for annual mean concentration was 0.68. The underestimate mainly appears in heavily-polluted areas in the North China Plain where observations are largely influenced by significant local emissions that the current simulation cannot capture due to its relatively coarse resolution (Zhang et al. 2012). The model performs better in South China, where the $PM_{2.5}$ concentrations are lower. However, sulfate is still underestimated and nitrate overestimated even in South China, a common problem in many models. This may reflect heterogeneous chemistry that is not well-characterized by models. The underestimation of sulfate will also systematically underestimate the contributions of coal combustion to ambient $PM_{2.5}$. The inconsistency between the meteorology used for the simulation (year 2012) and that of the observations (2013, including with particularly unfavorable meteorological conditions during January 2013) may also partly account for the discrepancies.

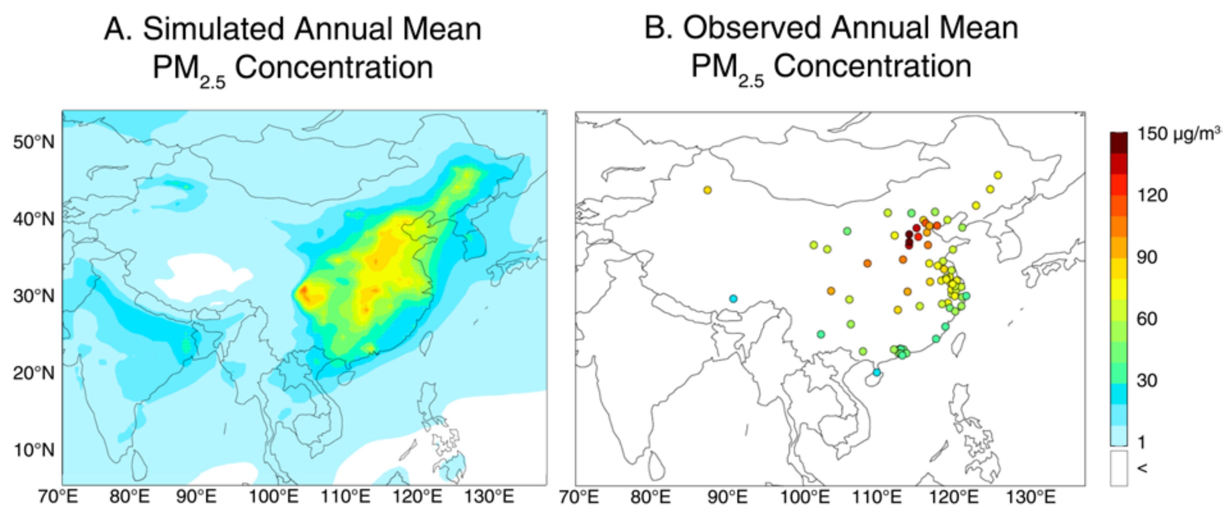


Figure 12. Simulated and observed annual mean $PM_{2.5}$ concentration.

Because we directly couple the source sector contributions from the simulations with the estimates of ambient concentration used for the GBD 2013, we also compared the spatial patterns and absolute levels from the simulations (Figure 12) with the GBD 2013 estimates. Although the methods are not directly comparable for any specific location given their different resolutions, spatial patterns and absolute levels were similar with levels generally ranging from 40–70 μg/m^3 in northeastern China with maximum values of approximately 130 μg/m^3, save for a few sporadic locations with levels above 200 μg/m^3.

Figure 13 shows the comparison between simulated and observed seasonal mean concentrations. $PM_{2.5}$ concentrations display clear seasonal variation with the highest values in winter, which is correctly reproduced by the model. The largest bias is in winter (−23.3%); the model performance is better during the other seasons, with biases between −13.3% and −10.8%. The correlations for seasonal means vary between 0.58 and 0.62. The simulations tend to underestimate $PM_{2.5}$ in northern China (between 36°N and 40°N), especially during the winter. This is probably due to the large underestimation of sulfate and OC during high-pollution episodes.

These results broadly agree with those of other evaluations of simulated aerosol concentrations in China that have been evaluated in some studies. Wang and colleagues (2013) reported annual biases of −10%, +31%, and +35% for sulfate, nitrate, and ammonia respectively, compared with observations at 22 sites in East Asia. Fu and colleagues (2012) found that annual mean BC and OC concentrations in rural and background sites were underestimated by 56% and 75%. Lou and colleagues (2014) compared seasonal mean sulfate, nitrate, ammonia, BC, and OC with measured concentrations at seven urban sites and seven remote sites, which show biases of −41%, 61%, −55%, and −69%.

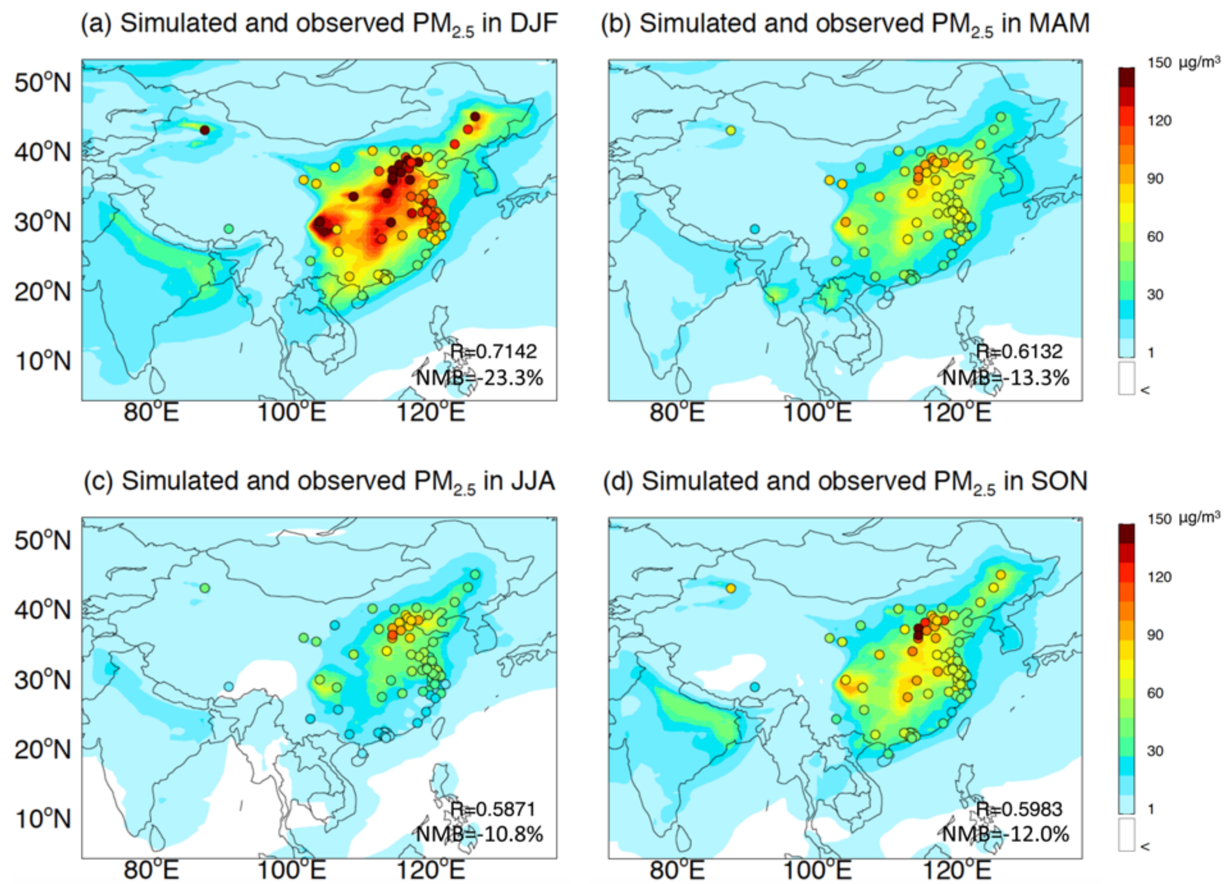


Figure 13. Simulated and observed seasonal $PM_{2.5}$ concentration. DJF = December-January-February, MAM = March-April-May, JJA = June-July-August, SON = September-October-November, NMB = normalized mean bias.

CONTRIBUTION OF COAL COMBUSTION AND OTHER MAJOR SOURCES TO $PM_{2.5}$ (2013)

Contributions of Total and Sectoral Coal Combustion to Simulated $PM_{2.5}$ Concentrations by GEOS-Chem

Figure 14 shows the simulated fractional percentage contributions from total coal-burning (Figure 14a) and different coal-burning sectors (Figures 14b–d). As these figures present percentage contributions from coal combustion and subsectors and not absolute contributions, the spatial patterns are also affected by the presence of other sources. For example, the relatively high power plant coal contribution in Xinjiang in northwestern China in part reflects the absence of other major sources, while the important absolute contribution of domestic coal combustion in many regions is masked by the presence of other sources throughout much of rural China.

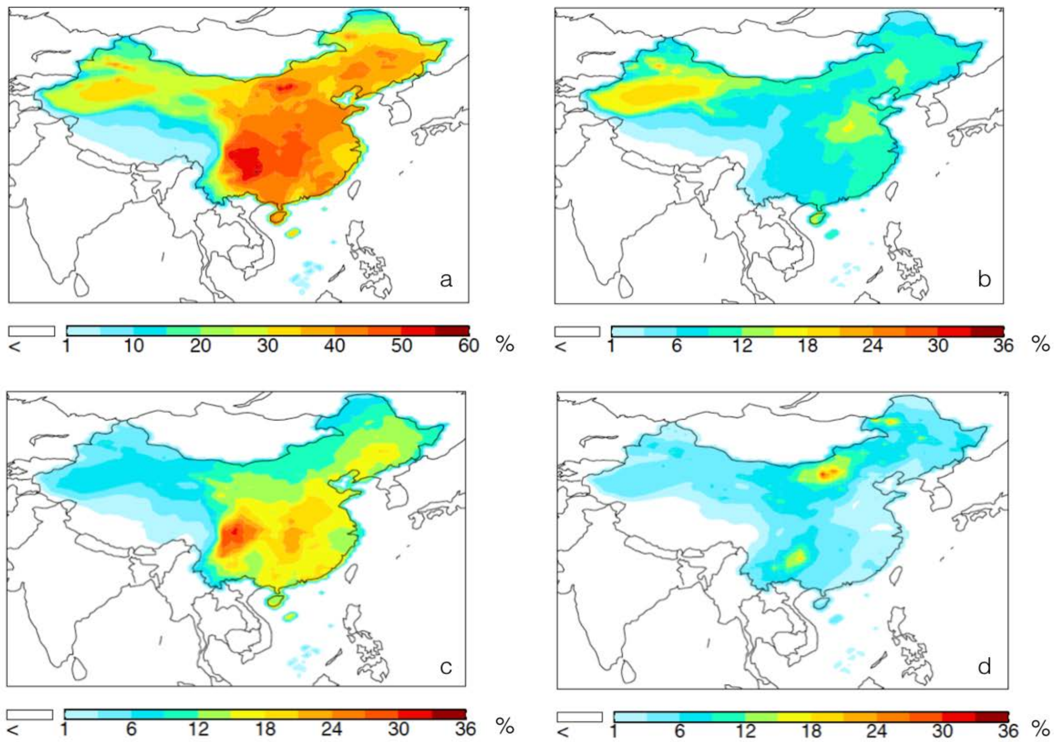


Figure 14. Simulated percentage contributions to ambient $PM_{2.5}$ in the (2013) base year from total coal-burning (a) and coal-burning in specific sectors (b-d for power plant, industrial, and domestic, respectively).

As shown in Figure 12, the ambient $PM_{2.5}$ concentration has a clear regional distribution with high values in the Sichuan Basin, North China Plain, central and eastern China. The highest concentration occurs in the Sichuan Basin. The contribution from total coal-burning (Figure 14a) has a similar spatial distribution to that of the annual mean $PM_{2.5}$ concentration, which illustrates the large influence of coal-burning on air quality. The largest contribution occurs in the southwestern city of Chengdu. The contribution from coal-burning accounts for more than 50% of the ambient $PM_{2.5}$ concentration in most areas of the Sichuan Basin, due to the dense population, high emissions, and unfavorable terrain in this area. The coal-burning contribution accounts for 40%–45% of ambient $PM_{2.5}$ in the North China Plain and 45%–50% in central China. In big cities in northeastern China and Inner Mongolia, coal-burning accounts for more than 40% in the northeast and more than 50% in cities in Inner Mongolia like Baotou and Hohhot. Coal-burning accounts for around 40% of the local $PM_{2.5}$ concentration in the northwestern city of Urumqi, although the absolute contribution is estimated to be less than 15 $\mu g/m^3$ due to the absence of other large sources of air pollutant emissions.

The contribution from each sector differs between regions. Contributions from coal-burning in power plants and industrial sources have the similar spatial distribution with the annual mean $PM_{2.5}$ concentration. As shown in Figure 14b, coal-burning in power plants has the largest contribution in the northern part of central and eastern China with the highest levels in Henan Province, where coal accounts for more than 15% of the total ambient $PM_{2.5}$ concentration. In most areas in central, eastern, and northeastern China, coal-burning in the power sector contributes 10% of the total $PM_{2.5}$. In the northwestern city of Urumqi, coal-burning in the power sector accounts for around 20% of the local $PM_{2.5}$ around, although the absolute contribution is estimated to be rather low (6 $\mu g/m^3$).

Industrial coal-burning (Figure 14c) is the largest source of ambient $PM_{2.5}$. This is especially true in the Sichuan Basin, where it accounts for around 30% of ambient $PM_{2.5}$. The largest contribution occurs in the city of Chengdu. Central China is also significantly influenced by industrial coal-burning, with the percentage contribution generally around 20%; the percentage in the southern regions is 21%–24%. Industries are densely located in these areas. In addition, the Sichuan Basin is surrounded by mountains that tend to trap the emissions and facilitate the formation of secondary pollutants. In other areas including southern, eastern, and northeastern China, the contribution of coal-burning in the industrial sector accounts for approximately 15%. Domestic coal-burning (Figure 14d) has a low contribution to ambient $PM_{2.5}$ in most areas in China, except for some individual regions in Guizhou province in southern China and in Inner Mongolia in northern China where raw coal is burned for heating. Domestic coal-burning in the above areas accounts for more than 15% of the ambient $PM_{2.5}$ in Guizhou and 25% in Inner Mongolia.

Note that as the relationship between emissions and ambient concentrations is nonlinear, and given the approach to estimate sector contributions by comparing standard (all sources) and sensitivity (a specific source sector removed), the sum of each of the coal subsectors does not add up to the simulated ambient concentration for all coal sources.

Percentage contributions to population-weighted annual average ambient $PM_{2.5}$ from coal, coal subsectors, and other major source sectors are provided in Tables 3 and 4 — for the entire country and for each province. On average, coal contributed 40% of the population-weighted exposure to $PM_{2.5}$ in China, with contributions reaching nearly 50% in Chongqing, Guizhou, and Sichuan provinces. Industrial coal contributions were more than 20% in Chongqing, Hubei, and Sichuan, while domestic coal contributions were above 10% in Guizhou and Inner Mongolia.

Table 3. Population-Weighted Percentage of Ambient $PM_{2.5}$ Attributable to Coal and Coal Sector Emissions by Province (2013).

Location	Total Coal	Industrial	Power Plant	Domestic
China	40.3	17.4	9.5	4.3
Hong Kong	22.1	10.6	4.4	1.5
Anhui	39.6	17.7	11.5	1.9
Beijing	37.5	14.2	9.6	5.9
Chongqing	49.1	22.4	6.9	7.4
Fujian	39.0	17.2	9.4	3.2
Gansu	37.4	14.4	6.5	7.5
Guangdong	33.7	15.2	7.2	2.6
Guangxi	39.1	15.6	7.6	4.3
Guizhou	49.2	15.9	6.9	12.2
Hainan	28.9	10.9	9.5	2.5
Hebei	38.9	14.9	9.7	5.3
Heilongjiang	36.9	13.3	9.7	7.4
Henan	43.2	18.1	12.8	3.1
Hubei	45.1	20.1	9.4	5.1
Hunan	45.6	19.9	8.0	5.7
Inner Mongolia	40.7	11.4	9.2	13.4
Jiangsu	37.8	16.7	11.1	1.6
Jiangxi	40.2	17.5	8.4	3.5
Jilin	38.0	15.8	10.2	5.2
Liaoning	35.8	15.1	10.0	3.1
Ningxia	37.2	14.0	6.4	7.9

Table continues on next page

***Table 3 continued*. Population-Weighted Percentage of Ambient $PM_{2.5}$ Attributable to Coal and Coal Sector Emissions by Province (2013).**

Location	Total Coal	Industrial	Power Plant	Domestic
Qinghai	30.4	11.5	7.1	6.3
Shaanxi	41.4	15.7	7.8	6.8
Shandong	40.6	17.2	12.4	2.5
Shanghai	33.4	14.3	9.0	1.6
Shanxi	40.6	13.9	8.7	7.7
Sichuan	47.8	26.4	6.6	4.6
Tianjin	36.8	15.5	9.9	3.9
Tibet	2.0	0.8	0.9	0.2
Xinjiang	25.5	5.7	13.9	4.2
Yunnan	28.9	13.3	4.7	6.5
Zhejiang	32.2	13.5	9.5	2.3

Nationally, transportation, biomass combustion, and non-coal industrial emissions were the next largest contributors to population-weighted ambient $PM_{2.5}$ concentrations. Non-coal industrial emissions were responsible for approximately 10% of population-weighted ambient $PM_{2.5}$ in most provinces, with transportation and biomass emissions contributing roughly 15%. Open burning contributions were lower and somewhat more variable. The contributions from solvent use indicate that removal of emissions from this sector results in a small **increase** in ambient $PM_{2.5}$. This is a result of a slight increase in inorganic aerosols due to a greater availability of nitric acid and hydroxyl radical that result from the reduction of VOCs.

Table 4. Population-Weighted Percentage of Ambient $PM_{2.5}$ Attributable to Emissions of Other Sectors by Province (2013).

Location	Noncoal Industrial	Transport	Biomass	Open Burning	Solvent Use
China	10.3	15.1	14.8	7.6	−0.1
Hong Kong	7.4	9.4	10.6	6.1	0.0
Anhui	10.1	16.6	15.4	7.6	−0.1
Beijing	10.9	17.6	12.1	7.2	0.0
Chongqing	10.1	12.9	17.1	7.0	−0.1
Fujian	10.2	11.8	14.2	8.0	−0.1
Gansu	10.9	16.0	18.2	8.6	−0.1
Guangdong	10.1	13.3	15.8	8.2	−0.1
Guangxi	9.5	12.8	17.0	8.2	−0.1
Guizhou	12.4	13.5	18.5	9.2	0.0
Hainan	6.9	8.3	12.0	5.8	−0.1
Hebei	11.9	17.8	13.9	8.4	−0.1
Heilongjiang	12.0	17.0	21.7	10.8	−0.1
Henan	9.0	16.9	12.2	6.5	−0.1
Hubei	10.0	15.0	14.7	7.3	−0.1
Hunan	9.8	14.8	14.3	8.0	−0.1
Inner Mongolia	13.5	14.7	16.6	11.3	0.0
Jiangsu	11.0	16.2	14.6	8.2	−0.1
Jiangxi	10.7	14.3	14.8	8.0	−0.1
Jilin	11.6	15.9	18.3	8.8	−0.1
Liaoning	10.9	14.6	13.5	6.8	−0.1
Ningxia	11.4	14.1	17.9	9.4	0.0

Table continues on next page

***Table 4 continued*. Population-Weighted Percentage of Ambient $PM_{2.5}$ Attributable to Emissions of Other Sectors by Province (2013).**

Location	Noncoal Industrial	Transport	Biomass	Open Burning	Solvent Use
Qinghai	10.0	15.4	12.0	6.6	0.0
Shaanxi	11.6	15.2	19.2	9.1	−0.1
Shandong	8.6	17.0	12.1	6.2	−0.2
Shanghai	11.8	13.4	12.4	8.4	−0.1
Shanxi	14.0	15.9	17.2	10.7	−0.1
Sichuan	10.0	13.3	17.0	6.0	−0.1
Tianjin	10.4	18.8	11.5	6.5	−0.2
Tibet	0.8	2.1	1.6	0.8	0.0
Xinjiang	6.0	7.1	11.2	6.4	0.0
Yunnan	11.0	10.8	15.1	7.3	0.0
Zhejiang	9.9	12.1	12.7	7.6	−0.1

SEASONAL VARIATION OF SIMULATED $PM_{2.5}$ CONCENTRATION IN CHINA

The approach to burden of disease estimation is based upon annual average concentrations and therefore does not consider seasonal variation in the level of ambient $PM_{2.5}$. Given the known seasonal variation in the level of ambient $PM_{2.5}$, as well as the seasonal variation in the relative contribution of different sources sectors, our sectoral burden estimates are limited by their annual nature. Depending upon seasonal variation in sectoral contributions may result in over- or under-estimation of sectoral contributions to burden. Here we describe the seasonal patterns of simulated ambient $PM_{2.5}$ and its sectoral contributions. Figure 15 compares simulated seasonal mean percentage contributions from coal-burning in winter (averaged from December to February) and in summer (averaged from June to August). Coal-burning contributed on average 35.4% in winter (excluding the Tibetan Plateau), peaking in individual areas in the Sichuan Basin, Inner Mongolia, and northeastern China with contributions of more than 50%. In summer, the absolute contributions from coal-burning decreased to approximately half of the winter contributions. However, the national average percentage contribution from coal-burning was increased to 48% in summer (excluding the Tibetan Plateau), with contributions as high as 59.5% in the Sichuan Basin. This is likely due to larger seasonal reductions in the contributions of other (noncoal) sources such as traffic, and the high temperature conversion of particulate nitrate to gaseous NO_x. In addition, seasonal changes in sectoral contributions in inland areas are more significant than those in coastal areas.

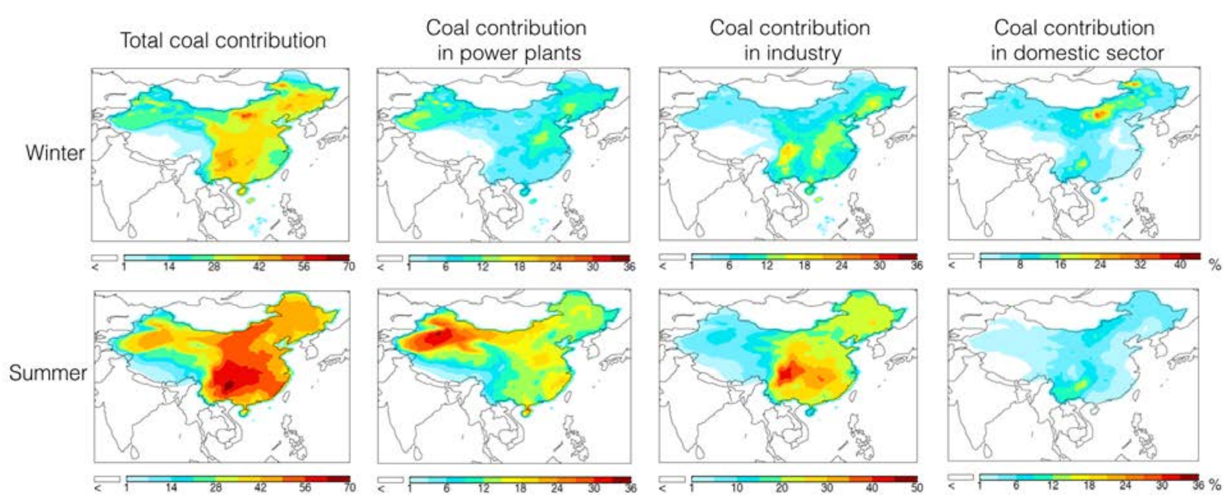


Figure 15. Percentage coal-burning contributions in summer and winter. Note that the scale varies by subsector and season to highlight differences by season for each subsector.

Figure 15 also shows the seasonal sectoral contributions from coal-burning in power plants, industrial, and domestic sectors. The absolute contributions from coal-burning in power plants and industry were slightly reduced in summer. Emissions from these two sectors are relatively constant throughout the year. However, increased summer temperatures lead to reduced nitrate levels but increases in sulfate formation. Due to reductions in total $PM_{2.5}$ and $PM_{2.5}$ contributed from other sources during summer, the percentage contributions from coal-burning in power plants and industry have largely increased. Domestic coal-burning is a more significant contributor in winter given its use in some areas as a heating fuel. Pollution due to domestic coal-burning mainly occurs in individual areas in Guizhou Province and Inner Mongolia. The largest contribution is as much as 40% in Inner Mongolia. Compared to power plants and industrial coal combustion, the percentage contribution from domestic coal-burning is decreased in the summer, with contributions of less than 5% of the ambient $PM_{2.5}$ concentration in most areas in China.

COMPARISON OF COAL AND OTHER SECTOR CONTRIBUTIONS TO SOURCE APPORTIONMENT AND OTHER RECENT ANALYSES

We compared our simulation results with other studies, including source-apportionment studies conducted by the MEP and China Coal Cap Project conducted by the Natural Resources Defense Council (NRDC). MEP released the source-apportionment results of several cities in China for the year of 2014, including Beijing, Tianjin, and Shijiazhuang, as shown in Figure 16.

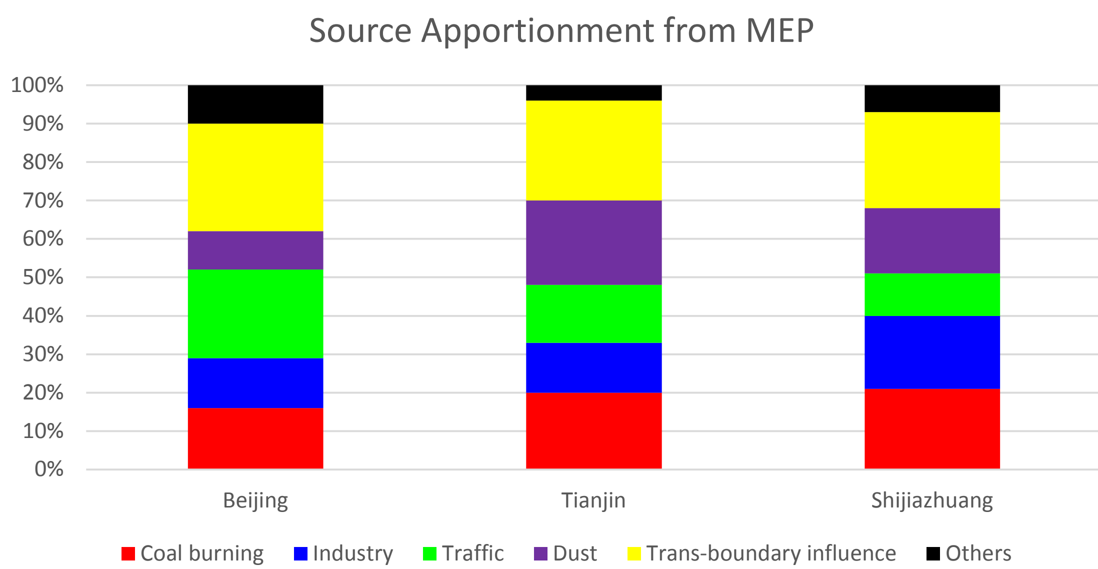


Figure 16. Source apportionment for cities in northern China from MEP.

The Ministry of Environmental Protection of China launched the source apportionment studies in 9 cities in 2013. China National Environmental Monitoring Center developed the Technical Guidance on Source Apportionment of Particulate Matters, which includes three methods, emission inventory, receptor models and air quality models (*http://www.mep.gov.cn/gkml/hbb/bwj/201308/W020130820340683623095.pdf*). MEP released the source apportionment results of several cities in China for the year of 2014, including Beijing, Tianjin and Shijiazhuang, as shown in Figure 16. All these three cities relied the Chemical Mass Balance (CMB) model for the source apportionment and used the air quality model to estimate the contributions of regional emissions. According to the MEP source apportionment results, coal-burning accounts for 16%, 20%, and 21% respectively in the three cities that were evaluated (Table 5).

Table 5. Comparison Between the Current Study and the MEP Source Apportionment

	MEP	This Analysis
Beijing	16%	20%
Tianjin	20%	23%
Shijiazhaung	21%	25%

In our simulations the value for each city is represented by the value of the grid where the city center is located. In our sensitivity simulations, we eliminated all emissions due to coal-burning within China. Thus the contribution here includes the contribution from both local coal-burning and long-range pollutant transport. However, the MEP estimates only reflect the contribution of coal-burning in the surrounding region. We therefore deducted the contributions from surrounding areas from our results, based upon the regional contribution in the MEP results (Beijing 28%, Tianjin 26%, Shijiazhuang 25% as shown in Figure 14). Despite differences in spatial resolution and the complexity of attribution to transportation in source apportionment, the contribution of transportation to ambient $PM_{2.5}$ in the simulations was similar to that estimated by source apportionment in Tianjin (15% according to MEP source apportionment vs 19% in our simulation) and Beijing (25% vs 18% in our simulation).

The China Coal Consumption Cap Project was launched in October 2013 by NRDC and uses the Weather Research and Forecasting-CMAQ model to simulate ambient $PM_{2.5}$ contributions from coal for 333 cities in China. The simulations we report are consistently lower than those from the NRDC, although these differences are in the expected direction, given differences inherent in the NRDC simulations. First, the emissions from coal-burning in the NRDC analyses include emissions both directly from coal-burning and from industries closely related to coal-burning. Second, the contribution of coal-burning for an entire province is an average of those estimates for cities inside each province, while our estimates are the averages (either area- or population-weighted) of the entire province. These differences likely account for the lower estimates in our simulations (Figure 17).

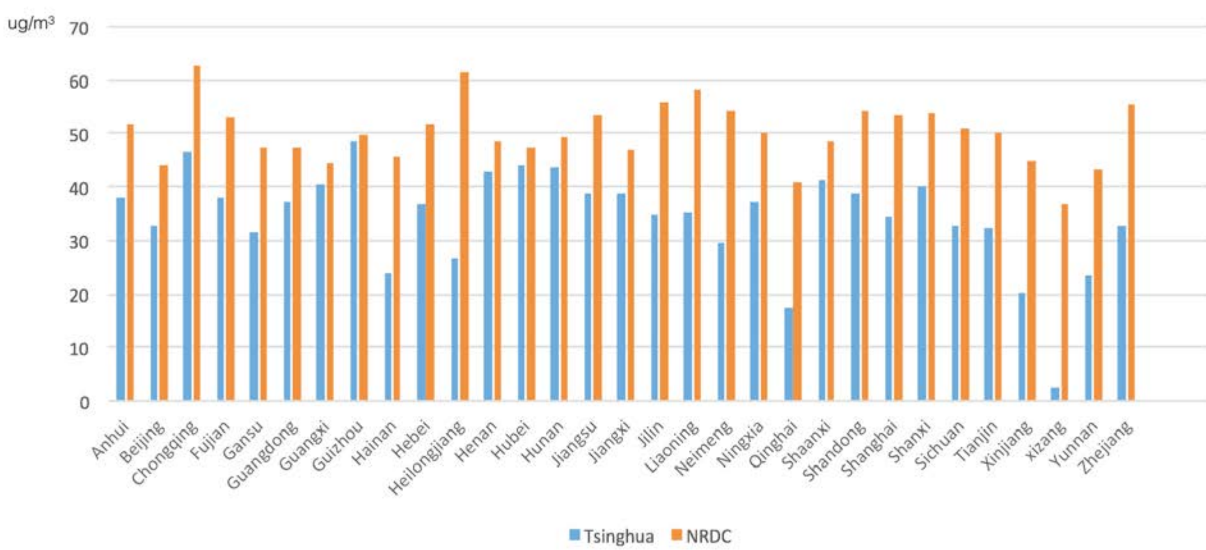


Figure 17. Comparison of GBD MAPS and NRDC estimated contribution of coal-burning to $PM_{2.5}$ levels.

