Health benefits of on-road transportation pollution control programs in China

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China started to implement comprehensive measures to mitigate traffic pollution at the end of 1990s, but the comprehensive effects, especially on ambient air quality and public health, have not yet been systematically evaluated. In this study, we analyze the effects of vehicle emission control measures on ambient air pollution and associated deaths attributable to long-term exposures of fine particulate matter ($PM_{2.5}$) and O_3 based on an integrated research framework that combines scenario analysis, air quality modeling, and population health risk assessment. We find that the total impact of these control measures was substantial. Vehicular emissions during 1998–2015 would have been 2–3 times as large as they actually were, had those measures not been implemented. The national population-weighted annual average concentrations of $PM_{2.5}$ and O_3 in 2015 would have been higher by 11.7 μg/m^3 and 8.3 parts per billion, respectively, and the number of deaths attributable to 2015 air pollution would have been higher by 510 thousand (95% confidence interval: 360 thousand to 730 thousand) without these controls. Our analysis shows a concentration of mortality impacts in densely populated urban areas, motivating local policymakers to design stringent vehicle emission control policies. The results imply that vehicle emission control will require policy designs that are more multifaceted than traditional controls, primarily represented by the strict emission standards, with careful consideration of the challenges in coordinated mitigation of both $PM_{2.5}$ and O_3 in different regions, to sustain improvement in air quality and public health given continuing swift growth in China's vehicle population.

traffic pollution control | scenario analysis | air quality | mortality impact | China

Air pollution in China has become a source of substantial environmental and social concern over the past 2 decades (1). To mitigate air pollution, the government has undertaken dramatic efforts to control emissions from major sources including power generation, industrial plants, and road transportation. Nevertheless, the national annual mean particulate matter ($PM_{2.5}$) concentration (population-weighted) increased from 35 μg/m^3 in 2000 to 52 μg/m^3 in 2013 (2). More recently, since the promulgation in 2013 of China's toughest-ever Air Pollution Prevention and Control Action Plan (3), the $PM_{2.5}$ concentration has substantially declined, by 32% in 2017 (4). Even with these recent improvements, China's annual mean $PM_{2.5}$ concentration is still ~3 times higher than the World Health Organization (WHO) standard level (i.e., 10 μg/m^3) for good health (5) and causes substantial health impacts. The 2017 Global Burden of Disease (GBD) study (6) estimated that 850 (95% confidence interval [CI 95]: 700–980) thousand deaths in China were attributable to ambient $PM_{2.5}$ exposure in 2015 and another 170 thousand related to ambient O_3 exposure. The more recent Global Exposure Mortality Model (GEMM) (7) study estimated the death toll attributable to $PM_{2.5}$ in China in 2015 to be even higher, at 2.47 million, using new epidemiological evidence.

Aside from high emissions, the proximity of vehicle emissions to dense urban populations (8, 9) makes the road transportation sector an important target for pollution control around the world. In China, the total vehicle population, including motorcycles, reached 327 million in 2018, an eightfold increase since the end of the 1990s (10). Road transportation has become a major source of ambient air pollution in many large Chinese cities (11). According to a report from Ministry of Environmental Protection in 2015 (12), mobile sources (primarily on-road vehicles) were responsible for 20–40% of ambient $PM_{2.5}$ concentrations in the central and eastern regions of China. With the significant progress in controlling pollution from coal combustion, vehicle emissions are now deemed responsible for 45% of ambient $PM_{2.5}$ concentrations in Beijing (13). Transportation emissions are also important precursors of urban O_3 pollution in China, responsible for one third to one half of the O_3 concentrations measured in large cities like Shanghai, Nanjing, and Hangzhou, in the summer of 2015 (14). Although $PM_{2.5}$ concentrations have declined since 2013, peak-season O_3 pollution has increased.

Globally, comprehensive control actions have been implemented to mitigate vehicle emissions in recent decades. Notably, in the United States the full phase-in of standards under the

Significance

On-road vehicles have become a significant source of air pollution in Chinese cities. Air pollution in China has substantial impacts on public health and is thought to be responsible for millions of deaths each year. The government has implemented comprehensive control measures to mitigate vehicle emissions since the late 1990s. In this paper, we analyze how the historical control actions during 1998–2015 affected air quality and public health. We constructed an integrated framework combining emission scenarios, air quality modeling, and population health risk assessment. Our results demonstrate that the control measures have led to substantial improvements in public health and provide insight into strategies for further mitigation of negative effects from China's future motorization on air quality and public health.

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Clean Air Act in 1981 marked a milestone for transportation pollution control, requiring new cars to be equipped with three-way catalytic converters and to use unleaded gasoline. In China, implementation of vehicle emission standards is also one of the major traffic pollution control policies. The China 1 emission standard, equivalent to Euro 1 (note: the emission standards for light-duty vehicles are written with Arabic numerals [1–6], while the standards for heavy-duty vehicles are indicated using Roman numerals [I to VI]), was implemented in Beijing in 1999 and nationwide in 2000 (15), eliminating high-emitting carbureted gasoline car engines. With implementation of increasingly stringent emission standards since then (under succeeding China 2/II to China 6/VI programs), more advanced engine and after-combustion treatment technologies have rapidly penetrated China's vehicle fleet, further reducing emission factors for both gasoline and diesel vehicles (16). Alongside the regulatory advances in emission standards and fuel quality, in-use inspection programs, transportation management strategies, promotion of electric vehicles, and economic incentives have been utilized by national and local policymakers (17).