$PM_{2.5}$ Attributable to Coal (and the Coal Sectors)

Because the ambient $PM_{2.5}$ concentrations used in GBD were estimated with higher resolution than estimates provided by GEOS-Chem and were calibrated with ground-based measurements, they were used as the basis for estimating the coal contribution to the exposure and disease burden. For this purpose, the $PM_{2.5}$ estimates used in GBD were multiplied by the population-weighted percentage of $PM_{2.5}$

attributable to coal derived from GEOS-Chem. The resulting population-weighted coal fractions for China as a whole and for each province specifically (the unit of the disease burden estimates) are presented in Figure 18 and in Table 6. High ambient concentrations of $PM_{2.5}$ in Xinjiang province in northwestern China are estimated by the GBD methodology. Mineral dust emissions from the Gobi Desert are important contributors to these elevated concentrations. As these estimated ambient concentrations are partitioned into sectoral contributions derived from the GEOS-Chem simulations, overestimates of absolute levels in Xinjiang may result (Figure 18). However, it is important to note that coal combustion still makes relatively large contributions in this region, for example it is estimated to be responsible for 25% of population-weighted ambient $PM_{2.5}$, with the 13.9% contribution from power plant coal emissions exceeding the national average of 9.5%.

Table 6. Mean (and 95% confidence interval) Population-Weighted Ambient $PM_{2.5}$ and Sector Contributions in China for 2013.

Subsector	Year	Lower	Mean	Upper
All ambient $PM_{2.5}$	2013	51.7	54.3	57.3
Total coal	2013	20.8	21.9	23.1
Power plant coal	2013	4.9	5.2	5.5
Industrial coal	2013	9.0	9.4	10.0
Domestic coal	2013	2.2	2.4	2.5
Noncoal industrial	2013	5.3	5.6	5.9
Traffic	2013	7.8	8.2	8.7
Domestic biomass burning	2013	7.6	8.0	8.5
Open burning	2013	3.9	4.1	4.3
Solvent use	2013	(0.1)	(0.1)	(0.1)

The contributions from solvent use indicate that removal of emissions from this sector results in a small **increase** in ambient $PM_{2.5}$.

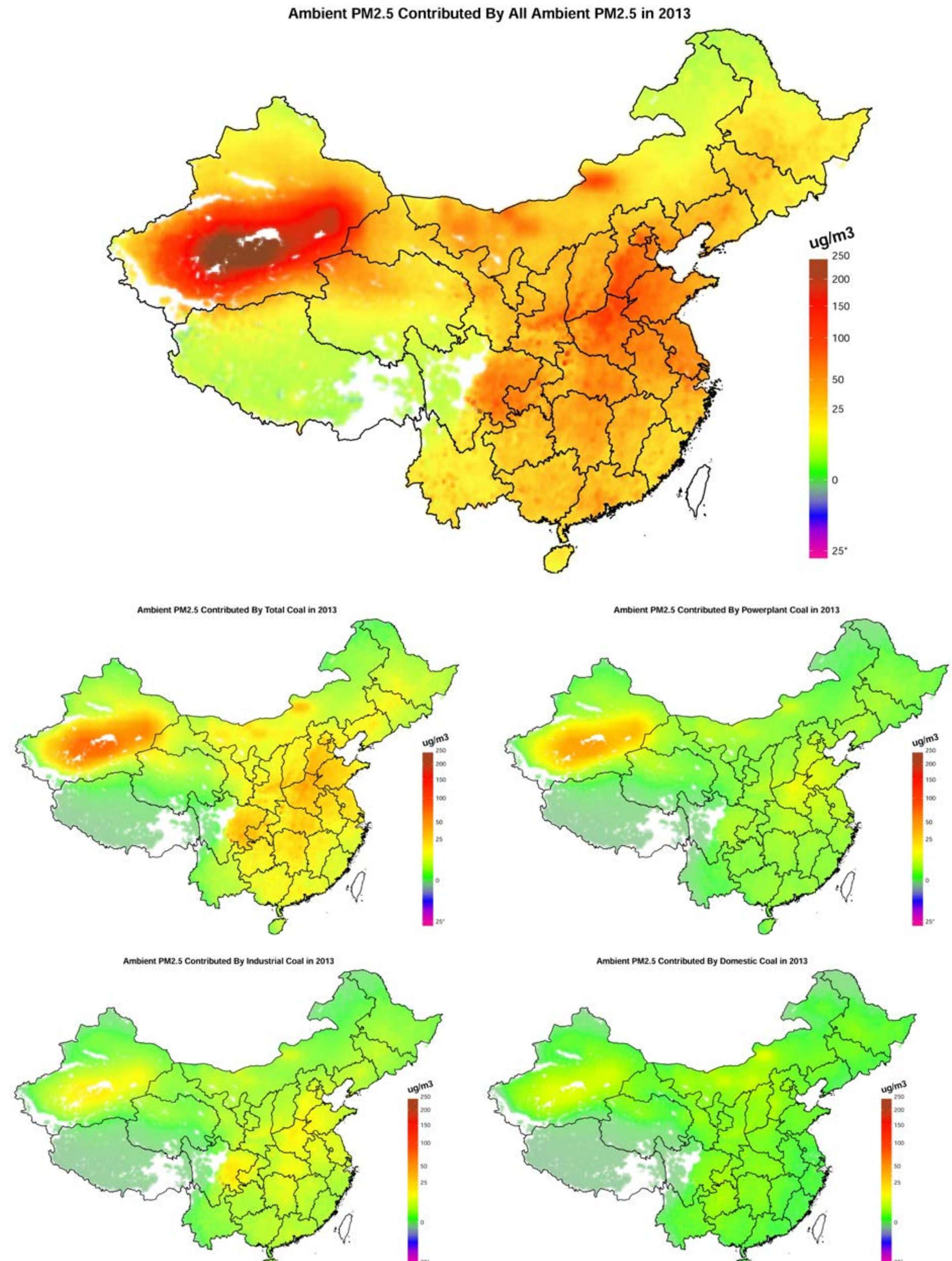


Figure 18. Estimated annual average concentrations of ambient $PM_{2.5}$ from all sources. Panels show the concentrations attributable to coal combustion in total and from power plants, industrial sources, and domestic sources.

$PM_{2.5}$ Attributable to Other Major Sources

Population-weighted $PM_{2.5}$ concentrations and levels of sector contributions for all provinces are provided in Figure 19. The highest total population-weighted $PM_{2.5}$ concentrations were in Beijing, Hebei, Henan, Shandong, and Sichuan. Coal contributions were highest in Beijing, Chongqing, Hebei, Henan, Hubei, Shandong, and Sichuan.

CURRENT (2013) BURDEN OF DISEASE ATTRIBUTABLE TO COAL-BURNING AND OTHER MAJOR SOURCES

The population-weighted fraction of $PM_{2.5}$ attributable to coal and each of the coal sectors (Table 6) was then used together with the IER relationship[7] (Appendices I and II) and the underlying disease burden to estimate the attributable disease burden at the level of each province in China. Estimates are presented for attributable deaths and the age-standardized rate of attributable DALYs (DALYs/100,000) in Tables 7–8 and Figure 20. Tabulations of provincial-level attributable deaths and DALYs for 2013 and future scenarios are available in Appendix V.

Note that total deaths are a function of the hazard as well as the size of the affected population (i.e., within each province) and the underlying prevalence of diseases affected by air pollution. Age-standardized DALY rates are adjusted for the population size and age distribution and are therefore more illustrative in terms of comparing the relative impact of $PM_{2.5}$ and specific source sectors between provinces, or periods of time.

Of the total 915,898 deaths attributable to $PM_{2.5}$ in 2013 in China, 40% (366,161 deaths) are attributable to the sum of all coal combustion. Industrial coal use, power plants, and domestic coal use contributed 17%, 9%, and 4%, respectively. Domestic biomass combustion and transportation sources each contributed 15% of the attributable deaths, followed by noncoal industrial sources (10%) and open burning (8%). Elimination of solvent use was estimated to lead to a small increase in deaths due to the estimated increase in ambient $PM_{2.5}$ associated with removal of solvent emissions, as discussed above.

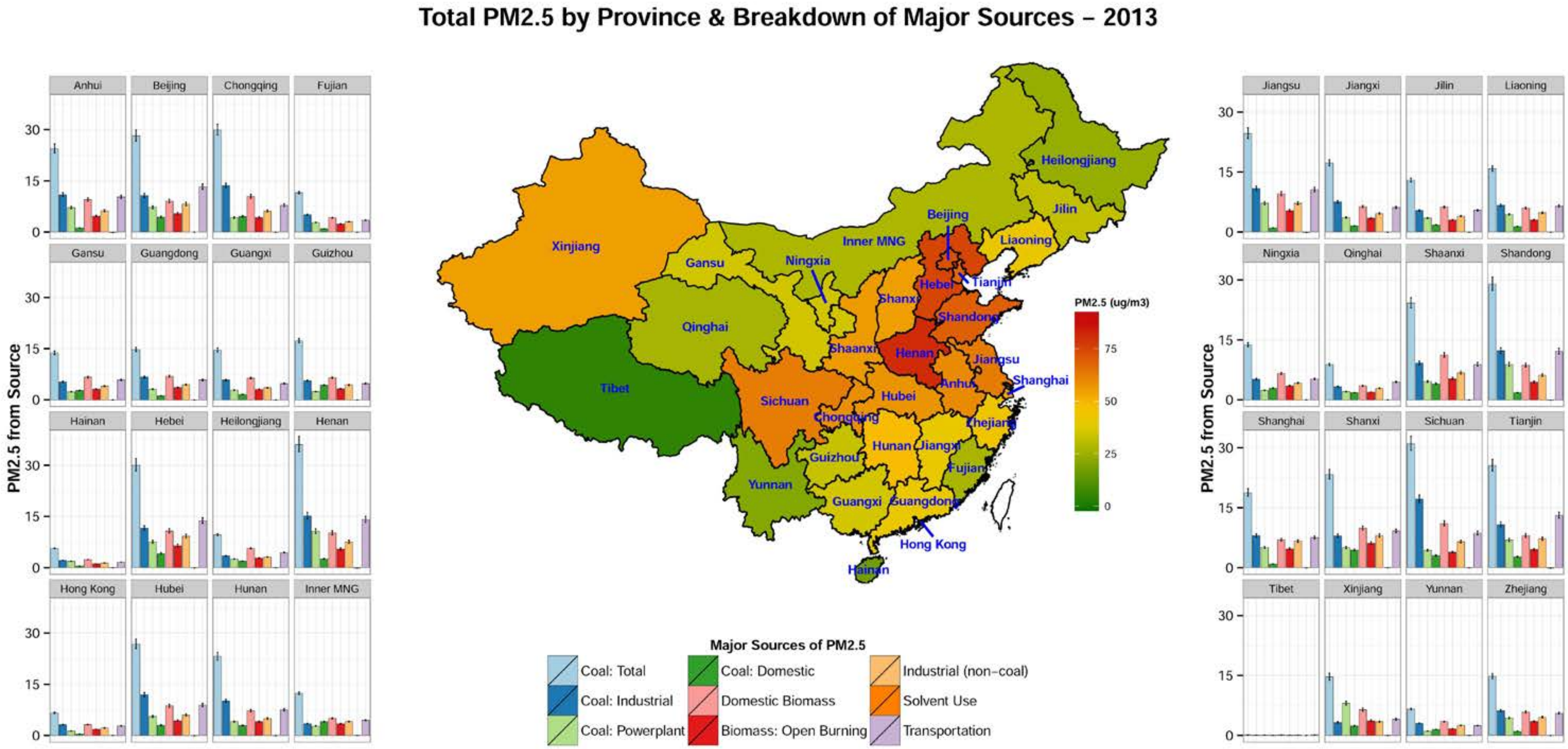


Figure 19. Population-weighted $PM_{2.5}$ concentrations and levels of sector contributions for all provinces.

Table 7. Source Sector Contributions to $PM_{2.5}$-Attributable Deaths (95% UI) in China, 2013

Subsector	Year	Lower	Mean	Upper
All ambient $PM_{2.5}$	2013	821,470	915,898	993,077
Total coal	2013	328,012	366,161	397,031
Power plant coal	2013	77,654	86,531	93,804
Industrial coal	2013	139,318	155,493	168,714
Domestic coal	2013	36,763	41,021	44,623
Noncoal industry	2013	85,137	94,881	102,795
Transportation	2013	123,182	137,395	148,899
Domestic biomass burning	2013	122,397	136,473	147,896
Open burning	2013	63,006	70,228	76,067
Solvent use	2013	−937	−866	−777

Table 8. Source Sector Contributions to $PM_{2.5}$-Attributable Age Standardized DALYs/100,000 (95% UI) in China, 2013

Subsector	Year	Lower	Mean	Upper
All ambient $PM_{2.5}$	2013	1118	1264	1383
Total coal	2013	447	505	553
Power plant coal	2013	105	119	130
Industrial coal	2013	188	213	233
Domestic coal	2013	51	58	64
Noncoal industry	2013	116	131	144
Transportation	2013	167	189	207
Domestic biomass burning	2013	167	189	207
Open burning	2013	86	97	107
Solvent use	2013	−1	−1	−1

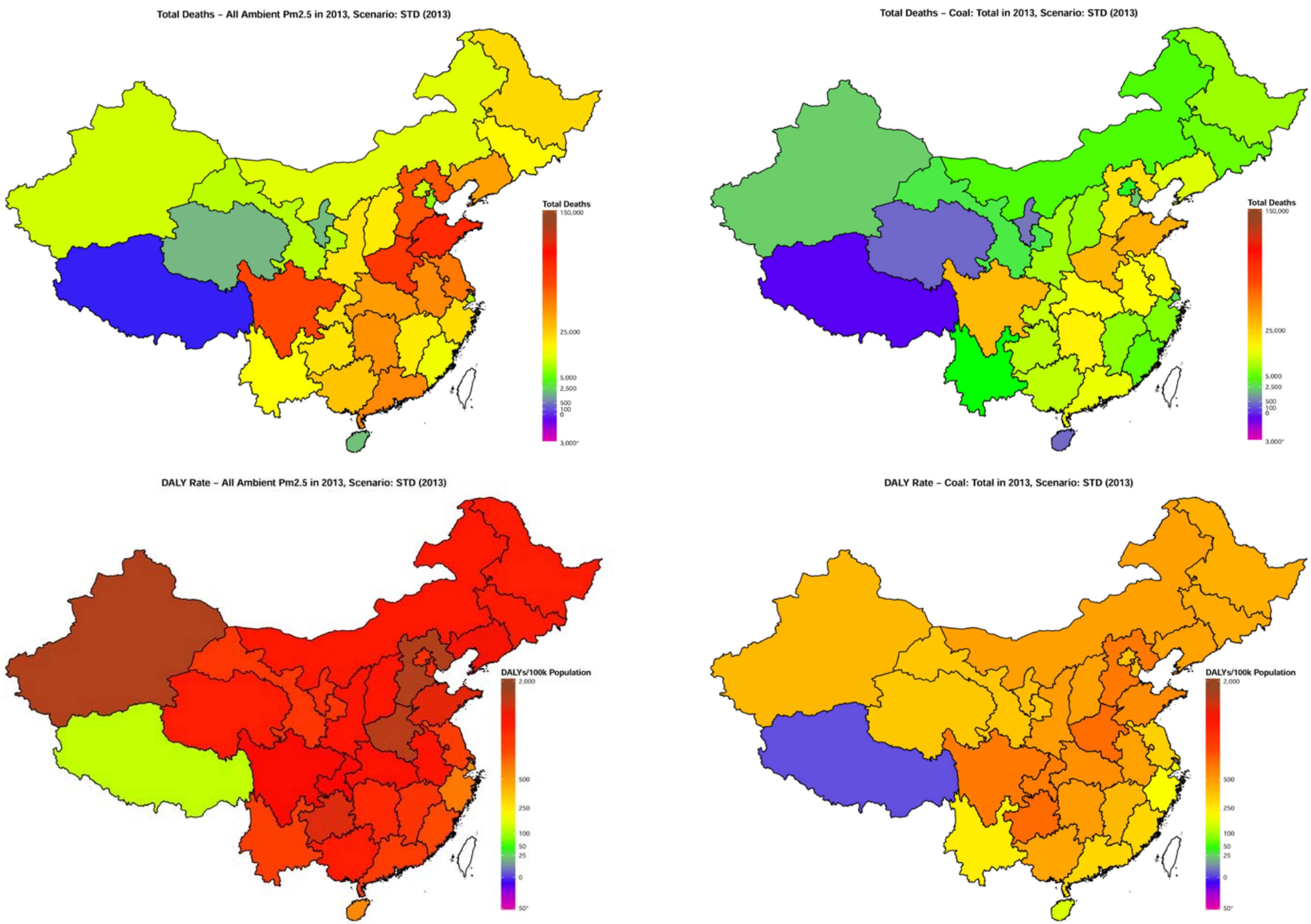


Figure 20. Deaths and age-standardized rates of DALYs (DALYs/100,000) attributable to ambient $PM_{2.5}$ and coal combustion in total by province in 2013.

FUTURE BURDEN OF DISEASE FROM COAL-BURNING

FUTURE SCENARIOS

In this study, we developed emission scenarios to quantify the effects of various measures on future air pollutant emissions based on the energy-saving policies and end-of-pipe control strategies (Table 1). The scenarios are developed with the same model structure as described previously (Wang et al. 2014; Zhao et al. 2013c). We developed two energy scenarios, BAU and PC. The BAU scenario is based on current legislation and implementation status until the end of 2012. In the PC scenario, we assume that new energy-saving policies will be released and enforced more stringently, including lifestyle changes, structural adjustment, and energy-efficiency improvements. For end-of-pipe control, we developed two strategies for each energy scenario, one is designed based on the Twelfth Five-Year Plan for Environmental Protection and the Clean Air Action Plan (abbreviated [1]), and the other is based on maximum feasible control strategies, which assumes that the technically feasible control technologies would be fully applied by 2030, regardless of the economic cost (abbreviated [2]), thereby constituting four emission scenarios (BAU[1], BAU[2], PC[1], and PC[2]). The definition of the energy scenarios and emission scenarios are summarized in Table 1. Detailed description of assumptions and development of the energy and emission scenarios are presented in Appendix IV.

Figure 21 shows the emissions by sector during 2013–2030 in China.

(a) NO_x

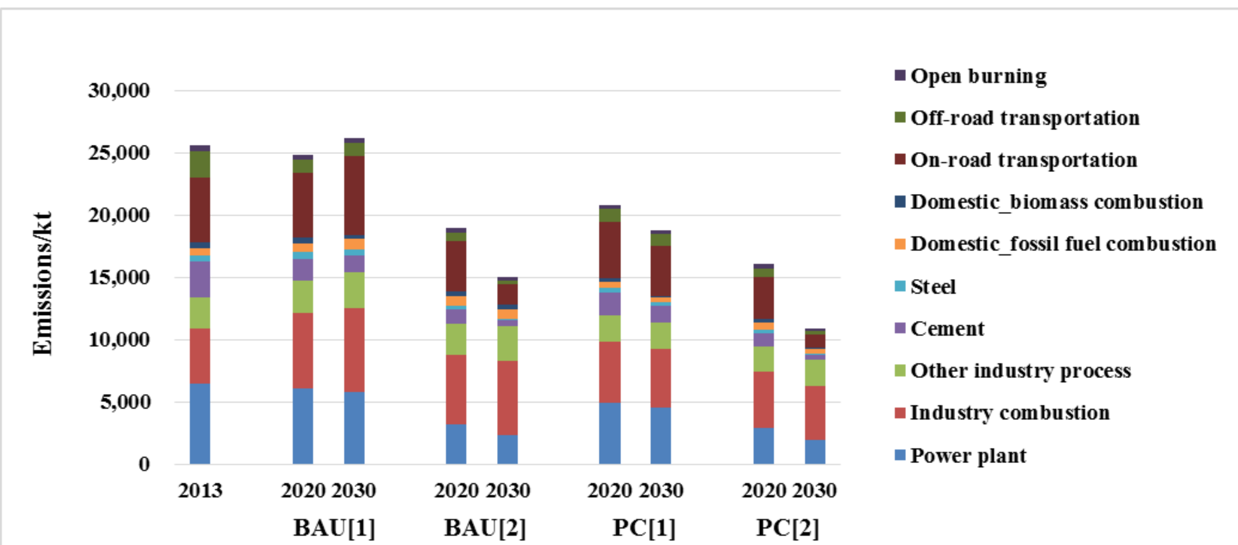
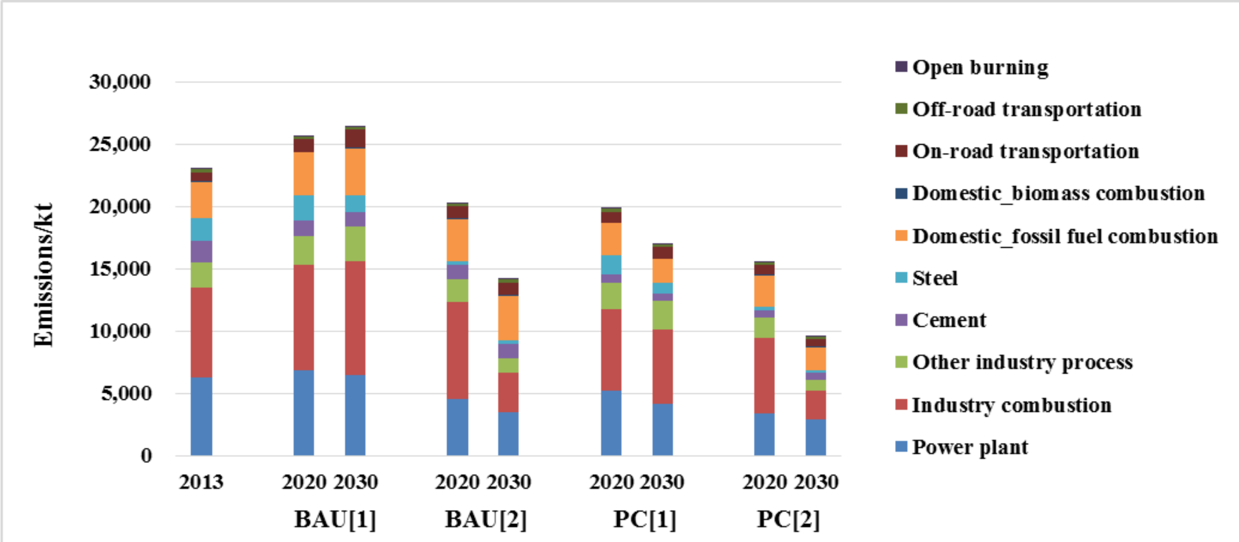


Figure 21. Emissions of major air pollutants in China during 2013–2030: (a) NO_x, (b) SO_2, (c) $PM_{2.5}$, (d) VOCs. *(Continues on next page.)*

(b) SO_2



(c) $PM_{2.5}$

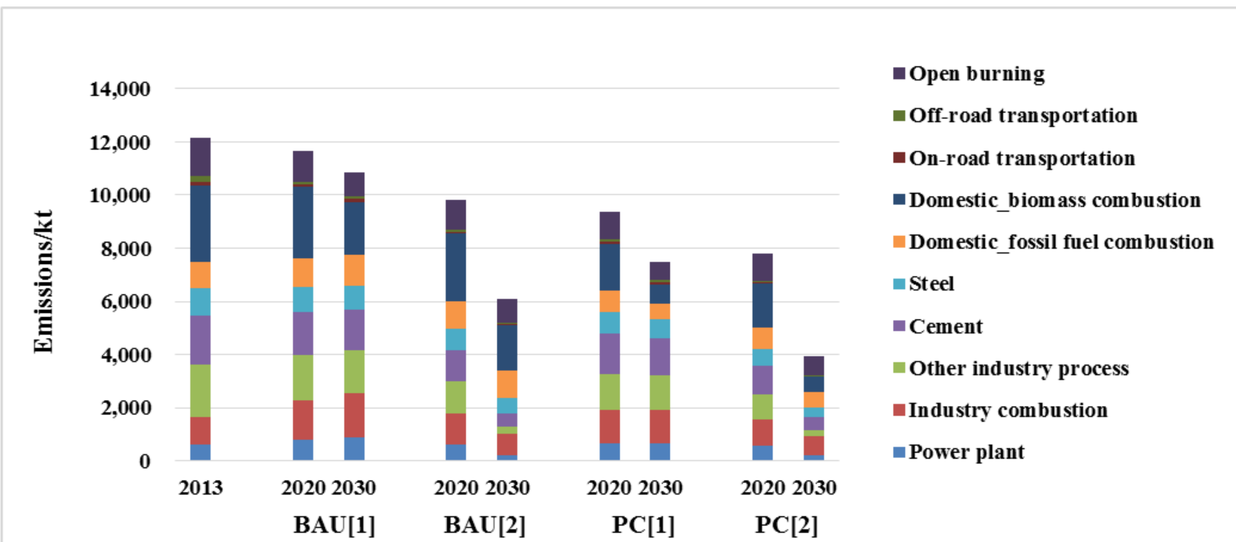


Figure 21. (*Continued*)

(d) VOCs

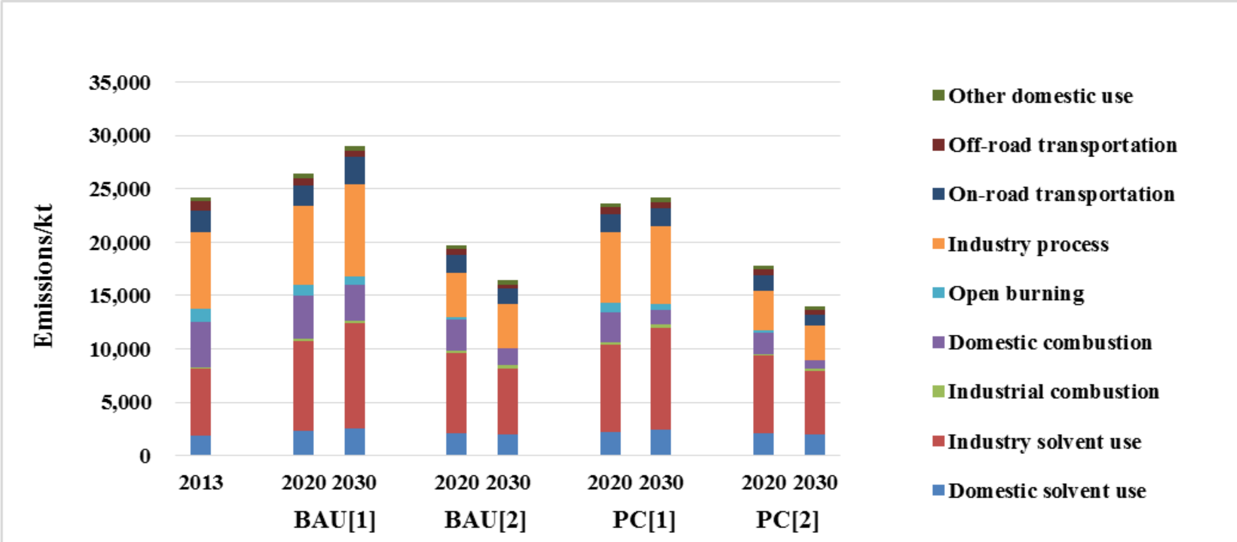


Figure 21. (*Continued*)

Under the BAU[1] scenario, NO_x emissions in China are projected to remain relatively stable from 2013 until 2030. The implementation of assumed energy-saving measures (PC[1]) and maximum feasible reduction end-of-pipe control measures (BAU[2]) are expected to reduce NO_x emissions by 28% and 42%, respectively, from the BAU[1] scenario. With the application of PC[2] scenario, the remaining emissions account for only 43% of the 2013 levels. The most effective control measures are the installation of selective catalytic reduction/selective noncatalytic reduction (SCR/SNCR) and the application of tight vehicle standards, which contribute to 54% and 28% of this reduction, respectively.

The SO_2 emissions in China are predicted to have a 14% growth from 2013 to 2030 with the current legislation and current implementation status. The application of advanced energy-saving measures (PC[1]) could lead to a substantial reduction in SO_2 emissions (36% reduction) from the BAU[1] scenario, exceeding the effect of maximum feasible reduction end-of-pipe control measures (BAU[2]) (28%). As the power sector has largely been equipped with FGD facilities in the base year, industrial boilers and industrial process contribute a lot of the SO_2 emission reduction achieved through installation of desulfurization facilities. In the maximum feasible reduction scenario (PC[2]), SO_2 emissions are estimated at only 36% of the BAU[1] scenario, or 42% of the 2013 levels.

Similar to NO_x, $PM_{2.5}$ emissions in China are projected to remain relatively stable up to 2030 under the current policies, resulting from the balance between growing energy consumption and existing control policies (in particular, vehicle emission standards). New energy-saving policies (PC[1]) and full application of the best available end-of-pipe control measures (BAU[2]) result in about 31% and 43% reduction in $PM_{2.5}$ emissions from the levels of the BAU[1] scenario, respectively. Full application of the best available technologies (PC[2]) could reduce PM_{10} and $PM_{2.5}$ emissions to about one quarter of the levels of the baseline projection or the base year. The most effective end-of-pipe control policies are the application of new emission standards for various industrial sources and domestic biomass burning. Replacement of coal and direct biomass burning with clean fuels are assumed in both urban and rural areas, with faster progress in the PC scenario (Appendix IV). If the best available technologies are fully applied, the $PM_{2.5}$ emissions would be reduced to about one third of the levels of baseline projection or the levels of the base year.

China's NMVOC emissions are estimated to increase by 20% from 2013 to 2030 under the BAU[1] scenario. The emissions from open burning and residential sectors are expected to decline as a result of the dwindling direct combustion of biomass in residential sector. By carrying out a series of energy-saving policies (PC[1]), total emissions are expected to decrease by 17% from the BAU[1] scenario. Another 37% could be reduced if the most strict end-of-pipe control measures are implemented (BAU[2]), and the most effective measures are the substitution with low solvent products and add-on removal technologies such as incineration and adsorption in the industrial and solvent use sectors. With full implementation of the best available technologies (PC[2]), the NMVOC emissions could be reduced to less than half of the levels of the BAU[1] scenario.

COMPARISON WITH OTHER PROJECTIONS

For NO_x, Ohara and colleagues (2007) projected NO_x emissions in China until 2020 by using the emissions for 2000 and three scenarios, a policy failure scenario, a best guess scenario, and an optimistic scenario. These projections of emissions for all three scenarios for 2010 were much lower than our estimates, indicating underestimation of economic growth during the 2000–2010 period. Amann and colleagues (2008) developed three scenarios until 2030 based on emissions in 2005. Their current legislation scenario assumed current legislation and current enforcement, while the advanced control technology scenario assumed across-the-board application of advanced control technologies; principally consistent with existing German legislation. The optimized scenario was a least cost optimization scenario that would achieve the same health benefit as the advanced control technology scenario. In an analysis conducted in coordination with that of the Amman team, Xing and colleagues (2011) projected NO_x emissions for 2020 with four scenarios based on year 2005 emissions, including a scenario assuming current legislation and implementation status, a scenario assuming improvement of energy efficiencies and current environmental legislation, a scenario assuming improvement of energy efficiencies and better implementation of environmental legislation, and a scenario assuming improvement of energy efficiencies and strict environmental legislation. Similar to Ohara and colleagues (2007), these projections for year 2010 emissions were also significantly lower than our estimates. As for the growth rates until 2020 or 2030, all the scenarios in these two studies projected a larger increase in emissions than our progressive control strategy, which assumes enforcement of the Twelfth Five-Year Plan, indicating these two studies did not anticipate stringent future control policies. Cofala and colleagues (2012) projected the NO_x emissions until 2030 with year 2010 emissions and four scenarios envisaging energy-saving measures at different stringency levels. The projected change rates for the period 2010–2030 range between 16% and −24%. Since no end-of-pipe control measures beyond the baseline were considered, it is only meaningful to compare these scenarios with our BAU[1] and PC[1] scenarios. The Xing and Amman research teams both predicted a stronger growth potential of China’s energy consumption in the future, leading to larger growth rate in emissions as above.

For SO_2, Ohara and colleagues (2007) predicted that SO_2 emissions would change by 27%, −11%, and −23% during 2010–2020 in the policy failure, best-guess, and the optimistic scenarios, respectively. Amann and colleagues (2008) failed to reproduce the declining trend during 2005–2010, but the control policies assumed in their most aggressive scenario (the advanced control technology scenario) resulted in a similar decline rate as our progressive control strategy. The growth rates projected in all the four scenarios of Xing and colleagues (2011) were higher than our BAU[1] scenario, indicating that this assumption of future SO_2 control policies were more conservative than our progressive control strategy based on the Twelfth Five-Year Plan. Cofala and colleagues (2012) predicted the SO_2 emissions would decrease by 20%–40% during 2010–2030 with four scenarios assuming different energy-saving policies. The differences are attributed to a stronger growth potential of China’s energy consumption than predicted in our study.

For PM, the PM_{10} emissions growth rate until 2020 of the least aggressive scenario in Xing and colleagues (2011) is comparable to our BAU[1] scenario, and the most aggressive scenario is comparable to our PC[1] scenario, indicating similar stringency levels of the control policies assumed in these two studies up to 2020. Amann and colleagues (2008) predicted an increase of $PM_{2.5}$ emissions during 2010–2030 in their current legislation scenario; the growth rates in the advanced control technology or optimized scenarios are close to our PC[1] scenario. Cofala and colleagues (2012) projected the change rate of $PM_{2.5}$ emissions for 2010–2030 between −20% and −34% with four energy scenarios. Our maximum feasible reduction scenario projects lower emissions than other previously developed scenarios.

For NMVOC, Ohara and colleagues (2007) predicted much higher growth rates for the 2010–2020 period in all three scenarios, as they assumed hardly any effective control measures. Xing and colleagues (2011) and Wei and colleagues (2011) have considered the effect of recent vehicle emission standards on NMVOC emissions, and assumed simple but progressively emerging control polices until 2020, and therefore achieved growth rates similar to ours for both baseline and progressive strategies. Given that China is still in the starting stage of NMVOC emission controls, and new policies could emerge only slowly in the next 5–10 years, emission trends would not be expected to deviate greatly from the baseline until 2020. However, control measures at different stringency levels might result in dramatically different emissions by 2030. Our study quantified the effect of potential new policies on NMVOC emission trends until 2030 and the maximum feasible reduction potential.

PROJECTIONS OF FUTURE AMBIENT $PM_{2.5}$ CONCENTRATIONS

Figure 22 shows the simulated ambient $PM_{2.5}$ concentration in each future scenario and contributions from total coal-burning and specific sources.

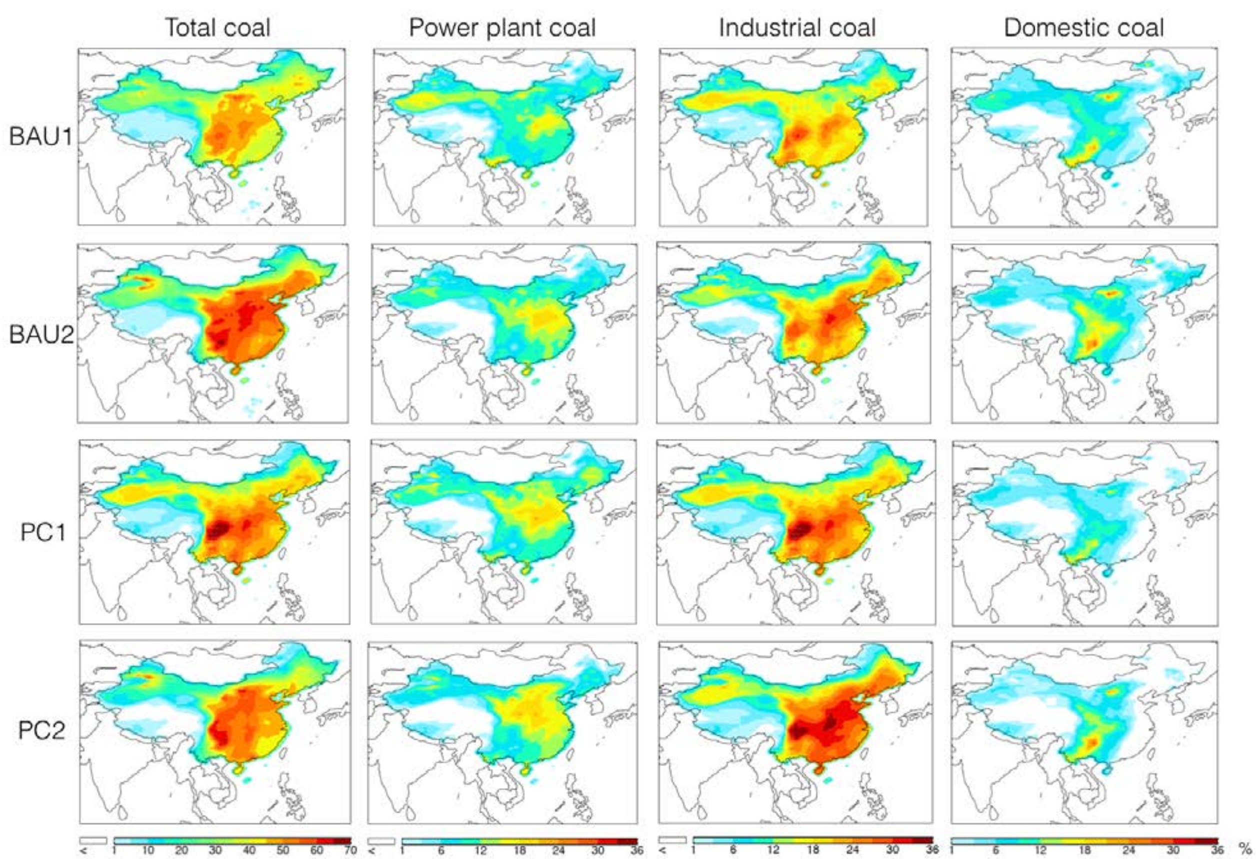


Figure 22. Simulated percentage contributions to $PM_{2.5}$ from coal combustion in future scenarios.

In BAU1, the $PM_{2.5}$ concentration remains at a high level, especially in the Sichuan Basin, the North China Plain and in central China. The highest value occurs in the Sichuan Basin. Concentrations in most areas in central China and in the North China Plain are also high. As the end-of-pipe emission control policies become more stringent, the ambient $PM_{2.5}$ concentration falls in BAU[2]. The highest concentration is in Sichuan Basin under this scenario. The PC[1] has a comparable $PM_{2.5}$ concentration as that of BAU2, as the energy policies are improved but the end-of-pipe control strategy does not change from the current trend. In PC[2], the ambient $PM_{2.5}$ concentration shows a significant decrease, with highest levels in individual areas in the Sichuan Basin and in the northern part of central and eastern China.

Coal-burning contributions to $PM_{2.5}$ concentration have similar spatial distributions in all the future scenarios, while the value decreases from BAU[1] to PC[2] as more emission control policies are put into effect. In BAU[1], coal-burning has the largest contribution in the Sichuan Basin, especially in the city of Chengdu. In scenarios BAU[2] and PC[1], the highest contributions also occur in central China and in the Sichuan Basin. In PC[2], the contribution from coal-burning decreases throughout the country.

As in the baseline conditions, contributions from the different coal subsectors have disparate spatial patterns and magnitude. Coal-burning in power plants has a larger influence in northern China, in all future scenarios. In other areas besides the northern part of central and eastern China, coal-burning in the power sector generally has a relative small contribution.

Industrial coal-burning, as the largest contributor to ambient $PM_{2.5}$, has different spatial distribution of concentration in future scenarios. In BAU[1], the largest contribution occurs in the Sichuan Basin. In northern China, contributions are generally low. In BAU[2], the $PM_{2.5}$ concentration due to industrial coal-burning falls in the Sichuan Basin, which leads to a different spatial pattern. The contributions in the Sichuan Basin and northern China are comparable under this scenario. The spatial patterns in the PC scenarios are similar to those in BAU scenarios. The largest contribution occurs in the Sichuan Basin in PC[1], while it is somewhat lower in both the Sichuan Basin and in northern China in PC[2]. Contributions of domestic coal have similar distributions in scenarios BAU[1] and BAU[2], with the highest levels in Guizhou Province. In the PC scenarios, as the new energy policies are carried out, the contributions due to domestic coal-burning decrease, with no change in spatial patterns.

Because ammonia emissions reductions are not part of any of the four future scenarios, but are an important emissions source from the agricultural sector and important contributors to secondary particle formation, a sensitivity analysis was conducted to evaluate the impact of a 20% reduction in ammonia emissions for the base year. Agricultural (from fertilizer and livestock) emissions accounts for ~90% of the total ammonia emissions in China. Results of this simulation indicate that agriculture emissions have a significant impact on ambient $PM_{2.5}$, contributing up to 10% (and up to ~8 $\mu g/m^3$) in some regions. The highest contribution occurs in the North China Plain. Estimation of the overall impact of agricultural ammonia emissions in 2013 indicate an (area) average contribution of 6.8 $\mu g/m^3$ in China with contributions as high as 36 $\mu g/m^3$ in the North China Plain (Figure 23). We did not include ammonia control in any of the future scenarios, given the current absence of specified policies to address ammonia emissions in China in the future. While this decision not to incorporate ammonia reductions in the future scenarios will affect the predictions of total ambient $PM_{2.5}$, we do note that ammonia is not limiting for sulfate formation, for which coals plays a larger role. In future analyses of projected sectoral contributions to ambient $PM_{2.5}$ and disease burden, incorporation of specific ammonia control scenarios should be considered given their important contribution to ambient $PM_{2.5}$, a contribution that would be expected to increase in importance with reductions in emissions from other sectors.

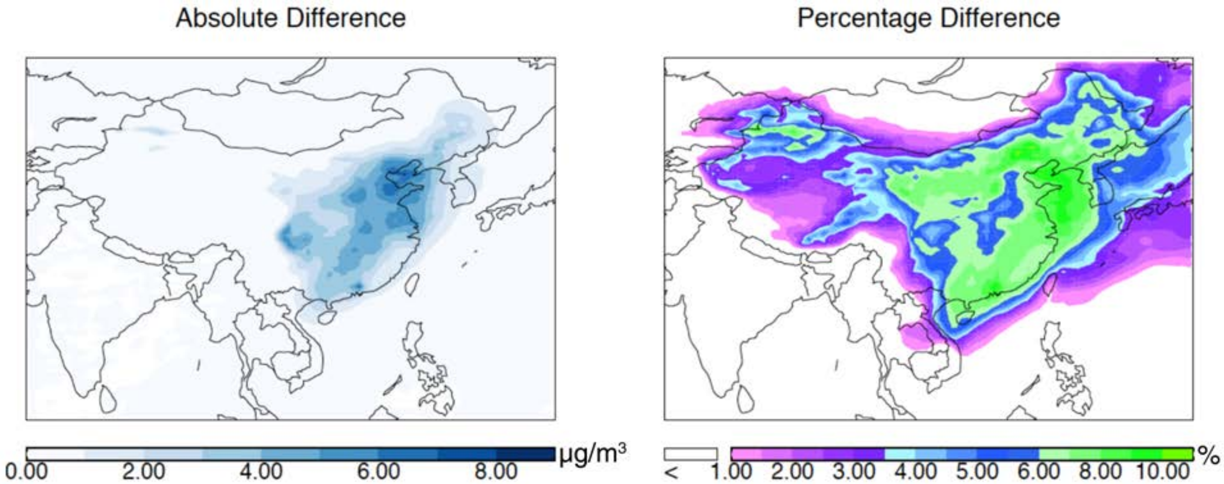


Figure 23. Absolute and percentage differences in $PM_{2.5}$ levels due to agricultural emissions.

DISEASE BURDEN ESTIMATION UNDER FUTURE SCENARIOS

To estimate the future (year 2030) burden for each of the four future scenarios it is necessary to estimate both future population-weighted concentrations (which are used to calculate the PAF) and future mortality.

$PM_{2.5}$ Projections

To calculate the population attributable fraction ($PAF_{PM2.5}$) in 2013 we used the gridded surface of (total) ambient $PM_{2.5}$ concentrations that was developed for GBD 2013 ($PM2.5_{GBD2013}$) together with the sector contributions estimated from GEOS-Chem, as described above. Specifically, the coal contributions to ambient $PM_{2.5}$ are calculated by multiplying the gridded values of $PM2.5_{GBD2013}$ by the gridded fractions of ambient $PM_{2.5}$ attributable to coal (f_{coal}) as estimated from GEOS-Chem simulations:

$$PM2.5_{coal} = f_{coal} * PM2.5_{GBD2013} .$$

To calculate $PAF_{PM2.5}$ for other years (i.e., 2030) requires estimating both f_{coal} and total ambient $PM_{2.5}$. f_{coal} is calculated from additional year 2030 GEOS-Chem simulations described above, which necessarily also simulate total ambient $PM_{2.5}$ ($PM2.5_{GChem2030}$), the *standard simulation*. In order to account for changes in the levels of total ambient $PM_{2.5}$ in between 2013 and 2030 we scaled $PM2.5_{GBD2013}$ by the change in total ambient $PM_{2.5}$ simulated by GEOS-Chem between 2013 and 2030:

$$PM2.5_{GBD2030} = \frac{\mathrm{PM2.5_{GChem2030}}}{\mathrm{PM2.5_{GChem2013}}} * PM2.5_{GBD2013} .$$

Note that for each of the different future scenarios (BAU[1], BAU[2], PC[1], and PC[2]) a different standard simulation was used to estimate $PM2.5_{GChem2030}$.

The resulting population-weighted $PM_{2.5}$ concentrations and levels of sector contributions for all provinces in 2030 are provided in Figures 24–27 for the four future scenarios. Spatial patterns and overall levels follow the same trends as described above based on the GEOS-Chem simulations.

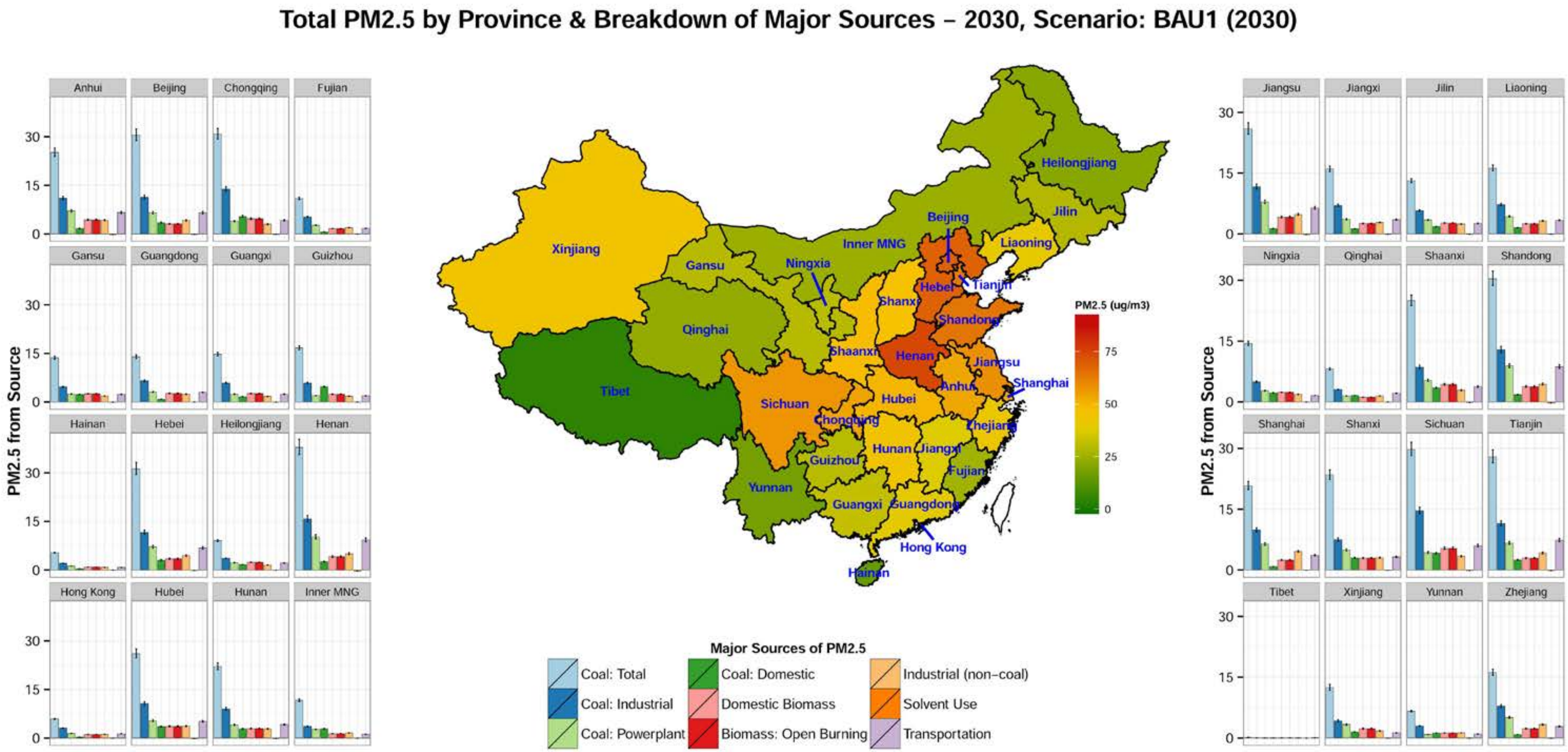


Figure 24. $PM_{2.5}$ levels by province and major air pollution sources in 2030 under scenario BAU1.

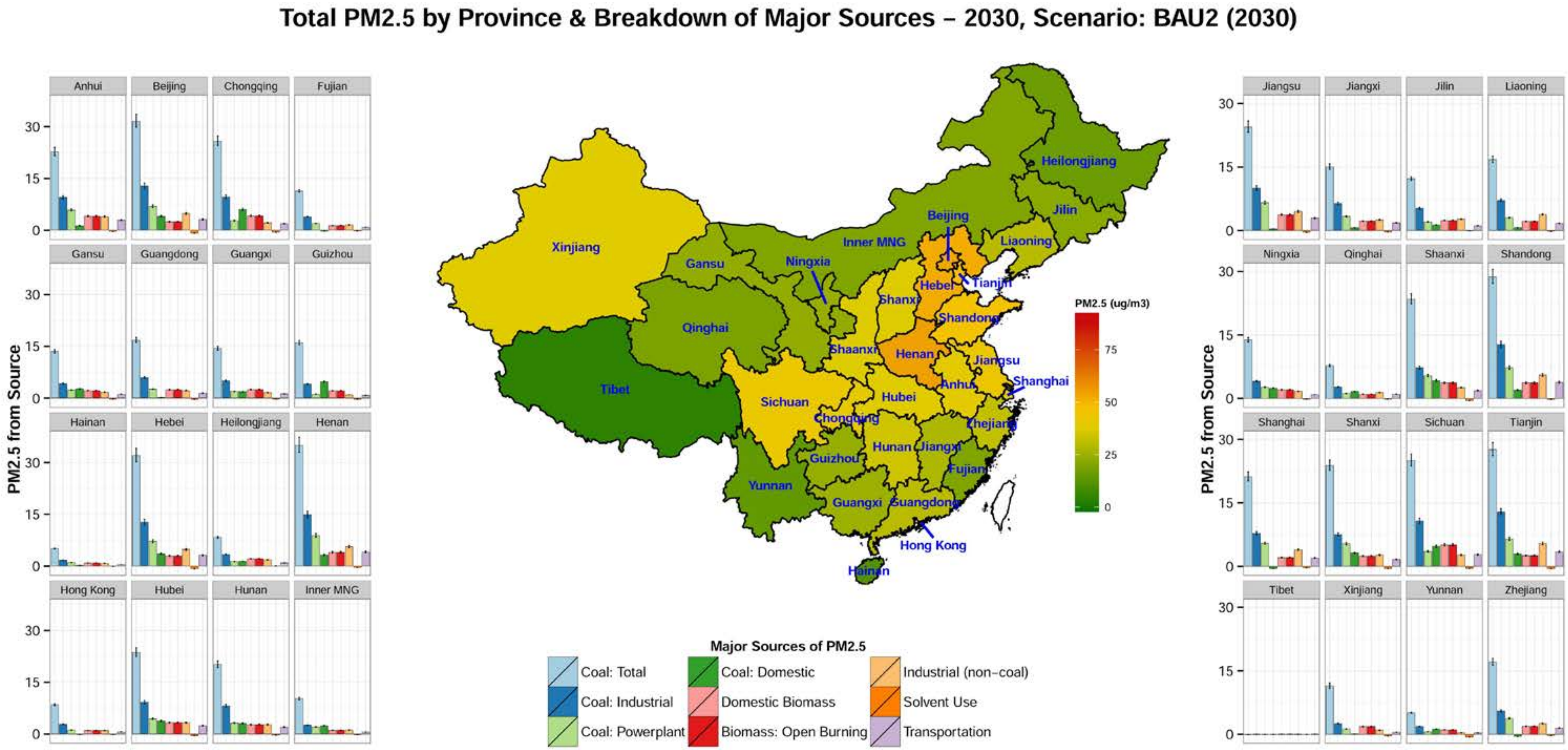


Figure 25. PM$_{2.5}$ levels by province and major air pollution sources in 2030 under scenario BAU2.

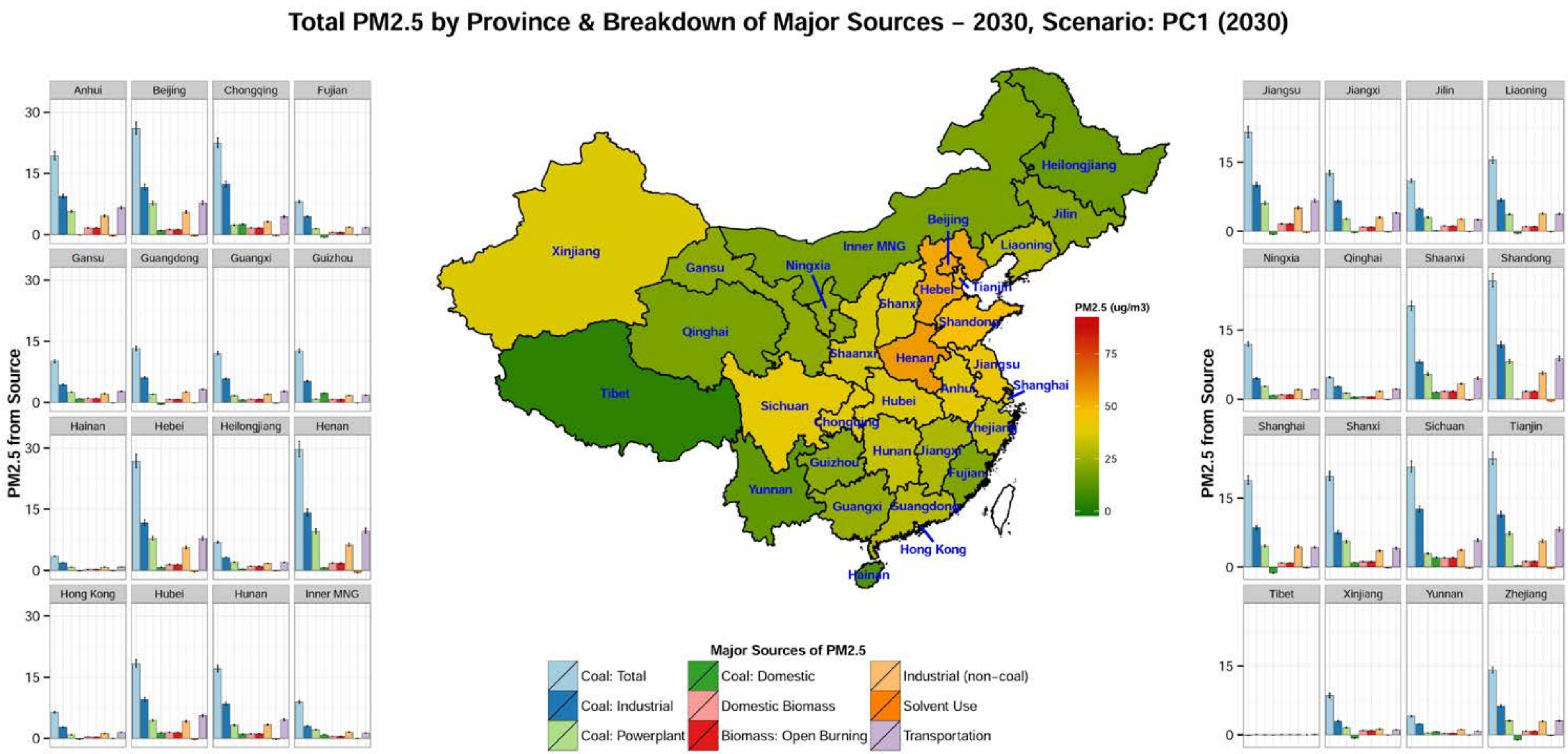


Figure 26. PM$_{2.5}$ levels by province and major air pollution sources in 2030 under scenario PC1.

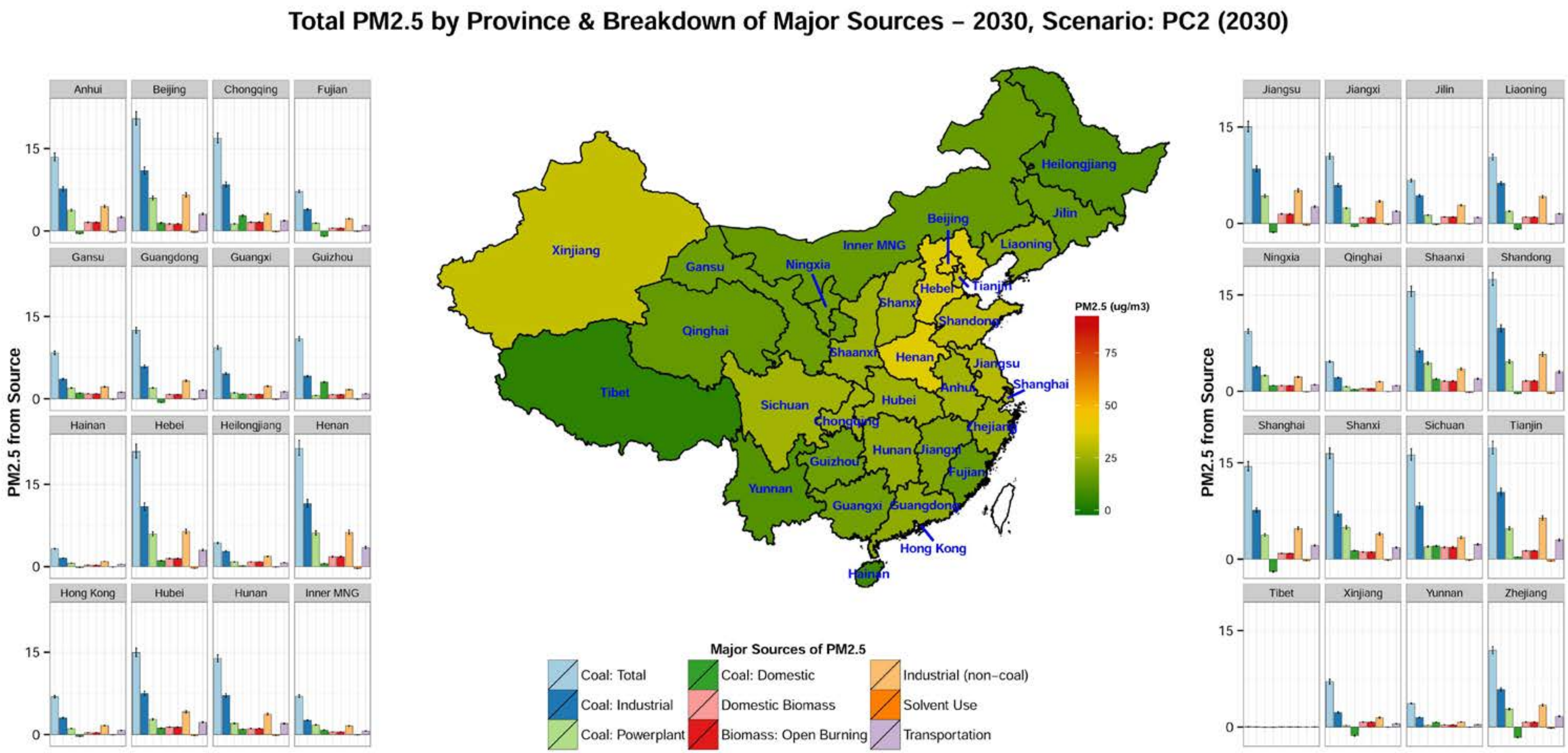


Figure 26. $PM_{2.5}$ levels by province and major air pollution sources in 2030 under scenario PC2.