Researchers have assessed the effects of traffic pollution controls around the world from many perspectives, such as emission mitigation, air quality improvement, and health damages reduction (17–20). The majority of studies of traffic pollution controls in China have assessed benefits solely in terms of their impacts on emissions at either the national (16, 21) or local level (22, 23). A few studies (24–26) have also evaluated the impacts of vehicular emissions on air quality in China, relying on chemical transport models to account for the complex nonlinear chemistry of secondary pollutants (e.g., $PM_{2.5}$, O_3). However, even these studies were limited by their focus on a few specific policies or regions or on the analysis of limited timeframes. They did not elucidate the long-term impacts on air quality across all regions of China. Regional differences are significant in China, a country with a large territory that has experienced urbanization and motorization at different rates in different areas. Consequently, until now policymakers have not been explicitly informed of the detailed air quality benefits, let alone the health benefits, from its 2-decade-long effort to control emissions from on-road vehicles.

The goal of this paper is to estimate the differences in ambient $PM_{2.5}$ and O_3 concentrations and in the mortality attributable to exposure to these pollutants between the current situation and the situation that would have existed if the government had not mandated aggressive pollution controls on transportation sources. We apply an integrated assessment framework (*SI Appendix*, Fig. S1) comparing two scenarios for the period from 2000 to 2015, namely scenarios with and without control measures, and utilize chemical transport modeling and health risk assessment to quantify the effects on ambient $PM_{2.5}$ and O_3 concentrations and the mortality from long-term exposure to these pollutants (*SI Appendix*, *Note S3*). We perform the air quality simulations at a spatial resolution of 36 km × 36 km across China, with finer 4-km-scale simulations for two coastal regions of Beijing–Tianjin–Hebei (BTH) and the Yangtze River Delta (YRD, including Jiangsu, Shanghai, Zhejiang, and Anhui). This study, a comprehensive assessment of health benefits of vehicle emission controls across China since the late 1990s, not only provides an important summary of China's experience with vehicular pollution controls, but also has implications for prioritizing future vehicular emission control strategies in China and other developing countries.

Results and Discussion

Emissions Mitigation. Vehicular emissions of nonmethane volatile organic compounds (NMVOCs) and carbon monoxide (CO) have generally declined in China from 2005 to 2015 (*SI Appendix*, Fig. S2). In contrast, NO_X emissions have increased—initially quite rapidly, but at a lower rate of increase more recently (Fig. 1). This is primarily because NO_X emission standards for heavy-duty diesel trucks (HDDTs), the major source of on-road NO_X emissions, were barely reduced during 2005–2013 under the China III standards compared to those of China II (16). The China IV emission standards, which targeted NO_X emissions and required HDDTs to use urea-based selective catalytic reduction converters, were delayed until 2014 due to inadequate production and supply of the required low-sulfur diesel fuels (i.e., sulfur content below 50 ppm) (17). NO_X emission factors for new HDDTs (i.e., g NO_X per kilometer traveled) declined little from 2005 to 2013 and led to an upward trend in total on-road NO_X emissions. In contrast, primary $PM_{2.5}$ emissions from HDDTs dropped significantly from 2010 to 2015 (*SI Appendix*, Fig. S3), owing to improvement in diesel engine technologies in response to increasingly stringent $PM_{2.5}$ emissions standards (from 0.10 to 0.02 g $PM_{2.5}$ per kWh) (15), and to rapid retirement of older HDDTs with much higher emissions.

Complicating emission control efforts, however, was a fourfold increase in China's vehicle population from 2005 to 2015, a far higher growth rate than that of NO_X emissions. Comparing actual emissions with a counterfactual scenario without emissions controls (Table 1), in 2015 China reduced on-road vehicle emissions of NMVOCs, CO, NO_X, and $PM_{2.5}$ by 84%, 88%, 59%, and 87%, respectively (*SI Appendix*, Fig. S4). The total emissions reductions of NMVOCs, CO, NO_X, and $PM_{2.5}$ over the 17-y period from 1998 to 2015 are estimated at 105.3 millions of metric tonnes (Mt) (CI 95: 76.8 to 151.6 Mt), 1,050.6 Mt (756.4 to 1502.3 Mt), 83.9 Mt (57.8 to 114.9Mt), and 12.8 Mt (9.0 to 17.6 Mt), respectively (*SI Appendix*, Fig. S2). These reductions are substantial in comparison with actual emissions. Had the control measures not been implemented, on-road vehicular emissions during the period would have been 2–3 times as large as they actually were.