Mortality Projections

To calculate the future mortality and burden of disease (DALYs) for the four future (year 2030) scenarios, we use mortality and DALY estimates, the annual rate of change from 1990 to 2013, and the attributable fraction for 1990 and 2013. As we are imposing specific scenarios related to air pollution and as air pollution is a known risk for mortality, we first isolate the expected mortality not due to air pollution for 2030. The mortality not attributable to air pollution was calculated as:

$$Mort_{U(year)} = Mort_{(year)} * \left(1 - PAF_{PM2.5(year)}\right),$$

where

$$Mort = Mortality\ Rate$$
$$Mort_U = Unattributable\ Mortality\ Rate$$
$$PAF_{PM2.5} = Population\ Attributable\ Fraction\ due\ to\ Ambient\ Air\ Pollution$$
$$year = Future\ year\ to\ estimate.$$

Next, we calculated the annualized rate of change in mortality (AROC) between 1990 and 2013 as:

$$AROC = \frac{\ln\left(\frac{Mort_{U(2013)}}{Mort_{U(1990)}}\right)}{(2013-1990)}.$$

Using the above we estimate the mortality not attributable to air pollution for future years

$$Mort_{U(year)} = Mort_{(2013)} * \exp\left(AROC * (year - 2013)\right).$$

We then calculate the PAF due to air pollution in future years based on the current level and the future scenarios as described above. Using the PAFs we than calculate the total mortality in the future as:

$$Mort_{(year)} = \frac{Mort_{U(year)}}{\left(1-PAF_{PM2.5(year)}\right)}.$$

The avoided burden due to reductions in emissions (the various scenarios) is calculated from the difference between the two scenarios: that with and without reducing $PM_{2.5}$ emissions.

Burden of Disease Attributable to Coal-Burning in the Future

Figure 28 and Table 9 present the total deaths attributable to ambient $PM_{2.5}$ overall and from coal-burning and other major sources for China as whole. Figure 29 shows the total deaths by province in 2013 and under the four future scenarios. Compared to 2013, each of the scenarios is predicted to lead to increases in total attributable future deaths: 38% increase for BAU1, 25% BAU2, 24% PC1, and 8% PC2. In comparison to the deaths attributable to $PM_{2.5}$ from coal, each of the scenarios predicts even larger increases in 2030: 52%, 71%, 45%, and 34% for BAU1, BAU2, PC1, and PC2, respectively.

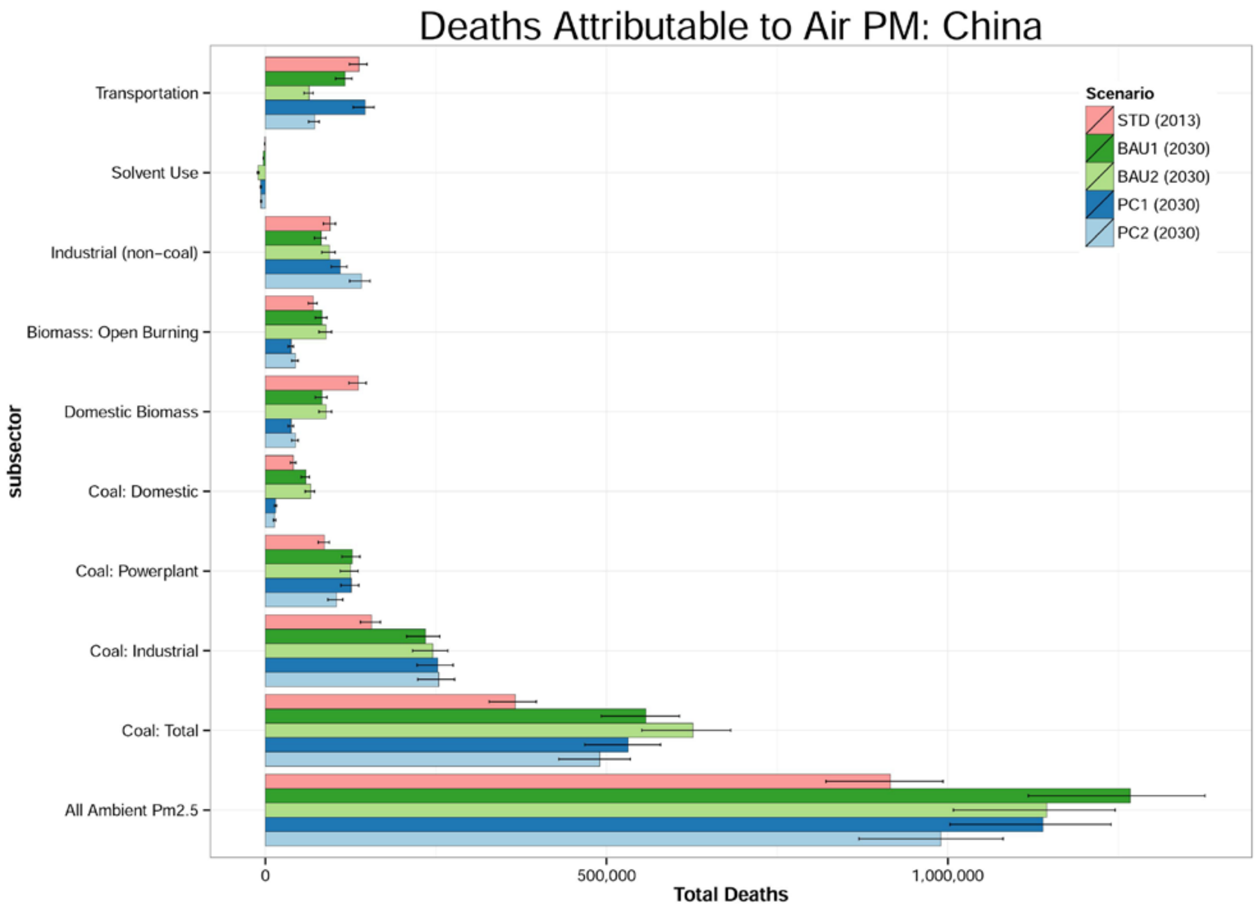


Figure 28. Deaths attributable to ambient $PM_{2.5}$ and from coal-burning and other major sources for China as whole for 2013 and for the four future (year 2030) scenarios.

Table 9. Total Deaths Attributable to Ambient $PM_{2.5}$ Overall and From Coal-Burning

Subsector	Year	Scenario	Lower	Mean	Upper
All ambient $PM_{2.5}$	2013	STD	821,470	915,898	993,077
All ambient $PM_{2.5}$	2030	BAU1	1,117,966	1,267,386	1,376,759
All ambient $PM_{2.5}$	2030	BAU2	1,008,043	1,144,814	1,245,158
All ambient $PM_{2.5}$	2030	PC1	1,003,207	1,139,228	1,239,026
All ambient $PM_{2.5}$	2030	PC2	869,752	990,108	1,080,938
Total coal	2013	STD	328,012	366,161	397,031
Total coal	2030	BAU1	492,152	557,761	606,848
Total coal	2030	BAU2	551,558	626,604	681,914
Total coal	2030	PC1	467,915	531,775	579,069
Total coal	2030	PC2	430,154	490,019	534,730

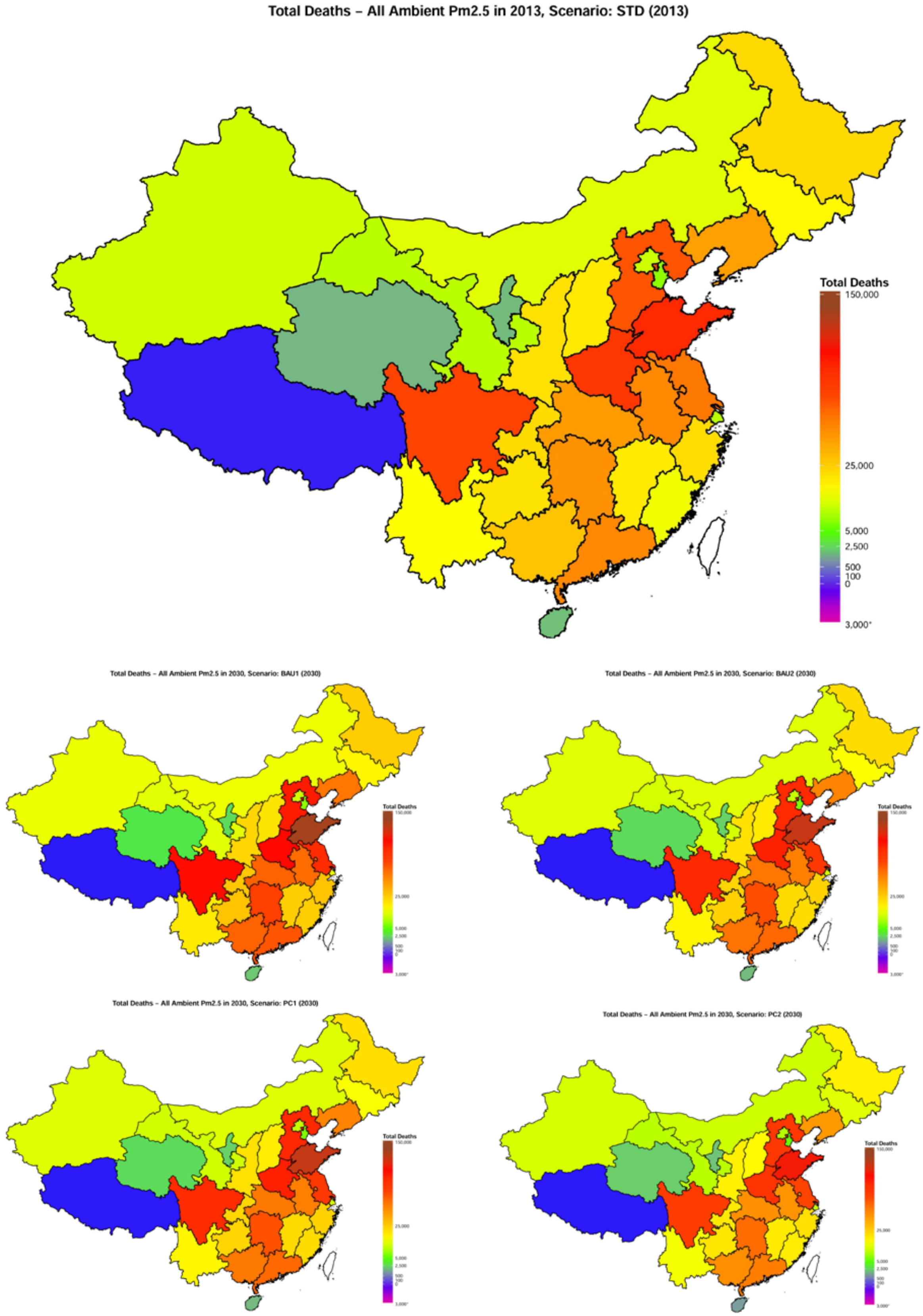


Figure 29. Deaths attributable to ambient $PM_{2.5}$ for all provinces for 2013 and for the four future (year 2030) scenarios.

Figure 30 shows the age-standardized DALY rates attributable to ambient $PM_{2.5}$ overall and from coal-burning and other major sources for China as whole in 2013 and under the four future scenarios. Figure 31 shows these results at the provincial level. Unlike the results for total deaths, the age-standardized DALY rates control for predicted changes in population size and age structure between 2013 and 2030. In contrast to the results for total deaths, reductions in coal-burning in the future result in net gains in healthy life-years reductions in lost years of healthy life per capita relative to 2013 under all four future scenarios. The PC2 scenario results in the largest reduction in DALYs per capita for total coal-burning and powerplant coal. The PC1 scenario results in the largest reduction in DALYs per capita for industrial coal.

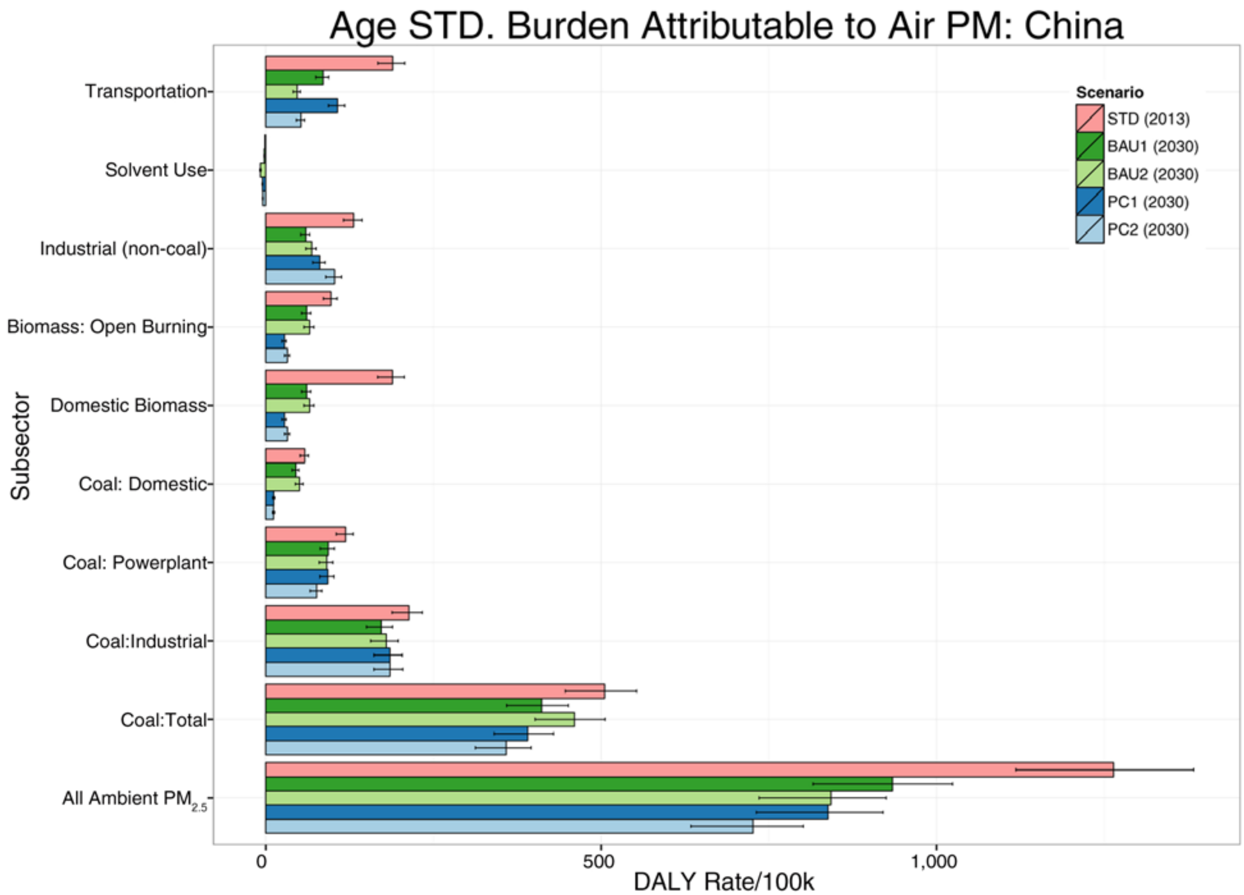


Figure 30. Age-standardized DALY rates attributable to ambient $PM_{2.5}$ and from coal-burning and other major sources for China as whole for 2013 and for the four future (year 2030) scenarios.

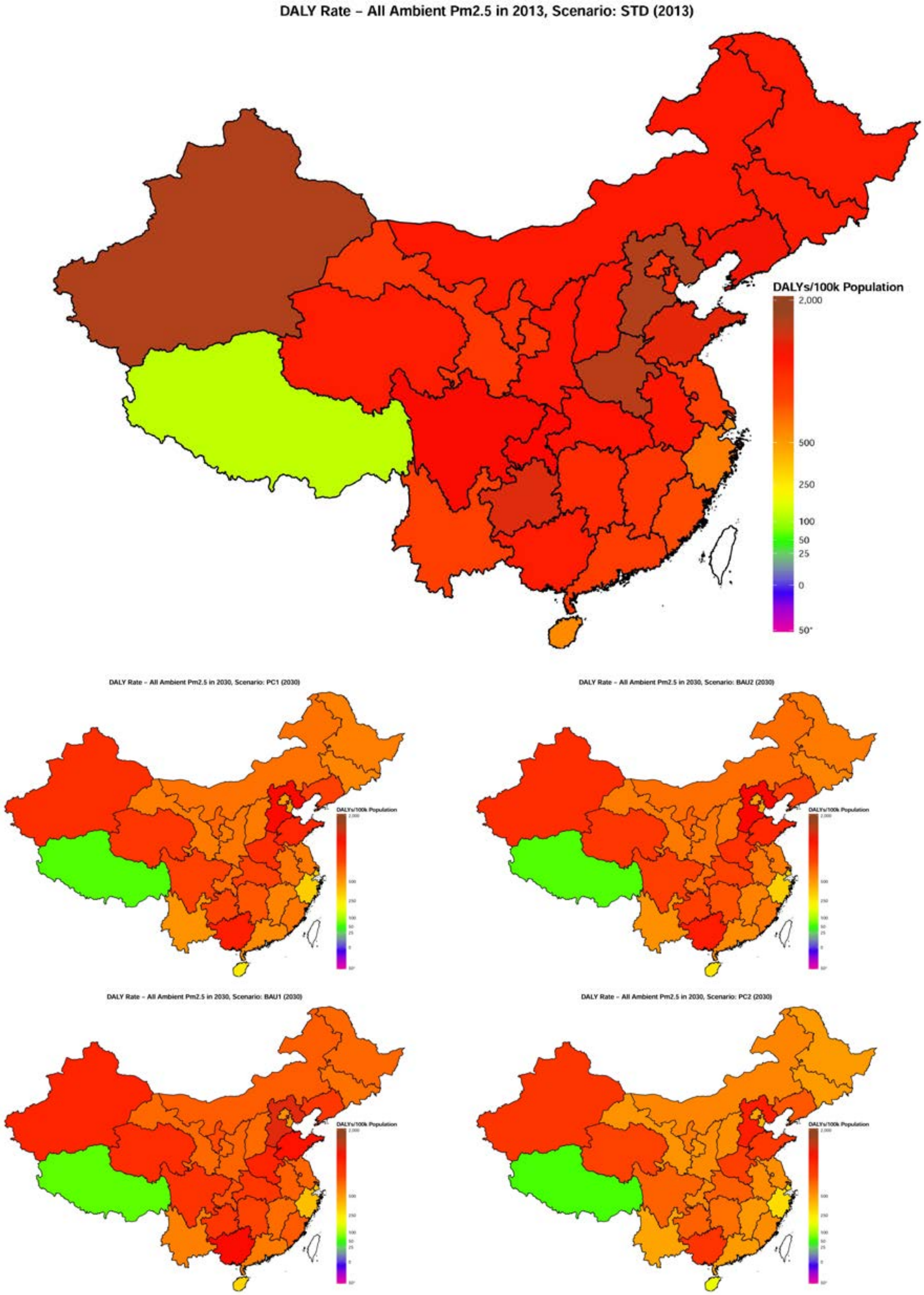


Figure 31. Age-standardized DALY rate (DALYs/100,000 population) attributable to ambient $PM_{2.5}$ for all provinces for 2013 and for the four future (year 2030) scenarios.

Provincial-Level Results

GBD 2013 estimated mortality and burden of disease for China at the provincial level for the first time ever. This enabled the contribution of ambient air pollution and other risk factors to be estimated at finer spatial scales than ever before. The text box, Provincial-Level Results, presents provincial-level estimates of 2013 and projected 2030 mortality attributable to coal-burning and other major sources.

Text Box: Provincial-Level Results

One of the unique aspects of this analysis is the estimation of disease burden and sectoral contributions at the provincial level for China. The table presents the ranking of ambient $PM_{2.5}$ and coal combustion as risk factors for mortality in 2013, relative to the 79 risk factors considered in the GBD. For China as a whole ambient $PM_{2.5}$ was the 5th leading morality risk factor with coal's contribution to ambient $PM_{2.5}$ ranking 12th. At the provincial level coal was the 9th leading risk factor for mortality in Chongquing and Sichuan; it ranked 11th in Anhui, Henan, Jiangsu, Jiangxi, Shaanxi, Shandong, and Tianjin. In Gansu, Hainan, Heilongjiang, Hong Kong, Jilin, Qinghai, Tibet, Xinjiang, and Yunnan coal had a substantially lower mortality risk than at the national level with rankings ranging from 15 to 49.

Table. Ranking of Ambient $PM_{2.5}$ and Total Coal Combustion as Risk Factors for Mortality in 2013 for China and All Provinces. Bold text indicates those provinces with rankings higher than the national ranking.

Location	Ambient $PM_{2.5}$ Rank	Coal Rank
China	5	12
Anhui	**4**	**11**
Beijing	**3**	12
Chongqing	5	**9**
Fujian	5	15
Gansu	6	16
Guangdong	**4**	14
Guangxi	5	14
Guizhou	7	12
Hainan	8	17
Hebei	5	13
Heilongjiang	7	18
Henan	5	**11**
Hong Kong	5	18
Hubei	5	14
Hunan	6	13
Inner Mongolia	6	15
Jiangsu	**4**	**11**
Jiangxi	6	**11**
Jilin	5	17
Liaoning	**3**	13
Ningxia	5	15
Qinghai	7	18
Shaanxi	5	**11**
Shandong	**4**	**11**
Shanghai	4	13
Shanxi	5	12

Table continues on next page

Table continued.

Location	Ambient $PM_{2.5}$ Rank	Coal Rank
Sichuan	5	**9**
Tianjin	**3**	**11**
Tibet	23	49
Xinjiang	7	18
Yunnan	8	18
Zhejiang	4	13

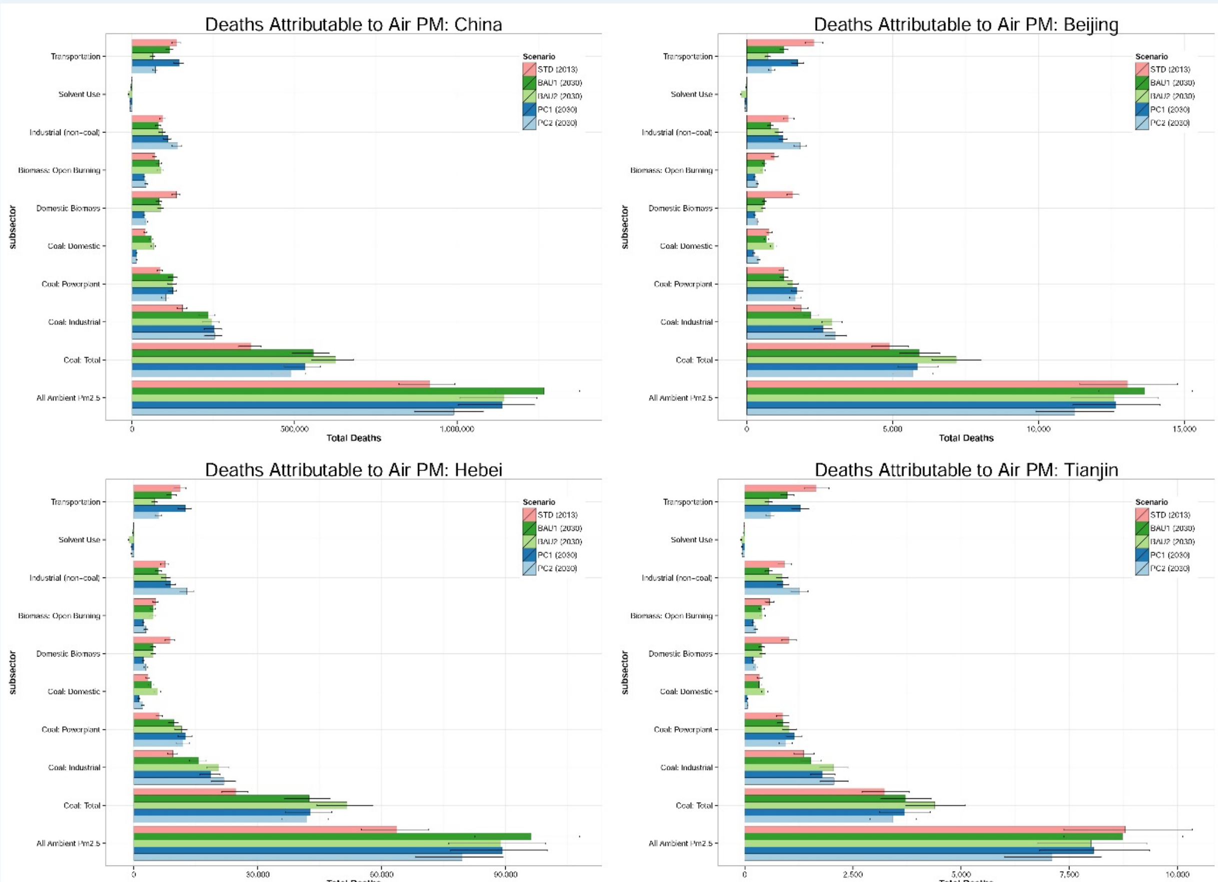


Figure. Total deaths attributable to ambient $PM_{2.5}$ overall and from coal-burning and other major sources for China, Beijing, Hebei, and Tianjin in 2013 and under the four future (2030) scenarios.

As an example of variation between provinces in and in comparison with the national-level estimates, of the deaths attributable to $PM_{2.5}$ in 2013 within the Jing-Jin-Ji (i.e., Beijing, Hebei, Tianjin), coal was responsible for 4900 (of 13,000), 24,600 (of 63,700), and 3200 (of 8800) in Beijing, Hebei, and Tianjin, respectively. Decreases in estimated $PM_{2.5}$-attributable mortality relative to that in 2013 were estimated for Tianjin for all four future scenarios and for the BAU2, PC1, and PC2 scenarios in Beijing, while patterns in Hebei generally followed the national projections of increases for all scenarios. These decreases projected in Beijing and Tianjin are not, however, due to reductions in the contribution from total coal combustion (or from power plant or industrial coal), but are due to decreases in the impact of other source sectors such as transportation, domestic (coal and biomass), and open biomass burning in most of the future scenarios. Although the contribution from coal combustion increases in all provinces under all scenarios, the increase is much larger in Hebei, which has the effect of offsetting decreased impacts from the other sectors.

SUMMARY AND CONCLUSIONS

This report presents estimates of the current and future burden of disease attributable to coal-burning in China. The disease burden attributable to the contribution of coal-burning to ambient $PM_{2.5}$ in 2013 for China as a whole and for all provinces was estimated in a core analysis and in separate analyses for major sectoral contributions — specifically transportation, noncoal industrial, and other sources (domestic biomass, open biomass burning, solvent use). Different energy scenarios and air pollution control scenarios were then developed to reflect different future emission pathways, and these were used for analysis of mortality and disease burden projections in the year 2030. We estimated both coal and other major source contributions to ambient $PM_{2.5}$ and their associated disease burdens under each of the future scenarios for the year 2030 — considering both future mortality projections and future emissions scenarios. Simulation of the impact of different air pollution source sectors on ambient pollution concentrations with a high-resolution chemical transport model incorporates complex and nonlinear relationships between emissions and ambient concentrations and allows for spatial disaggregation at a resolution appropriate for health impact analysis. Importantly, this analysis is unique in its inclusion of provincial-level estimates of underlying disease prevalence. Nonlinear relationships between exposure and resulting disease burden are also incorporated into the estimates.

Ambient $PM_{2.5}$ is a major contributor to mortality and disease burden in China, estimated to be responsible for 916,000 deaths in 2013, the 5th leading risk factor for mortality. Coal-burning was the most important single sectoral contributor to ambient $PM_{2.5}$, responsible for 40% of population-weighted $PM_{2.5}$ in China. In specific provinces (Chongqing, Guizhou, Sichuan) the contribution was nearly 50%. Compared with limited available source-apportionment analyses for major Chinese cities, our estimates of the contribution of coal combustion to ambient $PM_{2.5}$ were remarkably similar, although we estimated slightly higher contributions (3%–4%).

Given the large impact of coal combustion on ambient $PM_{2.5}$ concentrations, coal combustion was an important contributor to disease burden in China, with an estimate of 366,000 attributable deaths in 2013. $PM_{2.5}$ from coal combustion specifically was the 12th leading risk factor for mortality in China in 2013 and accounted for more deaths than high cholesterol, drug use, or second-hand smoking. Domestic biomass and coal combustion were together the next largest contributor to ambient $PM_{2.5}$ attributable mortality in 2013— with a combined impact (177,000 deaths) larger than that of industrial coal (155,000 deaths), transportation (137,000 deaths), or coal combustion in power plants (86,500 deaths). The attributable mortality (136,000 deaths) from domestic biomass combustion's contribution to outdoor air pollution was at the same level as that from industrial coal and transportation. Note that these contributions were from the impact of domestic combustion on levels of $PM_{2.5}$ in outdoor air. Although it is unlikely that impacts on disease burden from exposure to ambient $PM_{2.5}$ and household combustion of solid fuels are completely independent, GBD 2013 estimated that household air pollution from the combustion of solid fuels in China was estimated to account for an additional 807,000 deaths in 2013[19]. Given the large contributions of domestic combustion to disease burden via household air pollution exposure and its important contribution to the burden attributable to ambient $PM_{2.5}$, reductions in domestic biomass and coal emissions would be expected to lead to large reductions in disease burden and should be prioritized for future energy and air quality management strategies.

[19] Assuming complete independence, the combined impact of household and ambient air pollution (including ozone) in China was 1.6 million deaths in 2013, the third leading risk factor for mortality.

This report provides the first comprehensive assessment of the current and predicted burdens of disease attributable to coal-burning and other major sources in China at the national and provincial levels, but recent peer-reviewed studies provide more limited assessments using different data and methods. Lelieveld and colleagues (2015) conducted an analysis of source sector contributions to air pollution and the resulting year 2010 and 2050 disease burden, and they reported higher numbers of premature deaths in 2010 due to $PM_{2.5}$ exposure than did GBD 2013 (1.36 million vs. 0.857 million[20]). The contribution of coal-burning was not addressed per se, but they attributed 237 thousand deaths, or 18% of total $PM_{2.5}$-attributable deaths, to $PM_{2.5}$ from power plants vs. 86,500 deaths from $PM_{2.5}$ from coal-burning power plants, or 9% of total $PM_{2.5}$-attributable deaths in the current study. Lelieveld and colleagues used an earlier version of the IER (Burnett et al. 2014) and a coarser $PM_{2.5}$ spatial resolution (~100 × 100 km) than the current study; both differences could contribute to the differences in estimates. Chen and colleagues (2013) estimated the effect on Chinese life expectancy at birth of a policy that from 1950 to 1980 provided free coal for winter heating to cities north of the Huai River. Using a quasi-experimental model Chen and colleagues estimated that the policy resulted in TSP levels that were 55% higher north of the Huai River and life-expectancy decrements of 5.5 years in the population of Northern China. Differences in data and study design between their study and the current study preclude formal comparisons, but it is reasonable to assume that their estimates of effects on life expectancy are considerably greater than what would have been estimated using the data and methods of the current study. Limited and incomplete mortality data, including inattention to trends in child mortality, combined with spatial mismatching of the sites for which mortality and TSP data were obtained, may have affected their estimates.

Compared to 2013, all of the future scenarios are predicted to lead to increases in future deaths attributable to ambient $PM_{2.5}$ with increases of 38%, 25%, 24%, and 8% for BAU1, BAU2, PC1, and PC2, respectively, compared to attributable mortality in 2013 (916,000) (Figure 32). Specifically, ambient $PM_{2.5}$ was projected to be responsible for 1.3, 1.1, 1.1, and 0.99 million deaths in 2030, for the BAU1, BAU2, PC1, and PC2 scenarios, respectively. While the population-weighted mean exposure to $PM_{2.5}$ is projected to decrease under all scenarios (from 54 $\mu g/m^3$ in 2013 to 50, 38, 38, and 27 $\mu g/m^3$ in 2030 for BAU1, BAU2, PC1, and PC2, respectively) with the WHO Interim Target-1 reached under the PC2 scenario, the projected increase in mortality is due to the aging population and increased prevalence of IHD, stroke, COPD, and LC, leading to increases in the number of deaths attributable to exposure to ambient $PM_{2.5}$. These projections illustrate the importance of population dynamics in determining temporal trends in mortality attributable to ambient $PM_{2.5}$. While GBD 2013 estimated increases in exposure, the numbers of deaths and the death rate attributable to $PM_{2.5}$ between 1990 and 2013, future projections indicate increased attributable mortality, even with declining exposures (Figure 32). China implemented a new population policy in 2016 to move from one child per family to two children per family, which is expected to have an important impact on future demographics. However, given that our projections only consider 14 years forward (from the implementation of this policy in 2016), and that the impact of demographics on disease burden attributable to air pollution primarily results from the proportion of the population at older ages (> 65 years), our results are not likely to be sensitive to this new policy.

[20] Source of estimate: GBD Compare *http://vizhub.healthdata.org/gbd-compare/*. Accessed 01/07/16.

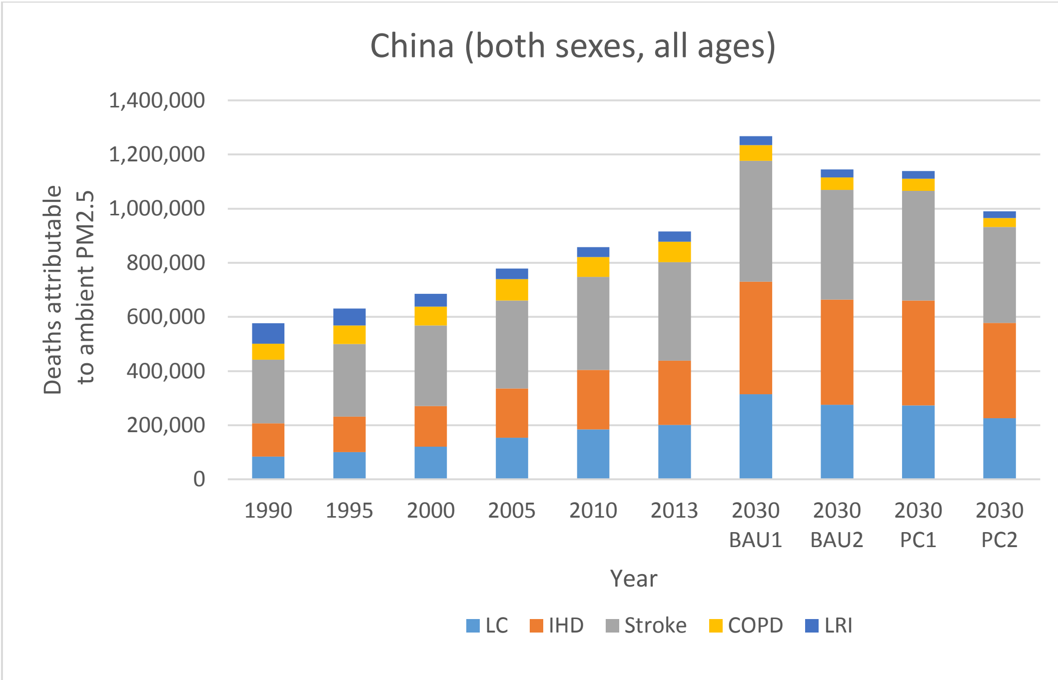


Figure 32. Deaths in China attributable to $PM_{2.5}$ 1990–2030 by year and cause. Deaths in 2030 are predicted under four scenarios of pollution reduction developed for GBD MAPS. Differences between 2013 and 2030 largely reflect the impact of population aging and changes in the prevalence of diseases affected by exposure to air pollution. Differences between the different 2030 scenarios reflect the impact of energy policies and pollution control.

These findings are similar to those discussed by Apte and colleagues (2015) who used the GBD 2010 exposures and IER function in combination with WHO future cause-specific mortality projections to estimate that holding $PM_{2.5}$ exposures constant from 2010 to 2030 would result in a 23% increase in mortality attributable to $PM_{2.5}$ by 2030 in China, due to a large increase in the size of the population over age 50, which substantially offsets a small projected decrease in the rate of cardiovascular disease. Similarly, the Organisation for Economic Co-operation and Development (OECD) projected a 56% increase in the rate of deaths attributable to $PM_{2.5}$ in China between 2010 and 2030, even with a projection of essentially no change in ambient concentrations (OECD 2012). Importantly, while the absolute numbers of deaths and DALYs rises in 2030 for all scenarios and especially for BAU1, age-standardized death and DALY rates results show that this is largely due to projected changes in population size and age structure, and that reducing PM from major sources has real health benefits. Strict control of PM levels is critical to stabilize or reduce burden in the face of changing demographics (Figure 33).

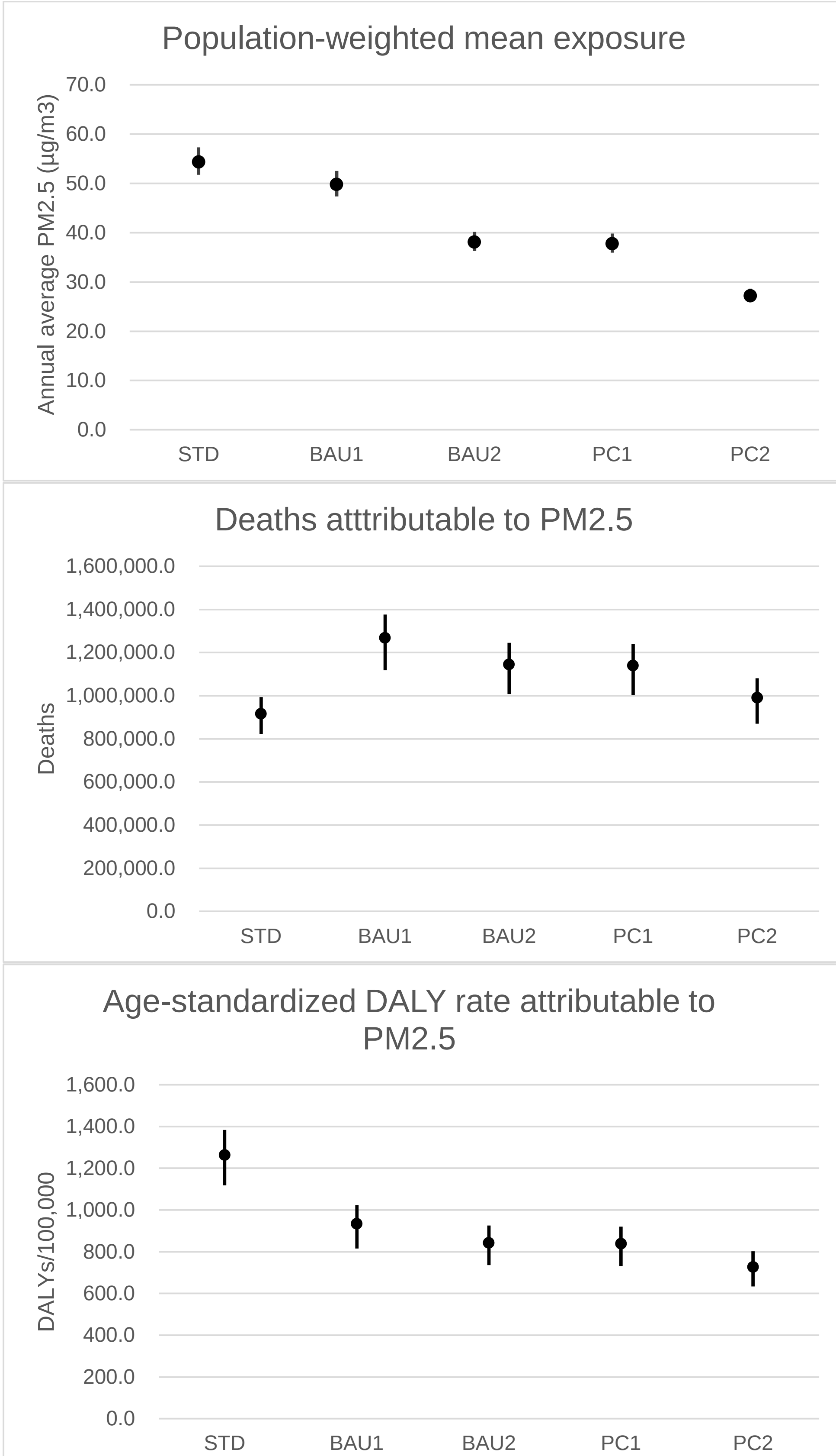


Figure 33. Year 2013 (STD) and projected (year 2030): population-weighted mean exposure, total deaths attributable to $PM_{2.5}$, and age-standardized DALYs/100,000 attributable to $PM_{2.5}$ for each of the four future scenarios. The bars show mean and upper and lower uncertainty intervals.

Further, the importance of coal combustion as a contributor to deaths attributable to $PM_{2.5}$ is also projected to increase under all future scenarios, such that coal combustion will remain the single largest sectoral contributor to ambient $PM_{2.5}$; whereas coal combustion was responsible for 40% of the mortality attributable to $PM_{2.5}$ in 2013, it is estimated to contribute to 44%, 55%, 47%, and 49% for BAU1, BAU2, PC1, and PC2, respectively. Even under the most stringent energy use and pollution-control future scenario, coal will remain the single largest sectoral contributor to ambient $PM_{2.5}$ in 2030. Further, the reductions in ambient $PM_{2.5}$ attributable to coal are lower than the reductions projected for the other sectors, resulting in an expected increase in the relative contribution of coal combustion to attributable disease burden. This finding highlights the urgent need for even more aggressive strategies to reduce emissions from coal combustion along with reductions in emissions from other sectors.

REFERENCES

Alexander B, Park RJ, Jacob DJ, Li QB, Yantosca RM, J Savarino, et al. 2005. Sulfate formation in sea-salt aerosols: constraints from oxygen isotopes. J Geophys Res–Atmos 110:D10; doi: 10.1029/2004JD005659.

Amann M, Kejun J, Jiming H, Wang S, Wei W, Jia X, et al. 2008. Scenarios for Cost-Effective Control of Air Pollution and Greenhouse Gases in China. Laxenburg, Austria:International Institute for Applied Systems Analysis.

Apte JS, Marshall JD, Cohen AJ, Brauer M. 2015. Addressing Global Mortality from Ambient $PM_{2.5}$. Environ Sci Technol 49(13):8057–8066

Bi XH, Feng YC, Wu JH, Wang YQ, Zhu T. 2007. Source apportionment of PM_{10} in six cities of northern China. Atmos Eviron 41:903–912.

Brauer M, Amann M, Burnett R, Cohen A, Dentener F, Ezzati M, et al. 2012. Exposure assessment for estimation of the global burden of disease attributable to outdoor air pollution. Environ Sci Technol 46(2):652–660; doi:10.1021/es2025752.

Brauer M, Freedman G, Frostad J, van Donkelaar A, Martin RV, Dentener F, et al. 2016. Ambient air pollution exposure estimation for the global burden of disease 2013. Environ Sci Technol 50:79–88; doi:10.1021/acs.est.5b03709.

Brook RD, Rajagopalan S, Pope CA 3rd, Brook JR, Bhatnagar A, Diez-Roux AV, et al. 2010. Particulate matter air pollution and cardiovascular disease: an update to the scientific statement from the American Heart Association. Circulation 121:2331–2378; doi:10.1161/CIR.0b013e3181dbece1.

Burnett RT, Pope CA III, Ezzati M, Olives C, Lim SS, Mehta S, et al. 2014. An integrated risk function for estimating the global burden of disease attributable to ambient fine particulate matter exposure. Environ Health Persp (Online) 122(4):397. Available: *http://dx.doi.org/10.1289/ehp.1307049*.

Cao J, Yang C, Li J, Chen R, Chen B, Gu D, et al. 2011. Association between long-term exposure to outdoor air pollution and mortality in China: a cohort study. J Hazard Mater 186:1594–1600.

Chen D, Wang YX, McElroy MB, He K, Yantosca RM, Le Sager P. 2009. Regional CO pollution in China simulated by the high-resolution nested-grid GEOS-Chem model. Atmos Chem Phys 9:3825–3839.

Chen DS, Cheng SY, Liu L, Chen T, Guo XR. 2007. An integrated MM5–CMAQ modeling approach for assessing trans-boundary PM_{10} contribution to the host city of 2008 Olympic summer games — Beijing, China. Atmos Eviron 41:1237–1250.

Chen H, Goldberg MS, Villeneuve PJ. 2008. A systematic review of the relation between long-term exposure to ambient air pollution and chronic diseases. Rev Environ Health 23:243–297.

Chen R, Kan H, Chen B, Huang W, Bai Z, Song G, et al. 2012. Association of particulate air pollution with daily mortality: The China Air Pollution and Health Effects Study. Am J Epidemiol 175:1173–1181.

Chen Y, Ebenstein A, Greensone M, Li H. 2013. Evidence on the impact of sustained exposure to air pollution on life-expectancy from China's Huai River policy. Proc Natl Acad Sci USA 110:12936–12941.

Cheng AY, Chen DS, Li JB, Wang HY, Guo XR. 2007. The assessment of emission-source contributions to air quality by using a coupled MM5-ARPS-CMAQ modeling system: a case study in the Beijing metropolitan region, China. Environ Modell Softw 22:1601–1616.

China Electricity Council. 2011. [Annual Development Report of China's Power Industry]. Beijing, China:China Electric Power Press. [In Chinese.]

Cofala J, Bertok I, Borken-Kleefeld J, Heyes C, Klimont Z, Rafaj P, et al. 2012. Emissions of Air Pollutants for the World Energy Outlook 2012 Energy Scenarios. Laxenburg, Austria:International Institute for Applied Systems Analysis.

Dong GH, Zhang P, Sun B, Zhang L, Chen X, Ma N, et al. 2012. Long-term exposure to ambient air pollution and respiratory disease mortality in Shenyang, China: a 12-year population-based retrospective cohort study. Respiration 84:360–368.

Energy Research Institute in China (ERI). 2009. [China's Low Carbon Development Pathways by 2050: Scenario Analysis of Energy Demand and Carbon Emissions]. Beijing, China:China Environmental Science Press. [In Chinese.]

Energy Research Institute in China (ERI). 2010. [Guidebook for the Financing of Energy Efficiency and Renewable Energy Projects]. Beijing, China:China Environmental Science Press. [In Chinese.]

Fairlie TD, Jacob DJ, Park RJ. 2007. The impact of transpacific transport of mineral dust in the United States. Atmos Environ 41(6):1251–1266.

Forouzanfar MH, Alexander L, Anderson HR, Bachman VF, Biryukov S, Brauer M, et al. 2015. Global, regional, and national comparative risk assessment of 79 behavioural, environmental and occupational, and metabolic risks or clusters of risks in 188 countries, 1990–2013: a systematic analysis for the Global Burden of Disease Study 2013. Lancet 386:2287–2323.

Fountoukis C, Nenes A. 2007. ISORROPIA II: A computationally efficient thermodynamic equilibrium model for K^+-Ca^{2+}-Mg^{2+}-NH_4^+-Na^+-SO_4^{2-}-NO_3^--Cl^--H_2O aerosols. Atmos Chem Phys 7(17):4639–4659.

Fu T-M, Cao JJ, Zhang XY, Lee SC, Zhang Q, Han YM, et al. 2012. Carbonaceous aerosols in China: top-down constraints on primary sources and estimation of secondary contribution. Atmos Chem Phys 12:2725–2746; doi:10.5194/acp-12-2725-2012.

Geng G, Zhang Q, Martin RV, van Donkelaar A, Huo H, Che H, et al. 2015. Estimating long-term $PM_{2.5}$ concentrations in China using satellite-based aerosol optical depth and a chemical transport model. Remote Sensing Environ 166:262–270.

Hao JM, Wang LT, Shen MJ, Li L, Hu JN. 2007. Air quality impacts of power plant emissions in Beijing. Environ Pollut 147:401–408.

Heald CL, Collett JL Jr, Lee T, Benedict KB, Schwandner FM, Li Y, et al. 2012. Atmospheric ammonia and particulate inorganic nitrogen over the United States. Atmos Chem Phys 12:10295–10312.

Heald CL, Jacob DJ, Jones DBA, Palmer PI, Logan JA, Streets DG, et al. 2004. Comparative inverse analysis of satellite (MOPITT) and aircraft (TRACE-P) observations to estimate Asian sources of carbon monoxide. J Geophys Res 109:D23306; doi:10.1029/2004JD005185.

Health Effects Institute. 2010. Outdoor Air Pollution and Health in the Developing Countries of Asia: A Comprehensive Review. Special Report 18. Boston, MA:Health Effects Institute.

HEI NPACT Review Panel. 2013. Executive Summary. HEI's National Particle Component Toxicity (NPACT) Initiative. Boston, MA:Health Effects Institute.

Henze DK, Seinfeld JH. 2006. Global secondary organic aerosol from isoprene oxidation. Geophys Res Lett 33:L09812; doi:10.1029/2006GL025976.

Henze DK, Seinfeld JH, Ng NL, Kroll JH, Fu T-M, Jacob DJ, et al. 2008. Global modeling of secondary organic aerosol formation from aromatic hydrocarbons: High- vs. low-yield pathways. Atmos Chem Phys 8:2405–2420.

Jiang X, Zhang Q, Zhao H, Geng G, Peng L, Guan D, et al. 2015. Revealing the hidden health costs embodied in Chinese exports. Environ Sci Tech 49:4381–4388.

Kharol SK, Martin RV, Philip S, Vogel S, Henze DK, Chen D, et al. 2013. Persistent sensitivity of Asian aerosol to emissions of nitrogen oxides. Geophys Res Lett 40(5):1021–1026; doi:10.1002/grl.50234.

Laden F, Schwartz J, Speizer FE, Dockery DW. 2006. Reduction in fine particulate air pollution and mortality: extended follow-up of the Harvard Six Cities Study. Am J Respir Crit Care Med 173(6):667–672.

Lei Y, Zhang Q, He KB, Streets DG. 2011. Primary anthropogenic aerosol emission trends for China, 1990–2005. Atmos Chem Phys 11:931–954; doi:10.5194/acp-11-931-2011.

Lelieveld J, Evans JS, Fnais M, Giannadaki D, Pozzer A. 2015. The contribution of outdoor air pollution sources to premature mortality on a global scale. Nature 525(7569):367–371. Available: *www.nature.com/doifinder/10.1038/nature15371*.

Li L, Chen C, Huang C, Huang HY, Li ZP, Fu JS, et al. 2008. [Regional air pollution characteristics: simulation of O_3 and $PM_{2.5}$ over Yangtze River Delta region]. Huan Jing Ke Xue 29:238–245. [In Chinese.]

Liao H, Henze DK, Seinfeld JH, Wu S, Mickley LJ. 2007. Biogenic secondary organic aerosol over the United States: Comparison of climatological simulations with observations. J Geophys Res 112:D06201; doi:10.1029/2006JD007813.

Lim SS, Vos T, Flaxman AD, Danaei G, Shibuya K, Adair-Rohani H, et al. 2012. A comparative risk assessment of burden of disease and injury attributable to 67 risk factors and risk factor clusters in 21 regions, 1990–2010: a systematic analysis for the Global Burden of Disease Study 2010. Lancet 380:2224–2260.

Lin J-T, Pan D, Davis SJ, Zhang Q, He K, Wang C, et al. 2014. China's international trade and air pollution in the United States. Proc Natl Acad Sci USA 111:1736–1741; doi:10.1073/pnas.1312860111.

Loomis D, Grosse Y, Lauby-Secretan B, El Ghissassi F, Bouvard V, Benbrahim-Tallaa L, et al. 2013. The carcinogenicity of outdoor air pollution. Lancet Oncol 14:1262–1263.

Lou S, Liao H, Zhu B. 2014. Impacts of aerosols on surface-layer ozone concentrations in China through heterogeneous reactions and changes in photolysis rates. Atmos Environ 85:123–138; doi:10.1016/j.atmosenv.2013.12.004.

Ministry of Environmental Protection of China (MEP). 2014. [Bulletin of Urban Sewage Treatment Facilities, and Flue Gas Desulfurization/Denitrification Facilities of Coal-fired Power Plants]. Beijing. [In Chinese.]

National Bureau of Statistics (NBS). 2014a. China Energy Statistical Yearbook 2013. Beijing, China:China Statistics Press.

National Bureau of Statistics (NBS). 2014b. China Industrial Economy Statistical Yearbook 2014. Beijing, China:China Statistics Press.

National Bureau of Statistics (NBS). 2014c. China Statistical Yearbook 2014. Beijing, China:China Statistics Press.

National Development and Reform Commission (NDRC). 2011. [Bulletin of the progress in shutting down small power generation units]. Available: *http://www.sdpc.gov.cn/zcfb/zcfbgg/2011gg/t20110422_407267.htm* [accessed 5 May 2012]. [In Chinese.]

Newby DE, Mannucci PM, Tell GS, Baccarelli AA, Brook RD, Donaldson K, et al. 2015. Expert position paper on air pollution and cardiovascular disease. Eur Heart J 36:83–93b; doi:10.1093/eurheartj/ehu458.

Ohara T, Akimoto H, Kurokawa J, Horii N, Yamaji K, Yan X, et al. 2007. An Asian emission inventory of anthropogenic emission sources for the period 1980–2020. Atmos Chem Phys 7:4419–4444.

Organisation for Economic Co-operation and Development (OECD). 2012. OECD Environmental Outlook to 2050. Available: *http://dx.doi.org/10.1787/9789264122246-en*.

Park RJ, Jacob DJ, Chin M, Martin RV. 2003. Sources of carbonaceous aerosols over the United States and implications for natural visibility. J Geophys Res 108:4355; doi:10.1029/2002JD003190.

Park RJ, Jacob DJ, Field BD, Yantosca RM, Chin M. 2004. Natural and transboundary pollution influences on sulfate-nitrate-ammonium aerosols in the United States: implications for policy. J Geophys Res 109:D15204; doi:10.1029/2003JD004473.

Philip S, Martin RV, van Donkelaar A, Wai-Ho Lo J, Wang Y, Chen D, et al. 2014. Global Chemical Composition of Ambient Fine Particulate Matter for Exposure Assessment. Environ Sci Technol 48:13060–13068; doi: 10.1021/es502965b.

Pope CA 3rd, Burnett RT, Thun MJ, Calle EE, Krewski D, Ito K, et al. 2002. Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. JAMA 287(9):1132–1141.

Pope CA 3rd, Burnett RT, Turner MC, Cohen A, Krewski D, Jerrett M, et al. 2011. Lung cancer and cardiovascular disease mortality associated with ambient air pollution and cigarette smoke: shape of the exposure–response relationships. Environ Health Perspect 119:1616–1621; doi:10.1289/ehp.1103639.

Pye HOT, Chan AWH, Barkley MP, Seinfeld JH. 2010. Global modeling of organic aerosol: the importance of reactive nitrogen (NO_x and NO_3). Atmos Chem Phys 10:11261–11276; doi:10.5194/acp-10-11261-2010.

Pye HOT, Liao H, Wu S, Mickley LJ, Jacob D, Henze DK, et al. 2009. Effect of changes in climate and emissions on future sulfate-nitrate-ammonium aerosol levels in the United States. J Geophys Res 114:D01205.

Sava F, Carlsten C. 2012. Respiratory health effects of ambient air pollution: an update. Clin Chest Med 33:759–769; doi:10.1016/j.ccm.2012.07.003.

Shah PS, Balkhair T, Knowledge Synthesis Group on Determinants of Preterm/LBW births. 2011. Air pollution and birth outcomes: a systematic review. Environ Int 37:498–516; doi:10.1016/j.envint.2010.10.009.

Stieb DM, Chen L, Eshoul M, Judek S. 2012. Ambient air pollution, birth weight and preterm birth: a systematic review and meta-analysis. Environ Res 117:100–111; doi:10.1016/j.envres.2012.05.007.

The State Council of the People's Republic of China. 2006. [The Eleventh Five-Year Plan for National Economic and Social Development of the People's Republic of China]. Available: *http://www.gov.cn/gongbao/content/2006/content_268766.htm* [accessed 2 Feb 2013]. [In Chinese.]

The State Council of the People's Republic of China. 2012. [The Twelfth Five-Year Plan for Energy Saving and Emission Reduction]. Available: *http://www.gov.cn/zwgk/2012-08/21/content_2207867.htm* [accessed: 2 Feb 2013]. [In Chinese.]

The State Council of the People's Republic of China. 2013. [The Action Plan for Control and Prevention of Air Pollution]. Available: *http://www.gov.cn/zwgk/2013-09/12/content_2486773.htm* [accessed 10 Sep 2013]. [In Chinese.]

Tsinghua University Building Energy Research Center (THUBERC). 2009. [Annual Report on China Building Energy Efficiency]. Beijing, China:China Architecture & Building Press. [In Chinese.]

U.S. Environmental Protection Agency. 2009. Final Report: Integrated Science Assessment for Particulate Matter. EPA/600/R-08/139F. Washington, DC:U.S. EPA.

van Donkelaar A, Martin RV, Brauer M, Boys BL. 2015. Use of satellite observations for long-term exposure assessment of global concentrations of fine particulate matter. Environ Health Perspect 123:135–143; doi:10.1289/ehp.1408646.

van Donkelaar A, Martin RV, Brauer M, Kahn R, Levy R, Verduzco C, et al. 2010. Global estimates of ambient fine particulate matter concentrations from satellite-based aerosol optical depth: development and application. Environ Health Perspect 118(6):847; doi:10.1289/ehp.0901623.

Walker JM, Philip S, Martin RV, Seinfeld H. 2012. Simulation of nitrate, sulfate, and ammonium aerosols over the United States. Atmos Chem Phys 12:11213–11227.

Wang DX, Hu JL, Xu Y, Lv D, Xie XY, Kleeman, M, et al. 2014. Source contributions to primary and secondary inorganic particulate matter during a severe wintertime $PM_{2.5}$ pollution episode in Xi'an, China. Atmos Eviron 97:182–194.

Wang HL, Zhuang YH, Wang Y, Sun YL, Yuan H, Zhuang GS, et al. 2008. Long-term monitoring and source apportionment of $PM_{2.5}$/PM_{10} in Beijing, China. J Environ Sci 20:1323–1327.

Wang LT, Wei Z, Yang J, Zhang Y, Zhang FF, Su J, et al. 2014. The 2013 severe haze over southern Hebei, China: model evaluation, source apportionment, and policy implications. Atmos Chem Phys 14:3151–3173.

Wang S, Zhang Q, Martin RV, Philip S, Liu F, Li M, et al. 2015. Satellite measurements oversee China's sulfur dioxide emission reductions from coal-fired power plants. Environ Res Lett 10:114015.

Wang SX, Hao JM. 2012. Air quality management in China: issues, challenges, and options. J Environ Sci (China) 24:2–13; doi:10.1016/S1001-0742(11)60724-9.

Wang SX, Xing J, Chatani S, Hao JM, Klimont Z, Cofala J, et al. 2011. Verification of anthropogenic emissions of China by satellite and ground observations. Atmos Environ 45:6347–6358.

Wang SX, Zhao B, Cai SY, Klimont Z, Nielsen CP, Morikawa T, et al. 2014. Emission trends and mitigation options for air pollutants in East Asia. Atmos Chem Phys 14:6571–6603; doi:10.5194/acp-14-6571–2014.

Wang Y, Zhang QQ, He K, Zhang Q, Chai L. 2013. Sulfate-nitrate-ammonium aerosols over China: response to 2000–2015 emission changes of sulfur dioxide, nitrogen oxides, and ammonia. Atmos Chem Phys 13:2635–2652.

Wang Y, Zhang QQ, Jiang J, Zhou W, Wang B, He F, et al. 2014. Enhanced sulfate formation during China's severe winter haze episode in January 2013 missing from current models. J Geophys Res–Atmos 119(17); doi:10.1002/2013JD021426.

Wang YX, McElroy MB, Jacob DJ, Yantosca RM. 2004. A nested grid formulation for chemical transport over Asia: applications to CO. J Geophys Res–Atmos 109:D22; doi:10.1029/2004jd005237.

Wong CM, Vichit-Vadakan N, Kan H, Qian Z. 2008. Public Health and Air Pollution in Asia (PAPA): a multicity study of short-term effects of air pollution on mortality. Environ Health Perspect 116:1195–1202.

Wei W, Wang SX, Hao JM, Cheng SY. 2011. Projection of anthropogenic volatile organic compounds (VOCs) emissions in China for the period 2010–2020. Atmos Environ 45:6863–6871; doi:10.1016/j.atmosenv.2011.01.013.

World Health Organization (WHO). 2005. WHO Air quality guidelines for particulate matter, ozone, nitrogen dioxide and sulfur dioxide. Global Update 2005. Available: *http://www.euro.who.int/__data/assets/pdf_file/0005/78638/E90038.pdf?ua=1.*

World Health Organization (WHO). 2013. Review of Evidence on Health Aspects of Air Pollution – REVIHAAP Project Technical Report. WHO Regional Office for Europe. Copenhagen.

Wu L, Feng YC, Wu JH, Zhu T, Bi XH, Han B, et al. 2009. Secondary organic carbon quantification and source apportionment of PM_{10} in Kaifeng, China. J Environ Sci 21:1353–1362.

Wu Y, Zhang SJ, Li ML, Ge YS, Shu JW, Zhou Y, et al. 2012. The challenge to NO_x emission control for heavy-duty diesel vehicles in China. Atmos Chem Phys 12:9365–9379; doi:10.5194/acp-12-9365-2012.

Xing J, Wang SX, Chatani S, Zhang CY, Wei W, Hao JM, et al. 2011. Projections of air pollutant emissions and its impacts on regional air quality in China in 2020. Atmos Chem Phys 11:3119–3136; doi:10.5194/acp-11-3119-2011.

Xu J-W, Martin RV, van Donkelaar A, Kim J, Choi M, Zhang Q, et al. 2015. Estimating ground-level $PM_{2.5}$ in eastern China using aerosol optical depth determined from the GOCI satellite instrument. Atmos Chem Phys 15:13133–13144; doi:10.5194/acp-15-13133-2015.