Air Quality Benefit. China's efforts to control vehicle emissions since 1998 have delivered ever-increasing air quality benefits, with national annual average $PM_{2.5}$ concentrations (i.e., not weighted by population) in 2005 being 0.7 μg/m^3 (CI 95: 0.5 to 1.1 μg/m^3) lower than they would have been without controls and in 2015 being 3.9 μg/m^3 (CI 95: 2.5 to 6.0 μg/m^3) lower (*SI Appendix*, Figs. S5A and S6). The impacts of the controls on population-weighted exposures to annual average $PM_{2.5}$ pollution

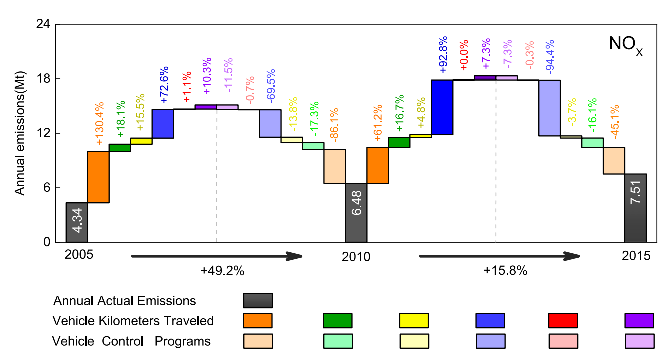

Fig. 1. Decomposition of the changes in vehicular NO_X emissions during the periods of 2005–2010 and 2010–2015, respectively. The gray bars represent the annual actual vehicle emissions (i.e., with controls). The dark-colored bars on the left of the dashed lines represent the impacts of vehicle kilometers traveled (*VKT*) by various vehicle fleets on emissions growth during each period. The light-colored bars on the right of the dashed lines represent the impacts of vehicle control programs on emissions reduction. Vehicle fleets include heavy-duty trucks (HDTs), light-duty truck (LDT), medium-duty passenger vehicle (MDPV), heavy-duty passenger vehicle (HDPV), light-duty passenger vehicle (LDPV), motorcycle, taxi, and bus.

Table 1. Description of the scenarios of on-road vehicle emissions

Scenarios	Descriptions
Scenario 1, with controls	On-road vehicle emissions as actually realized in China, E_1. This scenario includes the effects of all control measures of vehicular emissions implemented in China since 1998.
Scenario 2, without controls	On-road vehicle emissions without the actual control measures (i.e., a counterfactual scenario), E_2. Under this scenario, no tightened control policies are assumed to have been implemented since 1998. Specifically, all new vehicles from 1998 to 2015 are assumed to have only met pre-China I emission standards.

are estimated to be much greater, 1.8 μg/m^3 in 2005 and 11.7 μg/m^3 in 2015, reflecting the high densities of population in areas with high traffic densities and associated pollution. The five provincial-level regions where the controls are thought to have the greatest impact on population-weighted annual average $PM_{2.5}$ exposure in 2015 were Beijing (33.4 μg/m^3), Shanghai (28.2), Jiangsu (23.9), Tianjin (23.3), and Zhejiang (17.4) (Fig. 2*A* and *SI Appendix*, Table S1), all populous coastal areas.

By pioneering the implementation of new emissions standards, Beijing reduced vehicular emissions in 2015 by ~90% compared to a counterfactual scenario of no controls (*SI Appendix*, Fig. S7) and thus is projected to have caused the annual average $PM_{2.5}$ concentration in that year to be 19.2 μg/m^3 (CI 95: 12.5 to 29.7 μg/m^3) lower than it would have been without controls. The air quality impact of the policy in Beijing shows a spatial tendency common in urban agglomerations (Fig. 2*B*), with estimated impacts on annual $PM_{2.5}$ concentrations as large as 50–60 μg/m^3 in the urban core (i.e., within the city's Fifth Ring Road). For the YRD region, the estimated impact on the annual average $PM_{2.5}$ concentration in 2015 due to vehicle emission controls was to reduce them by 16.3 μg/m^3 (CI 95: 10.6 μg/m^3 to 25.2 μg/m^3) from the values that would have been anticipated without controls (Fig. 2*C*). However, unlike Beijing, in the YRD region the benefits are projected to have been relatively evenly distributed between rural and urban areas. This is partially because Beijing has expanded radially from a monocentric urban core, while the YRD region includes many cities that have well-developed suburban areas, reflecting a polycentric urban development strategy in the Shanghai region. Additionally, emissions are considered more easily diffused in the relatively flat YRD region compared to Beijing, which is topographically bounded by mountains in the west, north, and northeast (27).

Vehicle emission controls are also projected to have reduced China's seasonal (April–September) average 8-h maximum O_3 concentrations, with concentrations in 2015 being 4.3 parts per billion (ppb) (CI 95: 3.0 to 6.2 ppb) lower than they would have been without controls (*SI Appendix*, Fig. S5*B*). The greatest impacts are estimated to have occurred in the Sichuan–Chongqing and YRD regions (Fig. 3*A*). In contrast, in a few cities (most notably in the urban core of Beijing [Fig. 3*B*]), O_3 levels were projected to be higher than they would have been without the controls. There are a number of possible explanations for the increases in O_3 pollution in these areas. First, with abundant NO_X emission sources, O_3 production in the urban area of Beijing has been characterized as VOC-limited, implying that reductions of NO_X emissions may have little effect on O_3 concentrations or even increase them. Second, the reduction in ambient $PM_{2.5}$ concentrations due to vehicle emission controls might foster O_3 formation through photochemical or other mechanisms (28, 29). For example, $PM_{2.5}$ control could reduce termination reactions of some radicals (e.g., hydroperoxy radical, HO_2) that occur on particle surfaces, facilitating the HO_2+NO reaction and thus promoting O_3 formation (29). Lower levels of $PM_{2.5}$ also mean less particles reflecting or absorbing ultraviolet