Yang F, Tan J, Zhao Q, Du Z, He K, Ma Y, et al. 2011. Characteristics of $PM_{2.5}$ speciation in representative megacities and across China. Atmos Chem Phys 11:5207–5219.

Yang LX, Cheng SH, Wang XF, Nie W, Xu PJ, Gao XM, et al. 2013. Source identification and health impact of $PM_{2.5}$ in a heavily polluted urban atmosphere in China. Atmos Eviron 75:265–269.

Zhang HL, Li JY, Ying Q, Yu JZ, Wu D, Cheng Y, et al. 2012. Source apportionment of $PM_{2.5}$ nitrate and sulfate in China using a source-oriented chemical transport model. Atmos Eviron 62:228–242.

Zhang L, Liu L, Zhao Y, Gong S, Zhang X, Henze DK, et al. 2015. Source attribution of particulate matter pollution over North China with the adjoint method. Environ Res Lett 10(8):084011; doi:10.1088/1748-9326/10/8/084011.

Zhang P, Dong G, Sun B, Zhang L, Chen X, Ma N, et al. 2011. Long-term exposure to ambient air pollution and mortality due to cardiovascular disease and cerebrovascular disease in Shenyang, China. PLoS One 6:e20827; doi:10.1371/journal.pone.0020827.

Zhou M, Liu Y, Wang L, Kuang X, Xu X, Kan H. 2014. Particulate air pollution and mortality in a cohort of Chinese men. Environ Pollut 186:1–6.

Zhao B, Wang SX, Dong XY, Wang JD, Duan L, Fu X, et al. 2013a. Environmental effects of the recent emission changes in China: implications for particulate matter pollution and soil acidification. Environ Res Lett 8(2):024031; doi:10.1088/1748-9326/8/2/024031.

Zhao B, Wang SX, Wang JD, Fu J, Liu TH, Xu JY, et al. 2013b. Impact of national NO_x and SO_2 control policies on particulate matter pollution in China. Atmos Environ 77:453–463.

Zhao B, Wang SX, Xu JY, Fu K, Klimont Z, Hao JM, et al. 2013c. NO_x emissions in China: historical trends and future perspectives. Atmos Chem Phys 13:1–29; doi:10.5194/acp-13-1-2013.

Zhao Y, Zhang J, Nielsen CP. 2013d. The effects of recent control policies on trends in emissions of anthropogenic atmospheric pollutants and CO_2 in China. Atmos Chem Phys 13:487–508; doi:10.5194/acp-13-487-2013.

Zhao Y, Wang SX, Duan L, et al. 2008. Primary air pollutant emissions of coal-fired power plants in China: current status and future prediction. Atmos Environ 42:8442–8452; doi: 10.1016/j.atmosenv.2008.08.021.

Zhou M, Wang H, Zhu J, Chen W, Wang L, Liu S, et al. 2015. Cause-specific mortality for 240 causes in China during 1990–2013: a systematic subnational analysis for the Global Burden of Disease Study 2013. Available: *http://dx.doi.org/10.1016/S0140-6736(15)00551-6.*

APPENDIX I. POPULATION-ATTRIBUTABLE FRACTION DUE TO SPECIFIC SOURCES OF FINE PM FOR CURRENT CONDITIONS AND FUTURE REDUCTION SCENARIOS

POPULATION-ATTRIBUTABLE FRACTION FOR CURRENT CONDITIONS

For any $PM_{2.5}$ concentration z, suppose the contribution to z from a specific source is estimated in terms of the proportion of z, denoted by p. A person is exposed to multiple sources of PM simultaneously. Consequently we cannot attribute changes in exposure to any source over a specific range in concentration. We only know the magnitude of the change as $p \times z$. Since our IER risk function is nonlinear, we therefore cannot directly estimate the change in risk associated with a $p \times z$ change in exposure. If the risk function were linear in concentration, then all changes of $p \times z$ would yield the same change in risk; thus, we do not need to know where in the exposure distribution such a change occurs.

We suggest that the average relative risk for changes of $p \times z$ among all values of z represents a reasonable summary relative risk estimate of such changes. We show in the Appendix II that if we reformulate the IER relative risk function in p by

$$\frac{IER(z)}{IER((1-p)\times z)} = 1 + \{IER(z) - 1\} \times p\,,$$

then $\{\mathrm{IER}(z) - 1\} \times p$ represents the average change in relative risk over the interval from 0 to z of change in concentration of $p \times z$. Given such a reformulation, we also show in Appendix II that $PAF(z,(1-p) \times z) = p \times PAF(z)$. That is, the proportional attributable fraction due to a change in concentration from z to $(1-p) \times z$ can be represented by the proportion times the PAF at concentration z.

POPULATION-ATTRIBUTABLE FRACTION FOR FUTURE REDUCTION SCENARIOS

Suppose a specific air pollution mitigation strategy is designed to reduce emissions from a source such as coal. If we are only interested in the change in PAF associated with the change in exposure due to that source, then we hypothesize that the $PM_{2.5}$ concentration is reduced from z to $(1-p) \times z$ for proportion p. The PAF of the current versus future case is given by

$$PAF(z,(1-p)\times z) = \frac{IER(z) - IER((1-p)\times z)}{IER(z)}.$$

Further suppose we can decompose the reduction in $PM_{2.5}$ associated with a specific source by industry/sector contributions such that $p \times z = \sum_{s=1}^{S} p_s \times z$. Since we do not know over which concentration interval a reduction occurs due to any specific industry/sector, we estimate the PAF attributable to that industry/sector by the proportional method with

$$PAF^{(S)} = p_S \times PAF(z,(1-p)\times z)\,.$$

Now suppose we are interested in the joint impact of burden due to changes in a specific source and in addition the burden associated with exposure from all other sources combined. In this case we, as with the current condition analysis, do not know where in the exposure range the contribution from the source and remaining exposure from all other sources occurs. We estimate the PAF of these two contributors to total exposure by $p \times PAF(z)$ associated with the source and $(1 - p) \times PAF(z)$ associated with all other sources.

We thus use the proportional PAF formulation for both current and future conditions.

APPENDIX II: ESTIMATING THE POPULATION-ATTRIBUTABLE FRACTION FROM A SPECIFIC FINE PM SOURCE

We wish to estimate the change in relative risk (i.e., excess risk) for a change in ambient $PM_{2.5}$ concentrations denoted by z say, associated with a specific source of PM whose concentration is denoted by, $z_S < z$. One approach would be to evaluate the risk function at current ambient concentrations and at concentrations minus the estimated source-specific contribution. However, since the relative risk function used by GBD, termed the integrated exposure–response (IER) function, is not linear, changes in risk will depend not only on the magnitude of the change in concentration but where on the risk curve such a change occurs.

Although we can estimate what proportion of ambient concentrations are attributable to source S, we are not able to know over what range of concentration at any location did this source contribute to the observed value. Each source of pollution contributes to the total concentration, but they do not do so in distinctive intervals of concentration.

Furthermore, evaluating changes in risk associated with changes in source-specific exposure using a nonlinear function by subtracting the source contribution from the ambient level results in a logical inconsistency, in that the sum of the contribution from all sources that add up to the ambient fine particulate mass concentration will not equal the risk attributable to that concentration.

THE APPROACH

Each source of pollution contributes exposure throughout the concentration range. As such, we would like to apply a change in risk over all concentrations within a specific range that is the average of the change in risk of the IER risk function over that range. To do this we consider a risk model that is linear in concentration of the form

$$L(z, z_{cf}, \beta) = 1 + \beta * (z - z_{cf}),$$

for slope β. We thus seek to find β such that the average of incremental changes in the IER risk function equals the average of incremental changes in L over the range (z_{cf}, z_0), for any observed ambient concentration z_0 say.

We define an incremental change in relative risk at any concentration z to equal to the derivative of the function with respect to z. Let the GBD 2013 relative-risk function be denoted by f and its derivative with respect to z by f'. Thus the derivative of L with respect to z is β. The average of functions over their arguments can be obtained by integrating over the function on the interval $[z_{cf}, z_0]$ and dividing by the length of the interval $z_0 - z_{cf}$. That is

$$\beta = \frac{1}{z_0 - z_{cf}} \int_{z_{cf}}^{z_0} f'(z)dz = (f(z_0) - 1)/(z_0 - z_{cf})$$

since $f(z_{cf}) = 1$. Based on this construction we note that $f(z_0) = L(z_0)$. Then

$$L(z, z_{cf}, \beta) = 1 + \big((f(z_0) - 1)/(z_0 - z_{cf})\big) * (z - z_{cf}).$$

That is, L represents a line from 1 at z_{cf} to $f(z_0)$ at z_0. L is graphically displayed in Figure II.1 (dashed line) with the relative risk function f for $z_0 = 200$ µg/m^3.

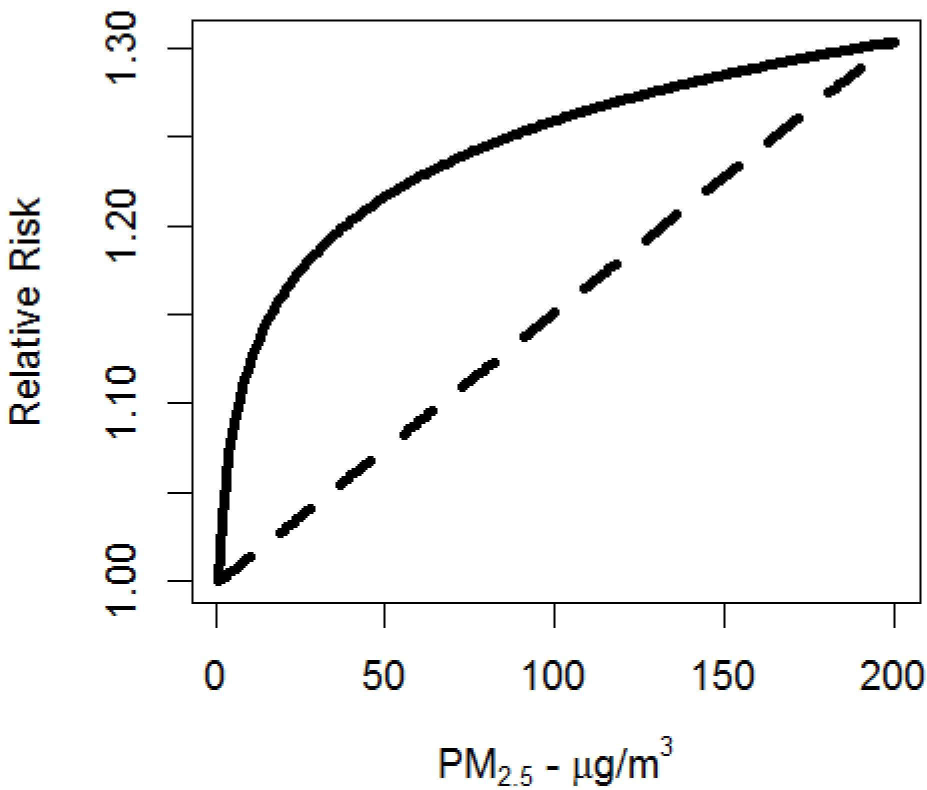


Figure II.1. GBD 2013 relative risk function (solid red line) and linear model (dashed black line) with equivalent average change in risk.

The slope of the line, $\beta = (f(z_0) - 1)/(z_0 - z_{cf})$, is greater than the change in relative risk for larger concentrations and is less than the change in relative risk for smaller concentrations, thus yielding the average of all changes over the range $[z_{cf}, z_0]$.

Logical Inconsistency Due to Positive Counterfactual

Since $z_{cf} > 0$, there could arise cases where $z_0 - z_{(z_0 z_S)} < z_{cf}$, where $z_{(z_0 z_S)}$ is the concentration attributable to source S at the location whose ambient concentration of $PM_{2.5} = z_0$. Thus, not all of the mass contribution from source S would be taken into account in the estimate of risk. To avoid this problem and ensure that the excess risk attributable to the sum of all sources of PM equals the excess risk attributable to the total PM mass, we alter our estimate of excess risk ΔR , denoted by $\Delta\widetilde{R}$ as

$$\Delta\widetilde{R}(z_S, z_0, z_{cf}) = \{ f(z_0) - 1\} * \left(\frac{z_S - z_{cf}}{z_0 - z_{cf}} \right) \equiv \{ f(z_0) - 1\} * p_{(z_S, z_0)}$$

where $p_{(z_S, z_0)}$ is the proportion of mass at location with concentration z_0 attributable to source S assuming values cannot be below the counterfactual. We thus have the desired property

$$\sum_{i=1}^{N} \Delta\widetilde{R}(z_{Si}, z_0, z_{cf}) = \Delta\widetilde{R}(\sum_{i=1}^{N} z_{Si}, z_0, z_{cf}) = \Delta\widetilde{R}(z_0, z_0, z_{cf}) = \{ f(z_0) - 1\}$$

for the N source concentrations $(z_{S_1}, ..., z_{S_N})$ and the fact that $\sum_{i=1}^{N} p_{(z_{Si}, z_0)} = 1$. This definition is also convenient since the source contributions are estimated as percentages from atmospheric models without any attribution to ambient levels.

Comparison to Proportional Source Approach

Here we show that the Proportional Source method, which estimates the PAF attributable to a proportional change in concentration from a specific source, p, say is $p * PAF$, where PAF is the PAF associated with the ambient concentration based on all sources, is equivalent to the PAF assuming the linear transformation in risk given by

$$1 + (f(z) - 1) * P ,$$

where P is the proportional change from the ambient concentration z to the counterfactual. Here setting $P = 1$ retains the original relative risk function $f(z)$.

The ratio of relative risk functions for $P = 1$ to $P = 1 - p$ due to the proportion of the ambient concentration, p, attributable to a source is given by

$$RR(z, p) = \frac{1 + (f(z) - 1) * 1}{1 + (f(z) - 1) * (1 - p)} = \frac{f(z)}{1 + (f(z) - 1) * (1 - p)} == \frac{f(z)}{f(z) * (1 - p) + p}$$

The PAF of this relative risk is (after some simple algebra!)

$$PAF(z, p) = \frac{RR(z, p) - 1}{RR(z, p)} = p * \left(\frac{f(z) - 1}{f(z)} \right) = p * PAF(z, 1)$$

where $PAF(z,1)$ represents the PAF associated with the ambient concentration itself.

Thus we conclude that assuming proportional changes of the PAF due to proportional changes in concentration is equivalent to assuming the relative risk is linear with respect to the proportional change in concentration with slope equal to the average change in risk from the counterfactual to any specific concentration.

APPENDIX III: DETAILED TABULATIONS OF EMISSIONS BY PROVINCE AND SECTOR

Table III.1. Emissions by Province (2013)/kt

	SO_2	NO_x	PM_{10}	$PM_{2.5}$	BC	OC	NMVOC	NH_3
Beijing	166.4	342.6	80.9	56.4	13.7	12.3	347.2	52.2
Tianjin	243.5	341.5	143.4	107.9	16.3	25.7	324.4	45.4
Hebei	998.0	1511.1	1159.9	868.3	154.7	225.4	1475.1	627.5
Shanxi	989.3	901.8	775.7	568.2	129.9	147.3	694.5	197.3
Inner Mongolia	1122.4	1145.8	629.4	472.1	111.9	139.0	652.3	366.4
Liaoning	940.3	1141.1	633.8	472.6	69.0	131.3	1042.6	382.3
Jilin	366.9	628.4	465.2	350.6	53.5	104.8	491.5	271.9
Heilongjiang	298.5	743.7	569.8	449.2	79.5	170.5	648.8	345.7
Shanghai	578.4	346.9	142.0	96.2	11.2	9.5	653.3	41.0
Jiangsu	1028.5	1426.4	894.5	647.8	76.5	161.3	2009.4	461.6
Zhejiang	1172.7	1061.0	411.8	276.8	31.4	47.7	1645.9	173.2
Anhui	611.5	1009.9	806.2	610.1	89.5	215.7	1041.3	417.1
Fujian	549.6	730.1	318.5	223.2	27.5	49.4	680.5	166.7
Jiangxi	411.3	559.5	433.0	283.9	36.8	67.4	480.7	238.5
Shandong	2361.2	2625.5	1441.1	1053.7	163.8	254.8	2481.9	789.5
Henan	1060.3	1796.3	1278.8	927.5	126.7	227.3	1418.5	954.4
Hubei	1235.2	1114.1	830.7	594.2	109.8	168.1	951.9	435.5
Hunan	803.1	849.4	735.4	523.5	89.3	147.0	799.4	483.4
Guangdong	996.1	1522.0	629.9	441.8	54.7	116.7	1609.0	365.5
Guangxi	743.9	699.2	649.4	492.9	50.5	143.9	750.4	352.3
Hainan	110.1	133.7	61.0	45.2	4.5	13.1	143.4	58.7
Chongqing	995.0	489.7	326.8	236.1	39.1	70.8	409.0	175.4
Sichuan	1593.5	1052.6	815.0	627.2	96.0	247.6	1249.0	785.7
Guizhou	1057.0	598.1	510.8	393.4	86.5	134.4	370.8	248.8
Yunnan	487.3	631.4	431.0	323.7	55.8	97.0	446.7	337.1
Tibet	5.9	24.4	10.2	8.2	1.3	2.6	16.0	97.8
Shaanxi	725.1	670.6	480.0	362.1	74.0	122.6	554.9	210.6
Gansu	277.9	366.6	280.6	215.8	35.3	67.2	272.8	165.2
Qinghai	56.3	117.3	76.0	57.5	9.2	12.6	61.9	86.3
Ningxia	210.0	193.7	123.2	87.9	13.3	15.5	94.7	40.2
Xinjiang	933.7	848.8	378.7	280.1	43.8	74.7	379.2	248.1
China	**23129.0**	**25622.9**	**16522.9**	**12154.0**	**1954.9**	**3423.0**	**24197.0**	**9621.3**

Table III.2. Emissions by Sector (2013)/kt

	SO_2	NO_x	PM_{10}	$PM_{2.5}$	BC	OC	NMVOC	NH_3
Power plants	6275.4	6463.6	1034.2	612.1	8.1	14.9		
Industrial combustion	7205.3	4385.1	1533.5	1028.3	142.5	41.1	129.2	
Industrial process	5624.9	5909.8	7546.6	4873.2	610.3	511.5	7132.8	215.0
Cement	1704.0	2884.8	2985.1	1866.7	11.3	33.9	291.1	
Steel	1859.8	532.6	1388.3	1024.2	37.7	48.2	269.0	
Domestic sources	2959.7	1087.5	4295.9	3853.2	951.9	2320.0	4653.0	918.6
Biofuel	72.4	477.9	2970.8	2878.0	503.7	1971.4	3353.6	
Transportation	973.5	7249.8	364.8	345.6	183.9	75.0	2913.0	
On-road	644.0	5138.2	121.2	114.8	52.4	33.5	2044.2	
Off-road	329.5	2111.6	243.6	230.8	131.5	41.5	868.9	
Solvent use							8155.3	
Others	90.2	527.1	1747.9	1441.6	58.3	460.6	1213.8	8487.6
Biomass open burning	90.2	527.1	1747.9	1441.6	58.3	460.6	1213.8	
Livestock farming								5489.8
Mineral fertilizer application								2997.9
National total emissions	**23,129.0**	**25,622.9**	**16,522.9**	**12,154.0**	**1954.9**	**3423.0**	**24,197.0**	**9621.3**

APPENDIX IV: DETAILED DESCRIPTION OF ASSUMPTIONS AND DEVELOPMENT OF THE ENERGY AND EMISSION SCENARIOS

DEVELOPMENT OF ENERGY SCENARIOS

We assume the annual average GDP growth rate to decrease gradually from 8.0% during 2011–2015 to 5.5% during 2026–2030, respectively (S.X. Wang et al. 2014). The national population is projected to increase from 1.36 billion in 2013 to 1.44 billion in 2020 and 1.47 billion in 2030, and the urbanization rate (proportion of people in urban areas) is assumed to increase from 53.7% in 2013 to 58% and 63% in 2020 and 2030, respectively.

For power plants, the PC scenario considers aggressive development plans for clean and renewable energy power generation. Therefore, the proportion of electricity production from coal-fired power plants is expected to decrease to 49% in 2020 and 41% in 2030 in PC scenarios, contrasted by 57% and 48% in BAU scenarios.

For the industry sector, we projected lower yields of energy-intensive industrial products in the PC scenario than in those of the BAU scenario because of a more conservative life style. The shares of less energy-intensive technologies are assumed to be higher in the PC scenario than in the BAU scenario.

For the residential sector, China's building area per capita in the PC scenario is expected to be 2–3 m^2 lower than that of the BAU scenario in both urban and rural areas. The energy demand for heating per unit area is somewhat lower in our PC scenario because of implementation of new energy conservation standards for the design of buildings. Replacement of coal and direct biomass burning with clean fuels are assumed in both urban and rural areas, with faster progress in the PC scenario.

For the transportation sector, the vehicle population per 1000 persons is projected at 380 and 325 in the BAU and PC scenarios, respectively. The PC scenario assumes an aggressive plan to promote electric vehicles and a progressive implementation of new fuel efficiency standards.

Figure IV.1 shows current and future energy consumption in China.

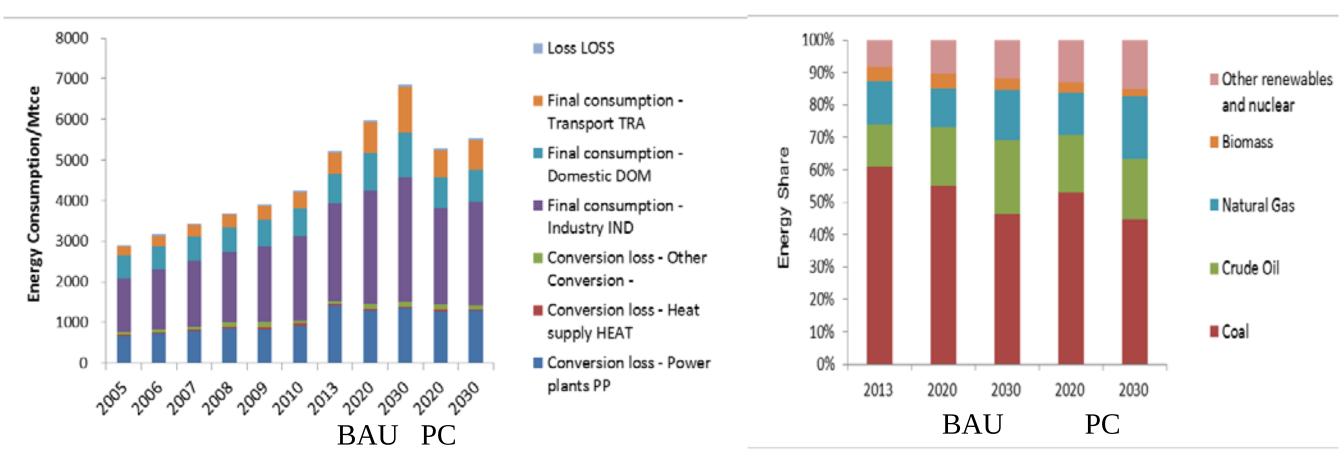


Figure IV.1. Energy consumption in China. Mtce = metric tons carbon equivalent.

By 2030, China's energy consumption is projected to increase by 31% and 7% from the 2013 level in the BAU and PC scenarios, respectively. Coal continues to dominate China's energy mix, but the proportion decreases from 61% in 2013 to 47% and 44% in 2030 under the BAU and PC scenarios, respectively. In contrast, the shares of natural gas and of other renewable energy and nuclear energy are estimated to increase from 13.5% and 8.3% in 2013 to 15.7% and 11.8% in 2030 under the BAU scenario, and 19.2% and 15.1% under the PC scenario, respectively.

DEVELOPMENT OF EMISSION SCENARIOS

The BAU[1]/PC[1] scenario is designed based on the Twelfth Five-Year Plan for Environmental Protection and the Clean Air Action Plan, and the assumption that high-efficiency control technologies will continue to spread gradually until 2030. The BAU[2]/PC[2] scenario assumes full application of the best available technologies in the world. Tables IV.1–IV.3 show the penetrations of major control technologies in key sectors.

Table IV.1. Penetrations of Major NO_X Removal Equipment Assumed in This Study (%)

					[1]		[2]	
Sector	Technology	Removal equipment	2010	2013	2020	2030	2020	2030
Power plants	Coal-fired power plants < 100 MW (exc. CFB)	NOC	11	0	0	0	0	0
		LNB	89	100	100	100	100	100
		LNB+SNCR	0	0	0	0	0	0
		LNB+SCR	0	0	0	0	0	0
	Coal-fired power plants ≥ 100 MW (exc. CFB)	NOC	11	6	3	1	0	0
		LNB	75	38	19	10	8	0
		LNB+SNCR	1	2	4	5	6	0
		LNB+SCR	12	54	74	84	86	100
	NGCC	NOC	21	15	7	4	0	0
		LNB	74	70	60	35	50	0
		LNB+SNCR	1	0	3	6	5	0
		LNB+SCR	5	15	30	55	45	100
Industry sector	Coal-fired industrial grate boiler	NOC	100	77	76	76	70	0
		LNB	0	18	18	18	20	0
		LNB+SNCR	0	0	0	0	0	0
		LNB+SCR	0	5	6	6	10	100
	Precalcined cement kiln < 2000 tons/day	NOC	70	58	39	19	20	0
		LNB	30	42	36	31	30	0
		LNB+SNCR	0	0	15	30	30	0
		LNB+SCR	0	0	10	20	20	100
	Precalcined cement kiln 2000–4000 tons/day	NOC	65	52	36	18	20	0
		LNB	35	47	39	32	30	0
		LNB+SNCR	0	1	15	30	30	0
		LNB+SCR	0	0	10	20	20	100

Table continues on next page

***Table IV.1 continued*. Penetrations of Major NO_x Removal Equipment Assumed in This Study (%)**

Industry Sector	Precalcined cement kiln ≥ 4000 tons/day	NOC	60	47	24	12	0	0
		LNB	40	52	26	13	0	0
		LNB+SNCR	0	1	30	45	60	0
		LNB+SCR	0	0	20	30	40	100
	Glass production – float process	NOC	100	76	52	26	0	0
		OXFL	0	16	24	56	50	0
		SCR	0	8	24	18	50	100
	Sintering	NOC	100	93	92	92	40	0
		SNCR	0	0	0	0	0	0
		SCR	0	7	8	8	60	100
	Nitric acid – dual pressure process	NOC	70	54	54	54	10	0
		ABSP	12	16	16	16	18	0
		SCR	18	30	30	30	72	100
		ABSP+SCR	0	0	0	0	0	0
	Nitric acid – other process	NOC	5	0	0	0	0	0
		ABSP	63	65	65	65	5	0
		SCR	32	30	30	30	15	0
		ABSP+SCR	0	5	5	5	80	100

Notes: CFB = Circulated Fluidized Bed, NOC = No Control, LNB = Low NO_x Burner, SCR = Selective Catalytic Reduction, SNCR = Selective Non-Catalytic Reduction, LNB+SCR = combination of LNB and SCR, LNB+SNCR = combination of LNB and SNCR, ABSP = Absorption Method, OXFL = Oxy-fuel Combustion Technology. The table gives the national average penetrations of major control technologies. However, the penetrations vary with provinces. The penetration of the key region is usually larger than that of other regions.

Table IV.2. Penetrations of Major SO_2 Removal Equipment Assumed in This Study (%)

Sector	Technology	Removal equipment	2010	2013	[1] 2020	[1] 2030	[2] 2020	[2] 2030
Power plants	Coal-fired power plants 100~300 MW (exc. CFB)	NOC	21	20	18	15	0	0
		FGD	79	80	82	85	100	100
	Coal-fired power plants ≥ 300 MW (exc. CFB)	NOC	7	7	5	3	0	0
		FGD	93	93	95	97	100	100
	CFB	NOC	47	47	34	20	0	0
		CFB-FGD	53	53	66	80	100	100
Industry sector	Coal-fired industrial grate boiler	NOC	4	0	0	0	0	0
		WET	95	70	68	68	68	0
		FGD	1	30	32	32	32	100
	Sintering	NOC	90	70	65	60	0	0
		FGD	10	30	35	40	100	100

Notes: CFB = Circulated Fluidized Bed, NOC = No Control, FGD = Flue Gas Desulfurization, CFB-FGD = Flue Gas Desulfurization for Circulated Fluidized Bed, WET = Wet Scrubber. The table gives the national average penetration of SO_2 removal equipment. However, the penetrations vary with provinces. The penetration of the key region is usually larger than that of other regions.

Table IV.3. Penetrations of Major PM Removal Equipment Assumed in This Study (%)

Sector	Technology	Removal equipment	2010	2013	[1] 2020	[1] 2030	[2] 2020	[2] 2030
Power plants	Coal-fired grate boiler	NOC	0	0	0	0	0	0
		CYC	12	10	0	0	0	0
		WET	88	90	100	100	100	100
	Pulverized coal combustion boilers	NOC	0	0	0	0	0	0
		CYC	0	0	0	0	0	0
		WET	0	0	0	0	0	0
		ESP	93	85	90	80	80	0
		FF	7	15	10	20	20	100
	CFB	NOC	0	0	0	0	0	0
		CYC	0	0	0	0	0	0
		WET	0	0	0	0	0	0
		ESP	100	85	90	80	80	0
		FF	0	15	10	20	20	100
Industry sector	Coal-fired industrial grate boiler	NOC	0	0	0	0	0	0
		CYC	0	0	0	0	0	0
		WET	95	85	85	85	60	0
		ESP	0	10	10	10	20	0
		FF	5	5	5	5	20	100
	Coal-fired industrial fluidized bed boiler	NOC	0	0	0	0	0	0
		WET	100	100	100	100	100	100
	Sintering - flue gas	NOC	0	0	0	0	0	0
		CYC	0	0	0	0	0	0
		WET	5	0	0	0	0	0
		ESP	75	75	75	75	70	0
		FF	20	25	25	25	30	100
	Sintering - fugitive	NOC	0	0	0	0	0	0
		CMN	60	50	50	50	30	0
		HIEF	40	50	50	50	70	100
Domestic sector	Coal-fired domestic boiler	NOC	8	0	0	0	0	0
		CYC	14	15	15	10	0	0
		WET	78	85	85	90	100	100

Notes: CFB = Circulated Fluidized Bed, NOC = No Control, CYC = Cyclone dust collect, WET = Wet Scrubber, ESP = Electrostatic precipitator, FF = Fiber filter, CMN = Common control of fugitive emissions, HIEF = High-efficiency control of fugitive emissions. The table gives the national average penetration of SO_2 removal equipment. However, the penetrations vary with provinces. The penetration of the key region is usually larger than that of other regions.

For power plants, the penetration of FGD is assumed to approach 100% by 2020. All new-built thermal power plants should be equipped with low-NO_x combustion technologies and flue gas denitrification (SCR/SNCR). Existing thermal power plants should be upgraded with low-NO_x combustion technologies, and large units (≥ 300 MW) should be upgraded with flue gas denitrification. SCR/SNCR will gradually penetrate to smaller units after 2013. For PM, HED would spread much more rapidly.