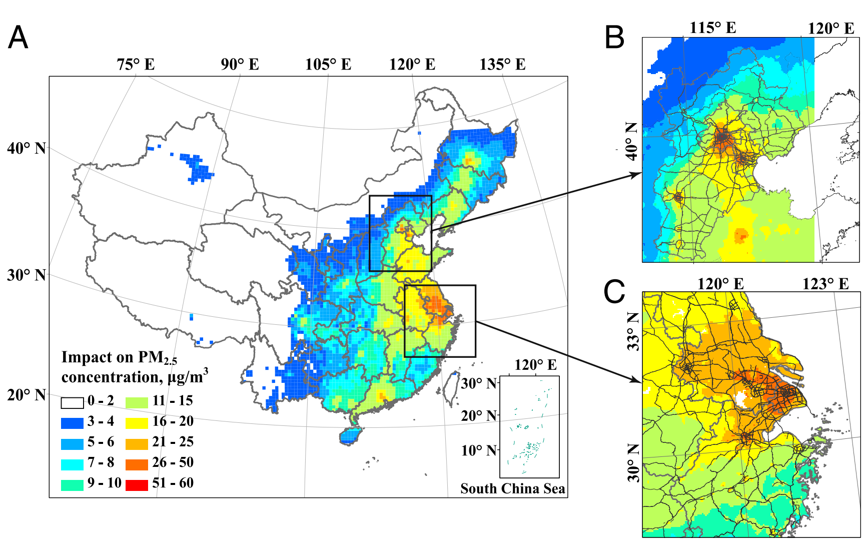

Fig. 2. Geographic distribution of the impacts of vehicle control programs since 1998 on annual spatial average $PM_{2.5}$ concentrations of China (*A*), BTH (*B*), and YRD (*C*) in 2015. The resolution of China is at 36 km × 36 km, while BTH and YRD are at a resolution of 4 km × 4 km. The data of impacts represent the projected difference between with- and without-control scenarios. Positive values imply the actual (i.e., with controls) $PM_{2.5}$ concentrations in that year are lower than they would have been without controls, and vice versa.

radiation from the sun, potentially strengthening the photochemical production of O_3.

Health Impact. The impact on air quality described above is expected to have led to almost 60 thousand (CI 95: 40 thousand to 90 thousand) fewer deaths attributable to 2005 ambient $PM_{2.5}$ levels and almost 463 thousand (CI 95: 310 thousand to 680 thousand) fewer deaths attributable to 2015 levels (*SI Appendix*, Fig. S6). This estimated effect is nearly 20% of the deaths attributable to 2015 ambient $PM_{2.5}$ pollution (7) and 2~8 times the annual deaths of traffic crashes in China (30). Major impacts were estimated for the coastal regions in East China, including the BTH region (50 thousand fewer deaths attributable to $PM_{2.5}$), the YRD region (125 thousand), and Shandong province (40 thousand) (Fig. 4*A* and *SI Appendix*, Table S2). These three regions account for less than 10% of China's territorial area and 27% of the population but more than 45% of the mortality impact of the 2015 transportation emissions control measures.

The implementation of vehicle emission controls also had impacts on the deaths attributable to ambient O_3 exposure. In 2005, 10 thousand (CI 95: 7 thousand to 15 thousand) fewer deaths attributable to O_3 exposure were projected, with the number rising to 47 thousand (CI 95: 32 thousand to 70 thousand) in 2015 (Fig. 4*B* and *SI Appendix*, Fig. S8). Coinciding with a dense population and NO_X-limited O_3 formation mechanism (31), the nearly 50% decline in cumulative traffic NO_X emissions since 1998 has greatly alleviated O_3 pollution in the East Coast and central China, accounting for over 50% of the reduction in mortality attributable to O_3 mitigation in China in 2015 (*SI Appendix*, Fig. S8 and Table S3). Although vehicle emission controls are estimated to have led to nearly 1 thousand additional deaths attributable to O_3 in Beijing in 2015, this increase is far less when compared to the reduction in deaths attributable to $PM_{2.5}$ in the same year, estimated at more than 17 thousand, through the same emission control efforts. In summary, the historical measures implemented to control vehicle emissions are projected to have resulted in significant health benefits across China when taking account of both $PM_{2.5}$ and O_3-related mortality (Fig. 4*C*).

Of course there are uncertainties in these estimates—both due to uncertainty in estimating the emissions changes resulting from the regulations and the corresponding changes in air quality and due to uncertainty about the shape and parameters of the true exposure–response (E-R) function relating $PM_{2.5}$ exposure to mortality. Considering only the parameter uncertainty in GEMM E-R function suggests uncertainty on the order of 10% of the best estimate. If this were the only source of uncertainty, the impact on deaths attributable to 2005 $PM_{2.5}$ exposure would be estimated to be between 54 thousand and 66 thousand (CI 95) fewer than would have occurred in the absence of controls; and the impact on deaths due to 2015 $PM_{2.5}$ exposure would be estimated to be between 420 thousand and 510 thousand. However, when uncertainties in estimating emissions (CI 95: −40% to +40%, see *SI Appendix, Note S1*), modeling concentrations (CI 95: −15% ~ +15%, see *SI Appendix, Note S2*), and population exposure (CI 95: −5% ~ +5%, see *SI Appendix, Note S2*) are considered, the uncertainty in the estimate of attributable deaths substantially increases. Accounting for these other sources of uncertainty, 40–90 thousand and 310–680 thousand attributable deaths due to $PM_{2.5}$ exposure are estimated to have occurred in 2005 and in 2015, respectively.