For the industry sector, FGDs are assumed to be promoted in large scales. Newly built industrial boilers are required to be equipped with LNB, and existing boilers in the “key regions” are beginning to be retrofitted with LNB. ESP and HED would be gradually promoted to replace the relatively low-efficient WET.

For the residential sector, HED and low-sulfur derived coal are assumed to be promoted gradually, both penetrating 20% and 40% of the total capacity by 2020 and 2030, respectively, in BAU[1]/PC[1]. In addition, we take into consideration the replacement with advanced coal stove, and advanced biomass stove (e.g., better combustion condition, catalytic stove) where applicable, which are beneficial for the reduction of PM and NMVOC. The BAU[2]/PC[2] scenario assumes the application of the best-available technology without considering economic cost.

For the transportation sector, In the BAU[1]/PC[1] scenario, all the current standards in Europe are assumed to be implemented in China gradually, and the time intervals between the releases of two-stage standards would be a little shorter than those of Europe. The implementation timeline of the emission standards is given in Figure IV.2.

Type	00	01	02	03	04	05	06	07	08	09	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
Light duty vehicle	1	1	1	1	1	2	2	2	3	3	3	4	4	4	4	4	5	5	5	5	6	6	6	6	6	6	6	6	6	6	6
Heavy duty diesel vehicle		1	1	1	2	2	2	3	3	3	3	3	3	4	4	4	5	5	5	6	6	6	6	6	6	6	6	6	6	6	6
Heavy duty gasoline vehicle				1	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
Motorcycle (2&4 strokes)				1	2	2	2	2	2	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3
Rural Vehicle							1	2	2	2	2	2	2	3	3	3	3	4	4	4	5	5	5	6	6	6	6	6	6	6	6
Tractors, machines									1	1	2	2	2	2	3A	3A	3A	3A	3B	3B	3B	4	4	4	4	4	4	4	4	4	4
Train, inland water															3A	3A	3A	3A	3B	3B	3B	4	4	4	4	4	4	4	4	4	4

Figure IV.2. The implementation time of the vehicle emission standards: BAU[1]/PC[1]/ BAU[2]/PC[2] scenario. Numbers 1–6 represent the Euro I to Euro VI vehicle emission standards. Numbers in black represent standards released by the end of 2010, and numbers in red represent standards to be released in the future.

The BAU[2]/PC[2] scenario assumes the same assumptions on the implementation timeline of new standards as the BAU[1]/PC[1] scenario.

For emissions from solvent use, the BAU[1]/BAU[1] scenario assumes that new NMVOC emission standards (similar to or slightly less stringent than EU Directive 1999/13/EC and 2004/42/EC, depending on the specific industry) will be released and implemented in key provinces as of 2013 and in other provinces as of 2020. Afterwards, the emission standards will gradually become more stringent. The penetration of selected control measures assumed for key sources are summarized in Table IV.4.

Table IV.4. Penetrations of Major Control Technologies for VOC in China

Solvent use type	Control technology	2010	[1] 2020	[1] 2030	[2] 2030
Paint use in interior wall of buildings	No control (GB18582-2001)	0	0	0	0
	Decrease of solvent content--GB18582-2008	100	70	0	0
	Decrease of solvent content--2004/42/EC stage 1	0	30	80	0
	Decrease of solvent content--2004/42/EC stage 2	0	5	20	100
Paint use in external wall of buildings	No control (solvent-based paint)	78	70	50	0
	Substitution with water-based paint	22	30	50	100
Paint use in vehicle manufacturing	No control (water-based primer, solvent-based paint for other parts)	97	35	0	0
	Substitution with water-based paint	2	15	30	0
	Adsorption, incineration	1	40	65	0
	Substitution + adsorption, incineration	0	0	5	100
Paint use in vehicle refinishing	No control (solvent-based paint)	92.5	80	40	0
	Substitution with high solids or water-based paint	7.5	20	60	100
Paint use in wood coating	No control (solvent-based paint)	89	50	15	0
	Incineration	0	15	25	20
	Substitution with high solids paint	4	15	25	20
	Substitution with water-based or UV paint	7	20	35	60
Offset printing	No control (solvent-based ink)	90	60	15	0
	Substitution with water-based or UV ink	10	20	30	10
	Add-on control technology	0	20	55	90
Flexography and rotogravure printing (for packaging)	No control (solvent-based ink)	64	30	0	0
	Substitution with low solvent or water-based ink	35	40	30	0
	Add-on control technology	1	10	30	0
	Substitution + add-on control technology	0	20	40	100
Flexography and rotogravure printing (for publication)	No control (solvent-based ink)	85	62.5	5	0
	Substitution with low solvent or water-based ink	15	22.5	40	0
	Add-on control technology	0	15	50	0
	Substitution + add-on control technology	0	0	5	100
Screen printing	No control (solvent-based ink)	85	62.5	5	0
	Substitution with low solvent or water-based ink	15	22.5	40	0
	Add-on control technology	0	15	50	0
	Substitution + add-on control technology	0	0	5	100
Adhesive use in wood processing	No control	97.5	90	60	0
	Add-on control technology	2.5	10	40	100
Adhesive use in manufacturing of shoes	No control (solvent-based adhesive)	87	70	50	10
	Substitution with low solvent adhesive	13	30	50	90
	Add-on control technology	0	0	0	0

APPENDIX V: TABULATIONS OF PROVINCIAL-LEVEL ATTRIBUTABLE DEATHS AND DALYS FOR 2013 AND FUTURE SCENARIOS

Table V.1. Source Sector Contributions to Annual Average $PM_{2.5}$ Concentration (μg/m^3) by Province, 2013*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial	Power Plant	Domestic	Noncoal Industrial	Transport	Biomass	Open Burning	Solvent Use
China	54.3	21.9	9.4	5.2	2.4	5.6	8.2	8.0	4.1	(0.1)
Hong Kong	29.9	6.6	3.2	1.3	0.5	2.2	2.8	3.2	1.8	(0.0)
Anhui	61.7	24.4	10.9	7.1	1.2	6.2	10.3	9.5	4.7	(0.1)
Beijing	75.1	28.2	10.7	7.2	4.4	8.2	13.3	9.1	5.4	(0.0)
Chongqing	60.9	29.9	13.6	4.2	4.5	6.1	7.9	10.4	4.2	(0.0)
Fujian	29.5	11.5	5.1	2.8	0.9	3.0	3.5	4.2	2.4	(0.0)
Gansu	36.7	13.7	5.3	2.4	2.7	4.0	5.9	6.7	3.2	(0.0)
Guangdong	43.8	14.7	6.7	3.1	1.1	4.4	5.8	6.9	3.6	(0.0)
Guangxi	37.3	14.6	5.8	2.8	1.6	3.5	4.8	6.3	3.1	(0.0)
Guizhou	35.2	17.3	5.6	2.4	4.3	4.4	4.8	6.5	3.2	(0.0)
Hainan	19.5	5.6	2.1	1.9	0.5	1.4	1.6	2.3	1.1	(0.0)
Hebei	77.2	30.0	11.5	7.5	4.1	9.2	13.7	10.7	6.5	(0.1)
Heilongjiang	26.0	9.6	3.5	2.5	1.9	3.1	4.4	5.7	2.8	(0.0)
Henan	83.5	36.1	15.1	10.7	2.6	7.5	14.1	10.2	5.4	(0.1)
Hubei	59.4	26.8	11.9	5.6	3.0	6.0	8.9	8.7	4.4	(0.1)
Hunan	50.8	23.2	10.1	4.1	2.9	5.0	7.5	7.3	4.1	(0.1)
Inner Mongolia	30.4	12.4	3.4	2.8	4.1	4.1	4.5	5.0	3.4	(0.0)
Jiangsu	65.2	24.7	10.9	7.2	1.0	7.2	10.6	9.5	5.4	(0.1)

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Table V.1 continued*. Source Sector Contributions to Annual Average $PM_{2.5}$ Concentration (μg/m^3) by Province, 2013

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial	Power Plant	Domestic	Noncoal Industrial	Transport	Biomass	Open Burning	Solvent Use
Jiangxi	43.0	17.3	7.5	3.6	1.5	4.6	6.1	6.3	3.4	(0.0)
Jilin	34.1	13.0	5.4	3.5	1.8	4.0	5.4	6.2	3.0	(0.0)
Liaoning	44.3	15.8	6.7	4.4	1.4	4.8	6.5	6.0	3.0	(0.0)
Ningxia	37.0	13.8	5.2	2.4	2.9	4.2	5.2	6.6	3.5	(0.0)
Qinghai	29.0	8.8	3.3	2.0	1.8	2.9	4.5	3.5	1.9	(0.0)
Shaanxi	58.4	24.2	9.2	4.6	4.0	6.8	8.9	11.2	5.3	(0.0)
Shandong	71.2	28.9	12.3	8.9	1.8	6.1	12.1	8.6	4.4	(0.1)
Shanghai	56.1	18.8	8.0	5.0	0.9	6.6	7.5	7.0	4.7	(0.1)
Shanxi	57.4	23.3	7.9	5.0	4.4	8.0	9.1	9.9	6.1	(0.0)
Sichuan	64.9	31.0	17.1	4.3	3.0	6.5	8.6	11.0	3.9	(0.1)
Tianjin	69.5	25.5	10.8	6.9	2.7	7.2	13.1	8.0	4.5	(0.1)
Tibet	6.4	0.1	0.1	0.1	0.0	0.0	0.1	0.1	0.1	(0.0)
Xinjiang	57.5	14.6	3.3	8.0	2.4	3.4	4.1	6.4	3.7	(0.0)
Yunnan	22.9	6.6	3.1	1.1	1.5	2.5	2.5	3.5	1.7	(0.0)
Zhejiang	45.9	14.8	6.2	4.4	1.0	4.5	5.5	5.8	3.5	(0.0)

*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ concentration.

Table V.2. Source Sector Contributions to Annual Average PM2.5 Concentration (μg/m3) by Province, 2030 under Scenario BAU1.*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial	Power Plant	Domestic	Noncoal Industrial	Transport	Biomass	Open Burning	Solvent Use
China	49.8	22.1	9.3	5.1	2.3	3.2	4.7	3.2	3.2	(0.1)
Hong Kong	28.2	5.9	3.1	1.4	0.3	1.2	1.3	1.1	1.1	0.0
Anhui	55.8	25.2	11.0	7.1	1.7	4.2	6.6	4.3	4.3	(0.2)
Beijing	70.5	30.5	11.3	6.5	3.5	4.2	6.6	3.1	3.1	(0.1)
Chongqing	56.2	30.8	13.8	3.9	5.4	3.0	4.2	4.7	4.7	(0.1)
Fujian	28.3	10.9	5.3	2.7	0.7	1.9	1.7	1.7	1.7	(0.0)
Gansu	31.8	13.7	4.7	2.5	2.3	1.8	2.3	2.5	2.5	(0.1)
Guangdong	41.0	14.0	6.5	3.1	0.9	2.4	3.0	2.6	2.6	(0.0)
Guangxi	34.3	14.7	5.9	2.4	1.6	1.7	2.4	2.6	2.6	(0.1)
Guizhou	32.5	16.7	5.9	2.0	4.7	1.8	1.9	2.4	2.4	(0.0)
Hainan	17.2	5.4	2.1	1.2	0.4	0.8	0.8	0.9	0.9	(0.0)
Hebei	70.6	31.3	11.6	7.2	3.1	4.4	6.8	3.5	3.5	(0.2)
Heilongjiang	23.6	9.1	3.6	2.3	1.7	1.5	2.2	2.4	2.4	(0.0)
Henan	76.8	37.9	15.8	10.2	2.6	5.0	9.3	4.1	4.1	(0.3)
Hubei	52.8	26.1	10.6	5.4	3.5	3.7	5.2	3.6	3.6	(0.2)
Hunan	46.6	22.1	9.0	4.1	2.8	2.8	4.2	2.9	2.9	(0.1)
Inner Mongolia	26.9	11.6	3.6	2.7	2.9	1.6	1.2	1.3	1.3	(0.0)
Jiangsu	60.4	25.9	11.6	7.9	1.3	4.8	6.4	4.2	4.2	(0.2)
Jiangxi	39.6	16.0	7.0	3.6	1.3	2.8	3.5	2.6	2.6	(0.1)
Jilin	31.3	13.1	5.7	3.4	1.8	2.4	2.6	2.7	2.7	(0.1)
Liaoning	42.4	16.3	7.2	4.3	1.5	3.2	3.4	2.5	2.5	(0.1)
Ningxia	32.8	14.4	5.0	2.8	2.2	1.9	1.6	2.4	2.4	(0.0)
Qinghai	25.8	8.1	3.1	1.5	1.6	1.5	2.1	1.2	1.2	(0.0)
Shaanxi	50.7	25.0	8.6	5.3	3.5	2.9	3.8	4.3	4.3	(0.1)
Shandong	66.5	30.4	12.9	8.9	1.8	4.4	8.7	3.8	3.8	(0.3)

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Table V.2 continued. **Source Sector Contributions to Annual Average PM2.5 Concentration (µg/m3) by Province, 2030 under Scenario BAU1.***

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial	Power Plant	Domestic	Noncoal Industrial	Transport	Biomass	Open Burning	Solvent Use
Shanghai	54.5	20.8	9.9	6.4	0.8	4.6	3.6	2.5	2.5	(0.1)
Shanxi	49.6	23.4	7.5	4.9	3.0	3.0	3.2	2.9	2.9	(0.1)
Sichuan	59.2	29.7	14.6	4.3	4.1	3.4	6.0	5.3	5.3	(0.2)
Tianjin	65.5	27.9	11.5	6.7	2.5	4.2	7.4	2.9	2.9	(0.2)
Tibet	6.2	0.2	0.1	0.0	0.0	0.0	0.1	0.0	0.0	(0.0)
Xinjiang	46.7	12.4	4.2	3.3	1.5	1.7	1.3	2.3	2.3	(0.0)
Yunnan	19.9	6.6	3.0	0.9	1.2	1.3	1.0	1.2	1.2	(0.0)
Zhejiang	44.7	16.2	7.9	5.0	0.8	3.3	2.8	2.3	2.3	(0.1)
*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ concentration.										

Table V.3. Source Sector Contributions Annual Average $PM_{2.5}$ Concentration (µg/m^3) by Province, 2030 under Scenario BAU2.*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial	Power Plant	Domestic	Noncoal Industrial	Transport	Biomass	Open Burning	Solvent Use
China	38.1	21.0	8.2	4.2	2.2	3.1	2.2	2.9	2.9	(0.4)
Hong Kong	22.9	8.5	2.8	1.1	(0.1)	1.0	0.6	1.0	1.0	(0.1)
Anhui	41.1	22.7	9.5	5.8	1.3	3.9	2.9	4.0	4.0	(0.3)
Beijing	55.4	31.6	12.8	6.9	4.0	4.8	3.2	2.4	2.4	(0.8)
Chongqing	42.4	25.8	9.6	2.8	6.0	2.2	1.9	4.2	4.2	(0.5)
Fujian	22.1	11.4	3.9	2.0	(0.3)	1.6	0.9	1.3	1.3	(0.2)
Gansu	25.5	13.5	4.2	2.3	2.7	1.7	1.2	2.2	2.2	(0.2)
Guangdong	32.9	16.8	5.9	2.6	0.2	2.2	1.5	2.4	2.4	(0.3)
Guangxi	27.4	14.4	5.0	1.9	1.9	1.6	1.2	2.5	2.5	(0.2)
Guizhou	26.2	16.1	4.1	1.2	4.8	1.0	0.8	2.1	2.1	(0.3)
Hainan	13.7	5.1	1.7	1.0	0.2	0.8	0.4	0.9	0.9	(0.0)

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Table V.3 continued*. Source Sector Contributions Annual Average $PM_{2.5}$ Concentration (μg/m^3) by Province, 2030 under Scenario BAU2.

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial	Power Plant	Domestic	Noncoal Industrial	Transport	Biomass	Open Burning	Solvent Use
Hebei	54.8	32.0	12.7	7.2	3.5	4.8	3.1	2.9	2.9	(0.7)
Heilongjiang	18.6	8.3	3.4	1.3	1.4	1.8	0.9	2.1	2.1	0.0
Henan	57.0	35.0	14.9	8.9	3.2	5.6	4.1	4.0	4.0	(0.4)
Hubei	40.0	23.6	9.2	4.4	3.8	3.3	2.4	3.3	3.3	(0.5)
Hunan	36.2	20.2	8.2	3.2	3.1	2.7	2.0	2.7	2.7	(0.3)
Inner Mongolia	22.1	10.2	2.6	2.0	2.4	1.2	0.6	1.1	1.1	(0.2)
Jiangsu	44.6	24.4	10.0	6.6	0.4	4.5	2.9	3.7	3.7	(0.4)
Jiangxi	30.6	15.0	6.4	3.3	0.6	2.5	1.8	2.2	2.2	(0.3)
Jilin	24.2	12.2	5.3	2.0	1.3	2.7	1.1	2.4	2.4	(0.1)
Liaoning	32.6	16.8	7.1	3.0	0.7	3.8	1.7	2.2	2.2	(0.2)
Ningxia	26.6	13.8	4.1	2.6	2.4	1.7	0.9	2.1	2.1	(0.2)
Qinghai	21.3	7.7	2.7	1.2	1.6	1.4	1.0	1.0	1.0	(0.1)
Shaanxi	39.5	23.4	7.2	5.4	4.2	2.6	1.9	3.7	3.7	(0.5)
Shandong	48.9	28.7	12.7	7.2	2.0	5.5	3.8	3.7	3.7	(0.2)
Shanghai	41.1	21.2	7.8	5.5	(0.5)	3.9	2.0	2.1	2.1	(0.3)
Shanxi	39.7	23.8	7.5	5.4	3.2	2.7	1.6	2.4	2.4	(0.5)
Sichuan	43.6	25.0	10.7	3.5	4.7	2.7	2.8	5.1	5.1	(0.5)
Tianjin	50.2	27.6	12.9	6.5	2.9	5.4	3.5	2.5	2.5	(0.5)
Tibet	6.1	(0.1)	0.0	(0.0)	(0.0)	0.0	0.0	0.0	0.0	(0.0)
Xinjiang	39.9	11.4	2.5	1.2	0.1	1.0	0.5	1.8	1.8	(0.4)
Yunnan	16.6	5.1	1.8	0.6	1.2	0.4	0.3	1.0	1.0	(0.6)
Zhejiang	34.3	17.1	5.5	3.8	(0.5)	2.5	1.5	1.9	1.9	(0.3)

*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ concentration.

Table V.4. Source Sector Contributions to Annual Average $PM_{2.5}$ Concentration (µg/m^3) by Province, 2030 under Scenario PC1.*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial	Power Plant	Domestic	Noncoal Industrial	Transport	Biomass	Open Burning	Solvent Use
China	37.8	17.7	8.4	4.3	0.5	3.6	4.9	1.2	1.2	(0.2)
Hong Kong	22.9	6.4	2.7	0.8	(0.3)	1.2	1.4	0.3	0.3	(0.0)
Anhui	41.2	19.3	9.4	5.7	(0.1)	4.6	6.6	1.7	1.7	(0.3)
Beijing	56.2	26.0	11.6	7.7	1.0	5.5	7.7	1.2	1.2	(0.3)
Chongqing	40.1	22.4	12.4	2.3	2.5	3.2	4.4	1.7	1.7	(0.2)
Fujian	21.9	8.1	4.4	1.5	(0.7)	1.8	1.7	0.6	0.6	(0.1)
Gansu	24.6	10.1	4.4	2.5	0.9	2.1	2.7	1.0	1.0	(0.1)
Guangdong	32.2	13.2	6.1	2.0	(0.4)	2.6	3.2	0.8	0.8	(0.1)
Guangxi	26.3	12.1	5.8	1.6	0.7	2.0	2.7	0.8	0.8	(0.1)
Guizhou	24.0	12.7	5.2	0.9	2.3	1.7	1.8	0.8	0.8	(0.1)
Hainan	13.1	3.5	1.9	0.8	(0.1)	0.8	0.8	0.3	0.3	(0.0)
Hebei	55.6	26.7	11.7	7.8	0.8	5.6	7.8	1.4	1.4	(0.3)
Heilongjiang	17.9	6.9	3.1	2.0	0.3	1.7	2.0	1.0	1.0	(0.1)
Henan	58.3	29.6	14.2	9.6	0.7	6.3	9.7	1.8	1.8	(0.5)
Hubei	38.9	18.3	9.5	4.4	1.3	4.2	5.6	1.4	1.4	(0.3)
Hunan	35.1	17.0	8.4	3.2	1.0	3.3	4.6	1.1	1.1	(0.3)
Inner Mongolia	21.1	8.9	3.0	2.1	0.9	1.5	1.3	0.5	0.5	(0.0)
Jiangsu	45.4	21.5	10.1	6.1	(0.7)	5.1	6.6	1.6	1.6	(0.3)
Jiangxi	30.5	12.6	6.6	2.7	(0.3)	3.0	4.0	0.9	0.9	(0.2)
Jilin	23.6	11.0	4.9	3.0	0.1	2.7	2.5	1.1	1.1	(0.1)
Liaoning	32.8	15.4	6.8	3.7	(0.4)	3.8	3.7	1.1	1.1	(0.1)
Ningxia	25.7	11.9	4.5	2.8	0.8	2.1	2.1	1.0	1.0	(0.1)
Qinghai	21.0	4.7	2.7	1.3	0.4	1.7	2.2	0.5	0.5	(0.1)
Shaanxi	37.9	20.2	8.1	5.4	1.5	3.4	4.5	1.7	1.7	(0.2)
Shandong	50.3	25.7	11.8	8.1	0.0	5.6	8.8	1.7	1.7	(0.5)

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Table V.4 continued*. Source Sector Contributions to Annual Average $PM_{2.5}$ Concentration ($\mu g/m^3$) by Province, 2030 under Scenario PC1.

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial	Power Plant	Domestic	Noncoal Industrial	Transport	Biomass	Open Burning	Solvent Use
Shanghai	42.6	18.8	8.6	4.6	(1.3)	4.4	4.3	0.9	0.9	(0.2)
Shanxi	39.2	19.7	7.5	5.6	1.0	3.5	4.1	1.2	1.2	(0.2)
Sichuan	41.6	21.8	12.6	2.9	2.0	3.7	5.8	2.0	2.0	(0.3)
Tianjin	51.3	23.5	11.4	7.3	0.4	5.6	8.2	1.2	1.2	(0.4)
Tibet	6.1	(0.1)	0.0	(0.0)	(0.0)	0.0	0.1	0.0	0.0	(0.0)
Xinjiang	39.1	8.5	3.0	1.6	(0.7)	1.3	1.1	0.9	0.9	(0.0)
Yunnan	16.3	4.1	2.4	0.4	0.7	1.2	0.8	0.4	0.4	(0.0)
Zhejiang	34.9	14.1	6.3	3.1	(1.1)	2.9	3.1	0.8	0.8	(0.1)

*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ concentration.

Table V.5. Source Sector Contributions to Annual Average $PM_{2.5}$ Concentration ($\mu g/m^3$) by Province, 2030 under Scenario PC2.*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
China	27.2	13.6	7.0	2.9	0.4	3.8	2.0	1.2	1.2	(0.2)
Hong Kong	18.0	6.9	3.0	1.1	(0.4)	1.6	0.8	0.3	0.3	(0.0)
Anhui	28.4	13.4	7.7	3.8	(0.5)	4.5	2.5	1.6	1.6	(0.2)
Beijing	40.4	20.4	11.0	6.0	1.5	6.6	3.1	1.3	1.3	(0.2)
Chongqing	28.4	16.9	8.4	1.3	2.8	3.2	1.9	1.6	1.6	(0.2)
Fujian	16.9	7.2	3.9	1.4	(1.0)	2.2	1.0	0.5	0.5	(0.1)
Gansu	18.4	8.3	3.6	2.0	1.0	2.2	1.2	0.9	0.9	(0.1)
Guangdong	24.7	12.5	5.9	2.0	(0.7)	3.3	1.6	0.8	0.8	(0.1)
Guangxi	19.7	9.3	4.6	1.1	0.9	2.3	1.3	0.8	0.8	(0.1)
Guizhou	18.7	10.9	4.1	0.6	3.0	1.7	0.9	0.8	0.8	(0.1)

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Table V.5 continued*. Source Sector Contributions to Annual Average $PM_{2.5}$ Concentration (μg/m^3) by Province, 2030 under Scenario PC2.

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
Hainan	10.0	3.3	1.5	0.6	(0.2)	0.9	0.4	0.3	0.3	(0.0)
Hebei	39.6	21.0	10.9	6.0	1.1	6.4	3.0	1.5	1.5	(0.3)
Heilongjiang	13.7	4.3	2.8	0.9	0.2	1.9	0.7	0.9	0.9	(0.0)
Henan	39.9	21.6	11.5	6.1	0.6	6.3	3.5	1.8	1.8	(0.3)
Hubei	27.3	14.9	7.5	2.7	1.2	4.1	2.2	1.3	1.4	(0.2)
Hunan	25.4	13.8	7.1	2.0	1.0	3.7	2.0	1.1	1.1	(0.2)
Inner Mongolia	17.1	7.0	2.6	1.8	0.8	1.6	0.6	0.5	0.5	(0.0)
Jiangsu	31.8	15.0	8.5	4.2	(1.4)	5.1	2.6	1.5	1.5	(0.2)
Jiangxi	22.2	10.4	5.9	2.4	(0.5)	3.4	1.9	0.9	0.9	(0.2)
Jilin	17.5	6.7	4.3	1.3	(0.2)	2.8	0.9	1.0	1.0	(0.1)
Liaoning	23.8	10.3	6.2	1.9	(0.9)	4.2	1.5	1.0	1.0	(0.1)
Ningxia	19.7	9.3	3.8	2.5	0.9	2.2	1.0	0.9	0.9	(0.1)
Qinghai	16.8	4.6	2.1	0.7	0.3	1.5	0.9	0.4	0.4	(0.0)
Shaanxi	27.2	15.5	6.3	4.4	1.9	3.5	1.9	1.6	1.6	(0.2)
Shandong	34.0	17.4	9.8	4.6	(0.3)	5.7	3.0	1.6	1.6	(0.3)
Shanghai	31.1	14.4	7.6	3.8	(1.9)	4.8	2.1	0.9	0.9	(0.2)
Shanxi	29.1	16.4	7.1	4.9	1.3	4.0	1.8	1.1	1.1	(0.2)
Sichuan	28.4	16.2	8.3	1.9	2.0	3.4	2.3	1.8	1.8	(0.2)
Tianjin	35.9	17.3	10.4	4.8	0.3	6.4	3.0	1.3	1.3	(0.3)
Tibet	6.0	0.1	0.0	(0.0)	(0.0)	0.0	0.0	0.0	0.0	0.0
Xinjiang	34.3	7.0	2.3	0.2	(1.3)	1.5	0.5	0.8	0.8	(0.0)
Yunnan	13.6	3.7	1.5	0.3	0.7	0.8	0.4	0.3	0.3	(0.0)
Zhejiang	26.4	11.9	5.8	2.8	(1.6)	3.4	1.7	0.8	0.8	(0.2)

*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ concentration.

Table V.6. Mean Deaths Attributable to $PM_{2.5}$ Exposure by Source and Province, 2013.*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
China	915,900	366,160	155,490	86,530	41,020	94,880	137,400	136,470	70,230	(870)
Hong Kong	4,780	1,060	500	210	70	350	450	510	290	0
Anhui	47,210	18,650	8,350	5,410	920	4,780	7,810	7,260	3,590	(70)
Beijing	13,050	4,890	1,850	1,250	770	1,430	2,300	1,580	950	0
Chongqing	25,460	12,480	5,660	1,760	1,900	2,560	3,290	4,370	1,780	(20)
Fujian	16,640	6,480	2,860	1,570	530	1,700	1,960	2,360	1,320	(10)
Gansu	11,600	4,330	1,670	740	860	1,270	1,850	2,110	1,000	(10)
Guangdong	47,660	16,090	7,250	3,450	1,250	4,820	6,300	7,500	3,910	(30)
Guangxi	30,840	12,020	4,800	2,350	1,320	2,920	3,920	5,230	2,530	(20)
Guizhou	23,760	11,670	3,780	1,640	2,900	2,970	3,220	4,390	2,190	(10)
Hainan	2,830	810	300	270	70	190	230	340	160	0
Hebei	63,680	24,630	9,450	6,170	3,420	7,550	11,290	8,780	5,280	(60)
Heilongjiang	25,930	9,510	3,420	2,510	1,910	3,090	4,390	5,630	2,800	(20)
Henan	79,070	34,180	14,380	10,080	2,450	7,150	13,330	9,740	5,150	(110)
Hubei	41,340	18,620	8,270	3,860	2,100	4,150	6,180	6,070	3,040	(50)
Hunan	44,940	20,460	8,890	3,570	2,570	4,390	6,610	6,440	3,590	(50)
Inner Mongolia	15,050	6,110	1,720	1,380	1,970	2,010	2,210	2,460	1,680	0
Jiangsu	53,160	20,090	8,860	5,880	840	5,850	8,620	7,770	4,370	(70)
Jiangxi	22,440	9,000	3,930	1,890	780	2,400	3,200	3,310	1,790	(20)
Jilin	19,040	7,200	2,980	1,930	990	2,190	3,010	3,460	1,670	(10)
Liaoning	40,710	14,550	6,150	4,040	1,270	4,420	5,950	5,500	2,780	(30)
Ningxia	2,700	1,010	380	170	210	310	380	480	250	0
Qinghai	2,510	760	290	180	160	250	380	300	160	0
Shaanxi	23,910	9,910	3,770	1,860	1,650	2,760	3,600	4,580	2,180	(20)

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Table V.6 continued*. Mean Deaths Attributable to $PM_{2.5}$ Exposure by Source and Province, 2013.

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
Shandong	88,550	35,870	15,160	10,920	2,240	7,650	15,000	10,770	5,530	(130)
Shanghai	11,680	3,900	1,670	1,050	190	1,380	1,570	1,450	980	(10)
Shanxi	22,270	9,040	3,050	1,920	1,780	3,170	3,550	3,850	2,420	(10)
Sichuan	69,300	33,030	18,220	4,610	3,230	6,920	9,190	11,810	4,150	(60)
Tianjin	8,790	3,230	1,360	870	340	910	1,650	1,010	570	(10)
Tibet	120	0	0	0	0	0	0	0	0	0
Xinjiang	13,650	3,350	740	1,780	570	850	970	1,510	880	0
Yunnan	18,110	5,150	2,370	840	1,180	2,000	1,940	2,720	1,320	0
Zhejiang	25,110	8,090	3,410	2,390	580	2,490	3,030	3,210	1,920	(20)

*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ and attributable deaths.