We note that the true uncertainty of air pollution-related mortality might be much greater than suggested above, as the CI 95 given above reflects only parameter uncertainty and therefore assumes that the E-R function (i.e., GEMM NCD+LRI in this paper) provides the correct synthesis of the epidemiological literature. Had these estimates been based on the GBD IER E-R function (6) they would have been nearly a factor of 4 lower, suggesting roughly 100 thousand fewer attributable deaths due to these controls (see details in *SI Appendix, Note S3.2*). Therefore, we conclude that the lack of knowledge of the true E-R function is the largest contributor to the true uncertainty of our results. Despite these large uncertainties, the impacts are substantial and point to the need for even more stringent controls on transportation sources going forward.

Policy Implications. China has experienced a period of dramatic growth in vehicle populations and traffic activity during the last 2 decades, making on-road transportation an increasingly important source of ambient air pollution. Our results show that, despite this growth in traffic, strict control measures have mitigated large amounts of vehicular emissions. The enforcement of vehicle emission control policies has also brought substantial benefits in mitigating health impacts from ambient air pollution across China, keeping the deaths attributable to 2015 air pollution (i.e., $PM_{2.5}$ and O_3) 510 thousand lower than the levels that would have been expected without the controls (*SI Appendix*, Table S4). Urban areas with comparatively dense populations, which typically bear the largest share of the public health impacts of traffic emissions, have also benefited the most from the vehicle emission controls. For example, the estimated impacts on deaths attributable to ambient air pollution per 1,000 km^2 in 2015 are ~3,000 for Shanghai and 1,000 for Beijing, significantly higher than those for the larger developed regions that include these cities (i.e., ~380 for YRD and ~240 for BTH, respectively) and for the nation as a whole (~50). This record indicates that

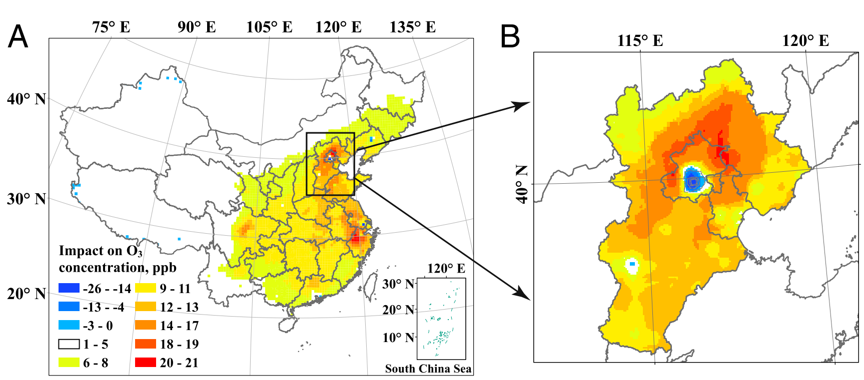

Fig. 3. Geographic distribution of the impacts of vehicle control programs since 1998 on seasonal (April–September) average 8-h maximum O_3 concentrations of China (*A*) and BTH (*B*) in 2015. The resolution of China is at 36 km × 36 km, while BTH is at a resolution of 4 km × 4 km. The data of impacts represent the projected difference between with- and without-control scenarios. Positive values imply the actual (i.e., with controls) O_3 concentrations in that year are lower than they would have been without controls, and vice versa.

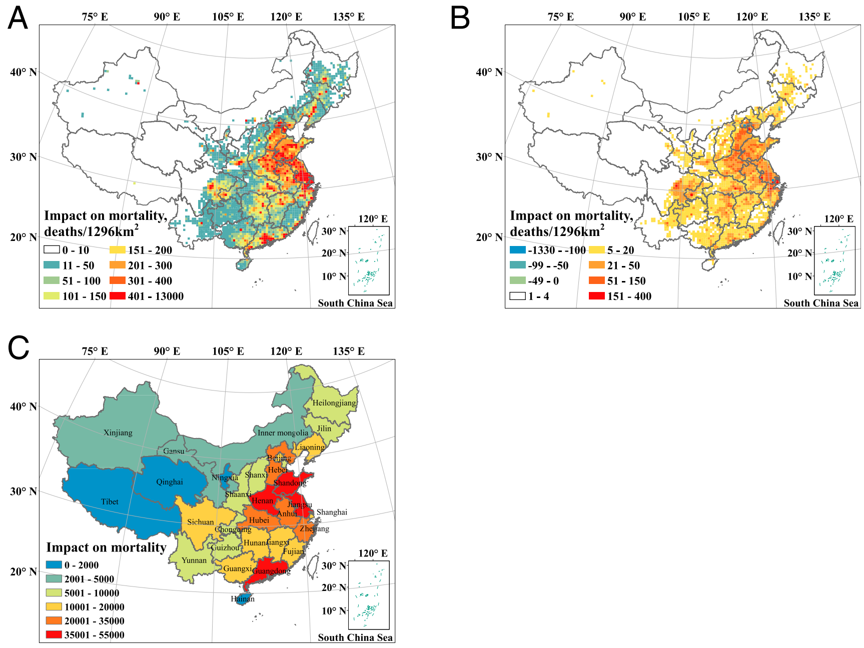

Fig. 4. Geographic distribution of the impacts of vehicle control programs since 1998 on mortality attributable to ambient $PM_{2.5}$ (*A*), O_3 (*B*), and both of $PM_{2.5}$ and O_3 (*C*) across China in 2015. *A* and *B* are illustrated at a resolution of 36 km × 36 km, while *C* reports provincial totals. The data of impacts represent the projected difference between with- and without-control scenarios. Positive values indicate that mortality attributable to air pollution in that year is lower than it would have been without controls, and vice versa.

continuous implementation of even more stringent policy measures may be crucial to mitigate impacts of ongoing growth in China's vehicle ownership and use on air pollution and public health. Especially in the most populous regions, prospective health benefits should drive implementation of more aggressive and locality-specific control policies.

With the implementation of the most stringent clean air policies to date in China since 2013, air pollutant emissions have been substantially mitigated, especially those from industrial and residential sectors, contributing to strong declines in average ambient $PM_{2.5}$ concentrations between 2013 and 2017 (4). On the other hand, vehicular emissions have been responsible for even increasing shares of local ambient $PM_{2.5}$ concentrations, especially in some densely populated urban areas of China. In response to the vehicle emission control challenges, the government's ongoing action plan for "winning the blue-sky defense battle" (2018–2020) has explicitly required strategic transitions in energy and transportation structures to accelerate air quality improvements (32). Ambitious vehicle emission control in China will indeed require policy designs that are more multifaceted other than the traditional controls, i.e., ever-tightening emission standards alone. Important policy levers for policymakers include promotion of fleet electrification, implementation of advanced emission inspection systems, urban redesign to minimize the need for automotive transport, and promotion of rail-based systems of all types for both urban passenger and regional freight transport. For example, fleet electrification (especially if served by increasing renewable or nuclear shares of power generation) can readily reduce the annual average concentrations of $PM_{2.5}$ and O_3, resulting in health benefits that are concentrated in urban areas of densely populated cities (33). By contrast, progress may be more mixed absent substantial penetration of electric vehicles. For example, the current decline of total on-road NMVOCs emissions might cease sometime between 2020 and 2025 even as the stricter China VI emission standard takes effect in 2020 (16). All such achievements and lessons gained in China can also help inform efforts to control vehicle pollution in other developing countries (e.g., India) that are now undergoing similarly fast urbanization and motorization.

Methods

Developing Vehicular Emissions Inventory and Policy Scenarios. We developed detailed emission factors based on vehicle type, emission standards, vehicle age, fuel type, and calendar year to analyze on-road vehicle emissions in China (17). This bottom-up approach was also employed here to calculate vehicle emissions of NMVOCs, CO, NO_X, and $PM_{2.5}$ for each Chinese province, as described by Eq. **1**.

$$E_{j,p,y} = \sum_t \sum_f \sum_i \left(10^{-6} \times VP_{f,t,i,p,y} \times VKT_{f,t,i,p,y} \times EF_{f,t,j,i,p,y}\right) \quad [1]$$

where $E_{j,p,y}$ is the annual total emissions of pollutant j for province p in the calendar year y (e.g., 2015), ton; $VP_{f,t,i,p,y}$ is vehicle population defined by fuel type f, vehicle classification t, and vehicle age i; $VKT_{f,t,i,p,y}$ is the corresponding annual fleet average VKT, km; and $EF_{f,t,j,i,p,y}$ means the average emission factor, g km^{-1}.

As shown in Table 1, we designed two scenarios, which indicate the emissions with and without vehicle emission control programs implemented in China since 1998, respectively. The differences between these two scenarios represent the effectiveness of the implemented control measures on vehicular emissions across China. Our previous study (17) developed the emission inventories for the two scenarios limited to 1998–2013, which are extended to 2015 in this paper based on the same methodology.

We have built probability-based distribution functions to address the uncertainties of key parameters of the emission model based on detailed experimental data or investigation results (17, 34). Uncertainties in emission factors and total vehicle emissions were analyzed with the Monte Carlo simulations by taking account of the probability distribution of these key model parameters or input variables. Uncertainty ranges of vehicular emissions at a 95% confidence level were estimated for each year by running 100 thousand trials of Monte Carlo simulations. The uncertainties of China's vehicular emissions under the with-control scenarios have been updated to 2015, with −19%∼+40% (CI 95), −37%∼+38%, −27∼+44%, and −28%∼+43% for NO_X, NMVOCs, CO, and $PM_{2.5}$ emissions, respectively. For the uncertainty under the without-control scenario, we referred to the 95% CI estimate for the year of 1998 (i.e., before the implementation of vehicle emission controls) under the with-control scenario, with −31%∼+37% (CI 95), −27%∼+44%, −28∼+43%, and −29%∼+35%, respectively, for above-mentioned pollutant emissions. More details about the parameters in calculations of vehicular emissions and the uncertainty information are presented in *SI Appendix*, *Note S1*, and Dataset S1 *A–C*.