Table V.7. Mean Deaths Attributable to $PM_{2.5}$ Exposure by Source and Province: 2030 under Scenario BAU1.*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
China	1,267,390	557,760	234,810	127,370	59,460	81,720	116,720	83,080	83,080	(3,040)
Hong Kong	6,750	1,410	730	330	70	280	310	250	250	0
Anhui	56,830	25,530	11,200	7,210	1,730	4,310	6,670	4,420	4,420	(190)
Beijing	13,640	5,910	2,180	1,260	670	810	1,270	600	600	(20)
Chongqing	30,170	16,510	7,370	2,120	2,900	1,610	2,250	2,510	2,510	(70)
Fujian	28,750	11,140	5,370	2,740	680	1,980	1,750	1,700	1,700	(20)
Gansu	15,140	6,480	2,230	1,170	1,110	870	1,110	1,210	1,210	(30)
Guangdong	63,390	21,660	10,110	4,830	1,330	3,720	4,590	4,060	4,060	(30)

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Table V.7 continued*. Mean Deaths Attributable to $PM_{2.5}$ Exposure by Source and Province: 2030 under Scenario BAU1.

Location	All Ambient $PM_{2.5}$	Coal related: Total Coal	Coal related: Industrial Coal	Coal related: Power Plant	Coal related: Domestic Coal	Noncoal related: Noncoal Industrial	Noncoal related: Transport	Noncoal related: Domestic Biomass Burning	Noncoal related: Open Burning	Noncoal related: Solvent Use
Guangxi	59,850	25,660	10,200	4,180	2,870	2,990	4,180	4,530	4,530	(90)
Guizhou	31,140	15,960	5,630	1,900	4,530	1,730	1,840	2,290	2,290	(40)
Hainan	3,150	970	380	220	70	150	140	160	160	0
Hebei	96,160	42,430	15,660	9,670	4,220	6,010	9,190	4,720	4,710	(220)
Heilongjiang	28,430	10,810	4,300	2,700	2,000	1,800	2,620	2,860	2,860	(40)
Henan	102,830	50,760	21,250	13,700	3,570	6,790	12,460	5,600	5,600	(430)
Hubei	59,640	29,410	11,970	6,030	4,000	4,130	5,800	4,090	4,090	(190)
Hunan	71,880	33,990	13,840	6,270	4,380	4,350	6,370	4,550	4,550	(180)
Inner Mongolia	16,540	7,090	2,220	1,630	1,750	960	740	810	810	(10)
Jiangsu	87,980	37,790	16,960	11,470	1,920	7,010	9,360	6,050	6,050	(250)
Jiangxi	28,890	11,660	5,120	2,630	910	2,050	2,560	1,880	1,880	(50)
Jilin	19,810	8,270	3,610	2,170	1,140	1,510	1,630	1,670	1,670	(40)
Liaoning	54,010	20,700	9,180	5,460	1,950	4,080	4,250	3,170	3,170	(80)
Ningxia	3,730	1,640	570	310	250	210	180	270	270	0
Qinghai	4,130	1,290	490	230	250	230	340	190	190	(10)
Shaanxi	26,400	13,000	4,470	2,730	1,820	1,510	1,940	2,240	2,240	(60)
Shandong	138,560	62,990	26,680	18,370	3,730	9,180	17,930	7,890	7,890	(550)
Shanghai	16,000	6,100	2,900	1,870	250	1,340	1,070	720	720	(20)
Shanxi	23,680	11,170	3,520	2,310	1,470	1,440	1,480	1,380	1,380	(40)
Sichuan	101,700	50,980	25,020	7,410	7,120	5,840	10,260	9,160	9,160	(300)
Tianjin	8,720	3,720	1,530	890	330	560	980	390	390	(20)
Tibet	120	0	0	0	0	0	0	0	0	0
Xinjiang	15,980	4,170	1,380	1,090	510	560	410	750	750	(10)
Yunnan	22,420	7,340	3,300	970	1,340	1,430	1,080	1,360	1,360	(10)
Zhejiang	30,960	11,220	5,470	3,500	580	2,290	1,960	1,610	1,610	(40)

*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ and attributable deaths.

Table V.8. Mean Deaths Attributable to $PM_{2.5}$ Exposure by Source and Province: 2030 under Scenario BAU2.*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
China	1,144,810	626,600	245,370	124,870	66,180	93,940	64,250	89,110	89,110	(10,680)
Hong Kong	6,070	2,240	750	300	(20)	270	170	270	270	(30)
Anhui	50,830	28,070	11,750	7,210	1,580	4,860	3,590	4,980	4,980	(420)
Beijing	12,590	7,180	2,920	1,580	920	1,100	710	560	560	(190)
Chongqing	26,940	16,380	6,090	1,750	3,830	1,360	1,210	2,650	2,650	(340)
Fujian	25,380	13,010	4,500	2,240	(300)	1,780	1,020	1,550	1,550	(230)
Gansu	13,750	7,280	2,270	1,260	1,450	910	630	1,190	1,190	(130)
Guangdong	58,130	29,620	10,470	4,630	380	3,880	2,570	4,320	4,320	(580)
Guangxi	54,330	28,500	9,970	3,820	3,670	3,180	2,450	4,960	4,960	(490)
Guizhou	27,980	17,150	4,390	1,240	5,110	1,050	860	2,270	2,270	(300)
Hainan	2,700	990	330	200	40	150	90	170	170	(10)
Hebei	88,840	51,630	20,550	11,560	5,760	7,820	5,050	4,730	4,730	(1,200)
Heilongjiang	25,270	11,160	4,510	1,750	1,850	2,450	1,240	2,880	2,880	40
Henan	92,970	56,880	24,200	14,410	5,320	9,120	6,690	6,610	6,610	(640)
Hubei	53,650	31,680	12,280	5,880	5,080	4,360	3,210	4,420	4,420	(640)
Hunan	65,700	36,570	14,710	5,710	5,560	4,950	3,620	4,950	4,950	(620)
Inner Mongolia	15,050	7,010	1,800	1,390	1,630	800	410	740	740	(130)
Jiangsu	79,090	43,350	17,810	11,640	670	7,960	5,180	6,610	6,610	(710)
Jiangxi	25,880	12,700	5,390	2,810	520	2,120	1,530	1,890	1,890	(280)
Jilin	17,820	8,970	3,840	1,470	950	1,960	830	1,730	1,730	(50)
Liaoning	48,850	25,140	10,670	4,530	1,040	5,690	2,500	3,220	3,220	(260)
Ningxia	3,430	1,790	530	340	310	220	120	270	270	(20)
Qinghai	3,780	1,380	490	210	290	250	180	180	180	(30)
Shaanxi	24,230	14,340	4,420	3,240	2,580	1,550	1,130	2,260	2,260	(300)

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Table V.8 continued*. Mean Deaths Attributable to $PM_{2.5}$ Exposure by Source and Province: 2030 under Scenario BAU2.

		Coal related				Noncoal related				
Location	All Ambient $PM_{2.5}$	Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
Shandong	124,970	73,110	32,390	18,380	4,970	14,120	9,740	9,460	9,460	(490)
Shanghai	14,410	7,430	2,730	1,910	(180)	1,380	700	730	730	(120)
Shanxi	21,760	13,040	4,040	2,900	1,760	1,440	860	1,310	1,310	(270)
Sichuan	89,710	51,430	21,930	7,230	9,770	5,510	5,660	10,420	10,420	(990)
Tianjin	8,000	4,390	2,050	1,030	460	860	550	400	400	(80)
Tibet	110	0	0	0	0	0	0	0	0	0
Xinjiang	15,070	4,390	1,000	530	120	380	180	660	660	(140)
Yunnan	20,020	6,100	2,220	740	1,450	460	410	1,240	1,240	(780)
Zhejiang	27,500	13,700	4,400	2,990	(390)	2,020	1,170	1,500	1,500	(240)
*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ and attributable deaths.										

Table V.9. Mean Deaths Attributable to $PM_{2.5}$ Exposure by Source and Province: 2030 under Scenario PC1.*

		Coal related				Noncoal related				
Location	All Ambient $PM_{2.5}$	Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
China	1,139,230	531,770	252,710	125,920	14,960	109,920	146,340	37,890	37,890	(6,700)
Hong Kong	6,070	1,700	720	220	(70)	310	370	80	80	0
Anhui	50,860	23,790	11,610	6,970	(100)	5,630	8,120	2,060	2,060	(380)
Beijing	12,650	5,850	2,610	1,720	240	1,240	1,740	280	280	(60)
Chongqing	26,310	14,670	8,060	1,530	1,650	2,090	2,870	1,090	1,090	(150)
Fujian	25,260	9,270	5,110	1,690	(750)	2,130	1,980	660	660	(60)
Gansu	13,520	5,550	2,420	1,380	520	1,160	1,500	560	560	(50)
Guangdong	57,650	23,520	10,920	3,670	(780)	4,620	5,700	1,420	1,420	(120)

Table continues on next page

Table V.9 continued*. Mean Deaths Attributable to $PM_{2.5}$ Exposure by Source and Province: 2030 under Scenario PC1.

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
Guangxi	53,330	24,490	11,730	3,290	1,360	4,100	5,480	1,650	1,650	(240)
Guizhou	26,760	14,070	5,810	960	2,510	1,890	1,990	890	890	(70)
Hainan	2,610	690	380	160	(20)	160	160	50	50	(10)
Hebei	89,220	42,700	18,640	12,520	1,240	8,970	12,450	2,290	2,290	(550)
Heilongjiang	24,710	9,490	4,270	2,750	460	2,380	2,710	1,400	1,400	(80)
Henan	93,650	47,500	22,770	15,310	1,120	10,100	15,550	2,910	2,910	(840)
Hubei	53,010	24,950	12,950	5,940	1,790	5,680	7,560	1,920	1,920	(420)
Hunan	64,950	31,480	15,560	5,890	1,830	6,150	8,410	2,060	2,060	(470)
Inner Mongolia	14,710	6,270	2,110	1,500	600	1,060	910	370	370	(20)
Jiangsu	79,580	37,760	17,690	10,680	(1,220)	8,900	11,550	2,810	2,810	(520)
Jiangxi	25,850	10,700	5,570	2,260	(270)	2,550	3,360	780	780	(140)
Jilin	17,610	8,130	3,600	2,210	100	1,980	1,870	850	850	(70)
Liaoning	49,000	23,050	10,110	5,460	(640)	5,700	5,570	1,580	1,580	(200)
Ningxia	3,380	1,580	600	370	100	270	280	130	130	(10)
Qinghai	3,750	850	490	230	80	310	400	90	90	(10)
Shaanxi	23,870	12,710	5,100	3,350	950	2,100	2,810	1,060	1,060	(140)
Shandong	126,120	64,490	29,560	20,290	10	14,140	21,970	4,220	4,220	(1,150)
Shanghai	14,610	6,460	2,930	1,560	(440)	1,510	1,470	320	320	(60)
Shanxi	21,640	10,850	4,090	3,030	560	1,930	2,190	640	640	(90)
Sichuan	87,910	45,950	26,520	6,210	4,290	7,750	12,280	4,150	4,150	(610)
Tianjin	8,060	3,690	1,790	1,140	60	880	1,280	190	190	(60)
Tibet	110	0	0	0	0	0	0	0	0	0
Xinjiang	14,950	3,440	1,120	710	(190)	500	410	340	340	(10)
Yunnan	19,790	4,940	2,890	500	850	1,450	980	420	420	(20)
Zhejiang	27,730	11,180	4,970	2,410	(890)	2,290	2,410	650	650	(90)

*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ and attributable deaths.

Table V.10. Mean Deaths Attributable to $PM_{2.5}$ Exposure by Source and Province: 2030 under Scenario PC2.*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
China	990,110	490,020	254,220	104,100	13,660	140,790	72,140	44,050	44,040	(6,170)
Hong Kong	5,290	2,020	890	320	(110)	480	3,850	100	100	(10)
Anhui	43,750	20,670	11,810	5,770	(770)	6,860	850	2,410	2,410	(340)
Beijing	11,240	5,680	3,040	1,670	410	1,820	1,480	350	350	(70)
Chongqing	22,510	13,360	6,670	1,010	2,190	2,510	1,250	1,250	1,250	(120)
Fujian	21,610	9,160	5,000	1,830	(1,250)	2,820	770	650	650	(90)
Gansu	11,680	5,310	2,290	1,270	680	1,380	3,290	580	580	(50)
Guangdong	51,360	25,630	12,170	4,110	(1,400)	6,760	3,050	1,660	1,660	(190)
Guangxi	46,160	21,850	10,750	2,540	2,040	5,380	1,120	1,940	1,940	(230)
Guizhou	23,140	13,490	5,100	740	3,760	2,080	90	950	950	(70)
Hainan	1,890	640	300	120	(30)	180	6,020	50	50	(10)
Hebei	79,520	41,890	21,880	11,930	2,180	12,860	1,090	2,930	2,930	(570)
Heilongjiang	20,710	6,510	4,220	1,310	230	2,850	7,090	1,330	1,330	(60)
Henan	81,410	43,900	23,420	12,390	1,130	12,760	3,720	3,720	3,730	(680)
Hubei	45,610	24,910	12,500	4,570	1,970	6,860	4,480	2,250	2,250	(340)
Hunan	57,080	31,030	15,880	4,510	2,170	8,330	500	2,480	2,480	(420)
Inner Mongolia	13,050	5,450	2,060	1,380	640	1,250	5,680	360	360	(30)
Jiangsu	69,380	32,850	18,570	9,270	(2,950)	11,210	1,890	3,220	3,220	(520)
Jiangxi	22,140	10,420	5,930	2,380	(510)	3,430	810	900	900	(160)
Jilin	15,130	5,770	3,720	1,110	(170)	2,450	2,670	880	880	(60)
Liaoning	42,680	18,450	11,190	3,390	(1,580)	7,470	160	1,760	1,760	(200)
Ningxia	3,000	1,420	580	370	140	340	180	140	140	(10)
Qinghai	3,340	940	430	150	60	310	1,490	80	80	(10)

Table continues on next page

Table V.10 continued*. Mean Deaths Attributable to $PM_{2.5}$ Exposure by Source and Province: 2030 under Scenario PC2.

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
Shaanxi	20,990	11,980	4,880	3,330	1,470	2,670	9,590	1,220	1,220	(130)
Shandong	109,140	55,770	31,370	14,730	(1,180)	18,420	880	5,200	5,200	(960)
Shanghai	12,880	5,980	3,150	1,560	(800)	1,980	1,170	370	370	(90)
Shanxi	19,140	10,810	4,610	3,240	890	2,580	5,930	730	730	(100)
Sichuan	73,670	42,070	21,510	5,050	5,320	8,760	590	4,790	4,790	(440)
Tianjin	7,100	3,420	2,060	950	70	1,260	0	260	260	(60)
Tibet	110	0	0	0	0	0	210	0	0	0
Xinjiang	14,130	3,090	1,010	250	(410)	630	480	310	310	(10)
Yunnan	17,090	4,640	1,890	340	940	980	1,520	430	430	(10)
Zhejiang	24,180	10,890	5,320	2,520	(1,480)	3,120	220	710	710	(150)

*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ and attributable deaths.

Table V.11. Age-Standardized DALY Rate (DALYs/100,000) Attributable to $PM_{2.5}$ by Source and Province: 2013.*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
China	1,264	505	213	119	58	131	189	189	97	(1)
Hong Kong	634	140	67	28	10	47	60	67	39	(0)
Anhui	1,307	516	231	150	26	132	216	201	99	(2)
Beijing	1,098	412	156	105	65	120	194	133	80	(0)
Chongqing	1,362	667	303	94	101	137	176	234	95	(1)
Fujian	878	342	151	83	28	90	104	125	70	(1)
Gansu	1,055	394	152	68	79	115	168	192	91	(1)
Guangdong	995	336	151	72	26	101	132	157	82	(1)
Guangxi	1,276	497	199	97	55	121	162	217	105	(1)
Guizhou	1,536	754	244	106	188	192	208	284	142	(1)
Hainan	619	177	67	58	15	42	51	74	35	(0)
Hebei	1,773	686	263	172	95	210	314	244	147	(2)
Heilongjiang	1,274	467	168	123	94	152	216	276	138	(1)
Henan	1,727	747	314	220	54	156	291	213	113	(2)
Hubei	1,330	599	266	124	68	134	199	195	98	(2)
Hunan	1,181	538	234	94	68	115	174	169	94	(1)
Inner Mongolia	1,299	527	148	119	170	174	191	212	145	(0)
Jiangsu	960	363	160	106	15	106	156	140	79	(1)
Jiangxi	1,083	435	190	91	38	116	154	160	86	(1)
Jilin	1,286	487	202	130	67	148	204	234	113	(1)
Liaoning	1,434	513	217	142	45	156	210	194	98	(1)
Ningxia	1,128	420	158	72	89	128	159	201	106	(0)
Qinghai	1,276	385	145	89	79	127	195	153	83	(1)
Shaanxi	1,311	543	206	102	90	151	198	251	119	(1)

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Table V.11 continued. Age-Standardized DALY Rate (DALYs/100,000) Attributable to $PM_{2.5}$ by Source and Province: 2013.*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
Shandong	1,506	610	258	186	38	130	255	183	94	(2)
Shanghai	622	208	89	56	10	74	83	77	52	(1)
Shanxi	1,308	531	179	113	104	186	208	226	142	(1)
Sichuan	1,410	672	371	94	66	141	187	240	84	(1)
Tianjin	1,147	422	178	114	45	119	216	132	75	(2)
Tibet	149	3	1	2	0	1	2	2	1	(0)
Xinjiang	1,796	441	97	234	76	112	127	198	116	(0)
Yunnan	958	272	125	44	63	106	103	144	70	(0)
Zhejiang	689	222	93	66	16	68	83	88	53	(1)

*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ and attributable DALY rate.

Table V.12. Age-Standardized DALY Rate (DALYs/100,000) Attributable to $PM_{2.5}$ by Source and Province: 2030 under Scenario BAU1.*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
China	934	411	172	93	45	60	86	61	61	(2)
Hong Kong	502	105	55	25	5	21	23	19	19	0
Anhui	809	364	159	103	25	61	95	63	63	(3)
Beijing	600	260	96	56	30	36	56	26	26	(1)
Chongqing	849	465	208	60	82	45	63	71	71	(2)
Fujian	819	317	153	78	19	56	50	48	48	(1)
Gansu	762	326	112	59	56	44	56	61	61	(1)
Guangdong	688	235	110	52	14	40	50	44	44	(0)

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Table V.12 continued*. Age-Standardized DALY Rate (DALYs/100,000) Attributable to $PM_{2.5}$ by Source and Province: 2030 under Scenario BAU1.

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
Guangxi	1409	604	240	98	68	70	98	107	107	(2)
Guizhou	1064	546	192	65	155	59	63	78	78	(1)
Hainan	344	106	41	24	8	16	16	18	18	(0)
Hebei	1522	672	248	153	67	95	146	75	75	(3)
Heilongjiang	771	293	117	73	54	49	71	77	77	(1)
Henan	1217	601	252	162	42	80	147	66	66	(5)
Hubei	1057	521	212	107	71	73	103	72	72	(3)
Hunan	934	442	180	82	57	57	83	59	59	(2)
Inner Mongolia	814	349	109	80	86	47	36	40	40	(1)
Jiangsu	775	333	149	101	17	62	83	53	53	(2)
Jiangxi	740	299	131	67	23	53	66	48	48	(1)
Jilin	727	304	133	80	42	56	60	61	61	(1)
Liaoning	1048	402	178	106	38	79	83	62	62	(2)
Ningxia	876	386	133	74	60	50	43	63	63	(1)
Qinghai	1161	363	137	66	71	66	95	52	52	(2)
Shaanxi	789	389	134	82	54	45	58	67	67	(2)
Shandong	1318	599	254	175	36	87	171	75	75	(5)
Shanghai	412	157	75	48	6	35	27	19	18	(1)
Shanxi	758	358	113	74	47	46	48	44	44	(1)
Sichuan	1093	548	269	80	77	63	110	98	98	(3)
Tianjin	567	241	99	58	22	36	64	25	25	(2)
Tibet	77	2	1	0	0	0	1	0	0	(0)
Xinjiang	1206	315	104	82	39	43	31	57	57	(0)
Yunnan	663	217	98	29	40	42	32	40	40	(0)
Zhejiang	415	151	73	47	8	31	26	22	22	(0)

*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ and attributable DALY rate.

Table V.13. Age-Standardized DALY Rate (DALYs/100,000) Attributable to $PM_{2.5}$ by Source and Province: 2030 under Scenario BAU2.*

		Coal related				Noncoal related				
Location	All Ambient $PM_{2.5}$	Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
China	843	460	180	91	50	69	47	65	65	(8)
Hong Kong	450	166	55	22	(2)	20	13	20	20	(2)
Anhui	721	398	167	102	22	69	51	71	71	(6)
Beijing	551	315	128	69	40	48	31	24	24	(8)
Chongqing	757	460	171	49	108	38	34	74	74	(10)
Fujian	720	369	128	64	(9)	51	29	44	44	(7)
Gansu	691	366	114	63	73	46	32	60	60	(7)
Guangdong	628	320	113	50	4	42	28	47	47	(6)
Guangxi	1279	671	235	90	87	75	58	117	117	(12)
Guizhou	958	587	150	43	175	36	29	78	78	(10)
Hainan	294	108	36	22	5	17	9	19	19	(1)
Hebei	1404	816	325	183	91	124	80	75	75	(19)
Heilongjiang	684	302	122	47	50	66	34	78	78	1
Henan	1100	673	286	171	63	108	79	78	78	(8)
Hubei	950	561	217	104	90	77	57	78	78	(11)
Hunan	850	473	191	74	72	64	47	64	64	(8)
Inner Mongolia	740	345	88	68	80	40	20	36	36	(6)
Jiangsu	694	380	156	102	6	70	46	58	58	(6)
Jiangxi	661	324	138	72	13	54	39	48	48	(7)
Jilin	652	328	141	54	35	72	30	64	64	(2)
Liaoning	947	487	207	88	20	110	48	62	62	(5)
Ningxia	805	420	124	80	74	51	28	62	62	(5)
Qinghai	1060	387	137	59	81	69	51	51	51	(7)

Table continues on next page

Table V.13 continued*. Age-Standardized DALY Rate (DALYs/100,000) Attributable to $PM_{2.5}$ by Source and Province: 2030 under Scenario BAU2.

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
Shaanxi	722	427	132	97	77	46	34	67	67	(9)
Shandong	1186	694	307	174	47	134	93	90	90	(5)
Shanghai	367	189	70	49	(5)	35	18	19	19	(3)
Shanxi	696	417	129	93	56	46	28	42	42	(9)
Sichuan	965	553	236	78	105	59	61	112	112	(11)
Tianjin	517	284	133	67	30	56	36	26	26	(5)
Tibet	73	1	0	0	0	0	0	0	0	(0)
Xinjiang	1137	331	76	40	9	29	13	50	50	(10)
Yunnan	592	180	66	22	43	14	12	37	37	(23)
Zhejiang	367	183	59	40	(5)	27	16	20	20	(3)

*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ and attributable DALY rate.

Table V.14. Age-Standardized DALY Rate (DALYs/100,000) Attributable to $PM_{2.5}$ by Source and province: 2030 under Scenario PC1.*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
China	838	390	185	93	12	81	107	28	28	(5)
Hong Kong	450	126	54	17	(5)	23	28	6	6	(0)
Anhui	721	337	165	99	(1)	80	115	29	29	(5)
Beijing	554	256	114	75	10	54	76	12	12	(3)
Chongqing	739	412	227	43	46	59	81	31	31	(4)
Fujian	717	263	145	48	(21)	60	56	19	19	(2)
Gansu	680	279	122	69	26	58	76	28	28	(3)

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Table V.14 continued*. Age-Standardized DALY Rate (DALYs/100,000) Attributable to $PM_{2.5}$ by Source and province: 2030 under Scenario PC1.

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
Guangdong	623	254	118	40	(8)	50	62	15	15	(1)
Guangxi	1255	576	276	77	32	97	129	39	39	(6)
Guizhou	916	482	199	33	86	65	68	30	30	(2)
Hainan	283	75	41	17	(2)	17	18	6	6	(1)
Hebei	1410	675	295	198	20	142	197	36	36	(9)
Heilongjiang	668	257	115	75	12	64	73	38	38	(2)
Henan	1108	562	269	181	13	119	184	34	34	(10)
Hubei	938	442	229	105	32	101	134	34	34	(7)
Hunan	840	407	201	76	24	80	109	27	27	(6)
Inner Mongolia	723	308	104	74	29	52	45	18	18	(1)
Jiangsu	698	331	155	94	(11)	78	101	25	25	(5)
Jiangxi	660	273	142	58	(7)	65	86	20	20	(3)
Jilin	644	298	132	81	4	72	69	31	31	(3)
Liaoning	950	447	196	106	(13)	111	108	31	31	(4)
Ningxia	794	370	140	86	24	64	66	30	30	(2)
Qinghai	1053	239	138	66	22	86	111	24	24	(3)
Shaanxi	711	379	152	100	28	63	84	32	32	(4)
Shandong	1197	612	281	193	0	134	209	40	40	(11)
Shanghai	372	165	75	40	(11)	38	37	8	8	(2)
Shanxi	692	347	131	97	18	62	70	20	20	(3)
Sichuan	945	494	285	67	46	83	132	45	45	(7)
Tianjin	521	239	116	74	4	57	83	12	12	(4)
Tibet	73	0	0	0	(0)	0	0	0	0	(0)
Xinjiang	1128	259	85	53	(14)	38	31	25	25	(1)
Yunnan	585	146	86	15	25	43	29	13	13	(0)
Zhejiang	370	149	66	32	(12)	31	32	9	9	(1)

*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ and attributable DALY rate.

Table V.15. Age-Standardized DALY Rate (DALYs/100,000) Attributable to $PM_{2.5}$ by Source and Province: 2030 under Scenario PC2.*

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
China	727	359	185	76	12	103	53	32	32	(4)
Hong Kong	390	149	66	24	(8)	36	16	7	7	(1)
Anhui	617	292	167	81	(11)	97	54	34	34	(5)
Beijing	490	248	133	73	18	80	37	16	15	(3)
Chongqing	631	374	187	28	61	70	42	35	35	(3)
Fujian	610	259	141	52	(35)	80	35	18	18	(3)
Gansu	587	267	115	64	34	70	39	29	29	(2)
Guangdong	552	276	131	44	(15)	73	35	18	18	(2)
Guangxi	1085	514	253	60	48	127	72	46	46	(5)
Guizhou	793	463	175	26	129	71	39	33	33	(3)
Hainan	203	69	33	14	(3)	20	9	6	6	(1)
Hebei	1255	661	345	188	34	203	95	46	46	(9)
Heilongjiang	558	176	114	35	6	77	29	36	36	(2)
Henan	961	518	277	146	13	151	84	44	44	(8)
Hubei	805	440	221	81	35	121	66	40	40	(6)
Hunan	734	399	204	58	28	107	58	32	32	(5)
Inner Mongolia	641	268	101	68	32	61	25	18	18	(1)
Jiangsu	605	287	162	81	(26)	98	50	28	28	(5)
Jiangxi	563	265	151	60	(13)	87	48	23	23	(4)
Jilin	551	210	136	40	(6)	89	30	32	32	(2)
Liaoning	826	357	217	66	(31)	144	52	34	34	(4)
Ningxia	702	333	136	88	32	80	38	32	32	(3)
Qinghai	935	263	122	43	18	86	51	24	24	(2)
Shaanxi	623	356	145	99	44	79	44	36	36	(4)

Table continues on next page

Table V.15 continued*. Age-Standardized DALY Rate (DALYs/100,000) Attributable to $PM_{2.5}$ by Source and Province: 2030 under Scenario PC2.

Location	All Ambient $PM_{2.5}$	Coal related				Noncoal related				
		Total Coal	Industrial Coal	Power Plant	Domestic Coal	Noncoal Industrial	Transport	Domestic Biomass Burning	Open Burning	Solvent Use
Shandong	1032	527	297	139	(11)	174	91	49	49	(9)
Shanghai	324	150	79	39	(20)	50	22	9	9	(2)
Shanxi	612	346	147	104	29	82	38	23	23	(3)
Sichuan	792	453	231	54	57	94	64	52	52	(5)
Tianjin	457	220	133	61	4	81	38	17	17	(4)
Tibet	71	1	0	(1)	(1)	0	0	0	0	0
Xinjiang	1067	233	76	19	(31)	48	16	24	24	(1)
Yunnan	505	137	56	10	28	29	14	13	13	(0)
Zhejiang	320	144	71	34	(20)	41	20	9	9	(2)

*Values in parentheses indicate that a reduction in source sector emissions (e.g., solvent use) leads to an increase in $PM_{2.5}$ and attributable DALY rate.

ADDITIONAL MATERIALS ON THE HEI WEB SITE

Additional Materials 1: Graphical Displays of Estimated Chinese National and Province-Specific Absolute and Age-Standardized $PM_{2.5}$-Attributable Deaths and DALYs by Air Pollution Source for 2013 and Four Future Scenarios in 2030

Additional Materials 2: Data Showing Estimated Chinese National and Province-Specific $PM_{2.5}$ Levels, Absolute and Age-Standardized $PM_{2.5}$-Attributable Deaths, and DALYs by Air Pollution Source and Their 95% Uncertainty Intervals for 2013 and Four Future Scenarios in 2030

ABBREVIATIONS AND OTHER TERMS

AAP	ambient air pollution
AS	active smoking
BAU	business as usual scenario
BC	black carbon
CMAQ	community multiscale air quality model
CNEMC	China National Environmental Monitoring Center
COPD	chronic obstructive pulmonary disease
CPS II	Cancer Prevention II
CTM	chemical transport model
DALY	disability-adjusted life year
FGD	flue gas desulfurization
GBD	global burden of disease
GBD MAPS	global burden of disease from major air pollution sources
GEOS-Chem	Goddard Earth Observing System global chemical transport model
HEI	Health Effects Institute
HAP	household air pollution
IARC	International Agency for Research on Cancer
IER	integrated exposure–response
IHD	ischemic heart disease
IHME	Institute for Health Metrics and Evaluation
ISA	Integrated Science Assessment
LC	lung cancer
LRI	lower respiratory infection
MEP	Ministry of Environmental Protection of China
MICS	model intercomparison study
Mt	metric ton
NBS	National Bureau of Statistics
NH_3	ammonia

NH_4^+	ammonium
NMVOC	nonmethane volatile organic compound
NO_3^-	nitrate
NO_x	nitrogen oxides
NRDC	Natural Resources Defense Council
OC	organic carbon
ORVOCs	other reactive volatile organic compounds
PAF	population attributable fraction
PC	(alternative) policy scenario
PM	particulate matter
PM_{10}	particulate matter $\leq$ 10 µm in aerodynamic diameter
$PM_{2.5}$	particulate matter $\leq$ 2.5 µm in aerodynamic diameter
SCR	selective catalytic reduction
SHS	second-hand smoke
SNCR	selective noncatalytic reduction
SO_4^{2-}	sulfate
STD	standard simulation
SO_2	sulfur dioxide
THUBERC	Tsinghua University Building Energy Research Center
TMREL	theoretical minimum risk exposure level
TSP	total suspended particulate
U.S. EPA	U.S. Environmental Protection Agency
VOC	volatile organic compound
WHO	World Health Organization

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