Modeling Air Quality Benefits of Vehicle Emission Controls. The Community Multiscale Air Quality (CMAQ) model (version 5.0.1), enhanced by the two-dimensional volatility basis set (2D-VBS) module, which has been proven to improve the secondary organic aerosol (SOA) simulation in previous studies (35, 36), was applied to evaluate the ambient surface $PM_{2.5}$ and O_3 concentrations. We used the 2010 meteorological data, in line with previous studies developing and validating the 2D-VBS module, as input of the Weather Research and Forecasting (WRF) model.

A multisector emission inventory developed by our group (37) was applied to quantify the total trends of five major pollutants (primary $PM_{2.5}$, NO_X, SO_2, NMVOCs, and NH_3) for 3 y (2005, 2010, and 2015). The methods and data sources for the on-road transportation sector are consistent with scenario 1 (with controls). For other sectors, the methods, data, and validation processes have been documented in previous papers (38, 39). We then substituted the on-road transportation emissions with those generated for scenario 2 (without controls) as inputs to CMAQ to estimate the air quality conditions without implementation of historical vehicle emission controls.

We performed the simulation at a horizontal resolution of 36km × 36km across China, as well as at a finer resolution of 4km × 4km in BTH and YRD. The differences of CMAQ-simulated concentrations of $PM_{2.5}$ and O_3 between the two scenarios were calculated to indicate the fractional reduction (FR) of air quality across China due to vehicle emission controls (Eq. **2**). These derived fractions were multiplied by high-resolution satellite-derived $PM_{2.5}$ concentration data for China (40) and the O_3 concentration data developed for the GBD analyses (6) to obtain the air quality impacts attributable to vehicle emission controls (Eq. 3), which is similar to previous studies (20, 41, 42).

$$FR = \frac{C_{CMAQ,E2} - C_{CMAQ,E1}}{C_{CMAQ,E1}} \quad [2]$$

$$\Delta C = FR \times C \quad [3]$$

where *FR* is the difference of $PM_{2.5}$ or O_3 concentrations attributed to the vehicular emission control programs; $C_{CMAQ,E1}$ and $C_{CMAQ,E2}$ represent CMAQ-modeled concentration using emissions under scenarios 1 and 2, respectively; ΔC is the reduced ambient concentrations related to vehicle emission controls; and *C* represent satellite-derived $PM_{2.5}$ and GBD O_3 concentrations.

Our previous studies have illustrated the detailed setup of CMAQ (33) and validation against field observation results (36, 43). For example, Zhao et al. (43) verified the good agreements of 1-h temporal patterns of $PM_{2.5}$ concentrations in Beijing and Shanghai against the concurrent observations (normalized mean biases [NMBs] of −8.5% and −13%, respectively, based on cross-season comparisons in May, August, and November). Ding et al. (44) evaluated the CMAQ simulation results of O_3 concentrations (from April to September) against the observation results released by the Ministry of Ecology and Environment. The daily-specific validations suggest most of the comparison dates were within a factor of 2. And the validation results for 2015 following the same method indicate the monthly NMBs of 2%∼19%, with an average NMB of 9.8%. The satellite-derived $PM_{2.5}$ concentrations have already been calibrated by surface observations, with small uncertainty of approximately ±5% on average (45). See details about the CMAQ modeling and calibrating in *SI Appendix*, *Note S2*.

Evaluating the Health Impacts. Exposure to ambient air pollution is associated with a range of acute and chronic health effects (5). We focus on the mortality impacts of long-term exposure to ambient air pollution in this study (see detailed discussion about chronic and acute effects of air pollution in *SI Appendix*, *Note S3.1*). $PM_{2.5}$ is currently the dominant contributor to mortality from long-term exposure to air pollution (46), and the evaluation of adverse health impacts associated with elevated ambient $PM_{2.5}$ concentrations has been supported by extensive epidemiological cohort studies (7, 47–49). Health impacts of O_3 have also been observed, which are independent of associations between $PM_{2.5}$ and mortality (50, 51). We limited our health effects on these two metrics, which are consistent with that of the GBD study (42) and the US Environmental Protection Agency's (US EPA) analysis on the Clean Air Act (52).

We apply GEMM (7) to estimate China's deaths attributable to outdoor $PM_{2.5}$ exposure for noncommunicable diseases (NCDs) and lower respiratory infections (LRIs). The relative risk (RR) functions for GEMM were constructed on the basis of 41 cohort studies only regarding outdoor air pollution from 16 countries, including a recent cohort study in China (53). The RR function used by GEMM could be expressed as Eq. **4**:

$$RR_{j,k}(C_i) = \begin{cases} \exp\left\{\dfrac{\theta_{j,k}\log\left(\dfrac{C_i - C_{cf}}{\alpha_{j,k}} + 1\right)}{1 + \exp\left\{-\dfrac{C_i - C_{cf} - \mu_{j,k}}{v_{j,k}}\right\}}\right\} & \text{if } C_i > C_{cf} \\ 1 & \text{otherwise} \end{cases} \quad [4]$$

where $RR_{j,k}(C_i)$ is the RR of exposure to annual average $PM_{2.5}$ concentration in grid cell *i* for health endpoint *j* in age group *k*; C_{cf} is a minimum observed level of exposure (i.e., 2.4 μg/m^3 in this study), below which no association between $PM_{2.5}$ and mortality is assumed; $\theta_{j,k}$,$\alpha_{j,k}$, $\mu_{j,k}$, and $v_{j,k}$ are estimated parameters that describe the overall shape of the concentration–response relationship, provided by Burnett et al. (7) (see details in *SI Appendix*, Table S5); and C_i is the annual average $PM_{2.5}$ concentration in grid cell *i*.

In contrast with GEMM's exclusive reliance on exposure to outdoor air pollution, the GBD adopted an integrated exposure–response (IER) model (47) that incorporated risk information concerning $PM_{2.5}$ from both outdoor pollution and other sources (e.g., secondhand smoke, household air pollution). A recent study (54) indicates that the GEMM results aligned better with the census-based estimation of deaths attributed to ambient $PM_{2.5}$ in China than the IER results. The differences between effect estimates given by GEMM and IER are analyzed in *SI Appendix*, *Note S3.2*.

We apply the relative risk of O_3 for chronic obstructive pulmonary disease (COPD) developed by ref. 6, as shown in Eq. **5**:

$$RR_j(C_i) = \begin{cases} rr^{(C_i - C_0)/10} & \text{if } C_i > C_0 \\ 1 & \text{otherwise} \end{cases} \quad [5]$$

where $RR_j(C_i)$ is the RR of exposure to seasonal (April–September) average 8-h maximum O_3 concentration in grid cell *i* for health endpoint *j* (i.e., COPD); *rr* is the relative risk of O_3 exposure for respiratory COPD mortality defined to be 1.06 (CI 95: 1.02–1.10) per 10 ppb of O_3 exposure; C_0 is theoretical minimum-risk concentration of O_3 for COPD, with a uniform distribution between 29.1 and 35.7 ppb, below which no health impacts are calculated; and C_i is the seasonal average 8-h maximum O_3 concentration in grid cell *i*.

The annual mortality attributable to ambient $PM_{2.5}$ and O_3 is estimated for each grid cell through Eqs. **6** and **7**, and then summed up for corresponding regions (55).

$$M_{i,j} = \sum_k P_{i,k} \times \hat{I}_{j,k} \times (RR_{j,k}(C_i) - 1), \text{ where } \hat{I}_{j,k} = \frac{I_{j,k}}{\overline{RR}_{j,k}} \quad [6]$$

$$\overline{RR}_{j,k} = \frac{\sum_{i=1}^{N} P_{i,k} \times RR_{j,k}(C_{i,s1})}{\sum_{i=1}^{N} P_{i,k}} \quad [7]$$

where $M_{i,j}$ is the mortality for health endpoint *j* for grid cell *i*; $P_{i,k}$ is the population of grid cell *i* in age group *k*, adapted on the basis of the Gridded Population of the World v4 database (https://sedac.ciesin.columbia.edu/data/collection/gpw-v4); $\hat{I}_{j,k}$ represents the hypothetical "underlying incidence" (i.e., cause-specific mortality rate) that would remain for age group *k* if air pollution were reduced to the theoretical minimum risk concentration throughout that region; $I_{j,k}$ is the reported national average annual disease incidence rate for endpoint *j* in age group *k*, provided by Stanaway et al. (6); $\overline{RR}_{j,k}$ represents the national population-weighted relative risk for health endpoint *j* in age group *k*, which is estimated through Eq. **7**; $RR_{j,k}(C_{i,s1})$ represents relative risk of grid cell *i* for health endpoint *j* in age group *k* under scenario 1 (i.e., with controls); and *N* is the number of cells.

The differences ($\Delta M_{i,j}$) between the numbers of deaths attributable to air pollution under scenario 2 ($M_{i,j}(S2)$, without controls) and scenario 1 ($M_{i,j}(S1)$, with controls) are calculated by Eq. **8** to represent the health effects of the vehicular emission control programs.

$$\Delta M_{i,j} = M_{i,j}(S2) - M_{i,j}(S1) \quad [8]$$

It is well known that the mortality impacts of chronic exposure to air pollution may be distributed years or even decades after the exposure. There is substantive literature on "cessation lag" which addresses this issue (56). Nonetheless, most health risk assessments [as evident in GBD (6), US EPA (52), WHO (57), and so forth] make the simplifying assumption that changes in mortality are evident in the same year as the changes in exposure. We follow this same convention, with the understanding that in reality the mortality impacts would be distributed over several years or decades after exposure (see an auxiliary analysis showing the effect of the cessation in *SI Appendix*, Table S6).

Uncertainty Analysis in Attributable Deaths. There is uncertainty in the models and parameters used to estimate the mortality attributable to air pollution. This includes uncertainty in the emission factors, uncertainty in the change in concentrations as a function of emissions, and uncertainty in the choice of E-R function and in the parameters used to calculate attributable mortality for any exposure concentration. Taken together these suggest greater uncertainty in estimates of the mortality impacts of specific air pollution policies than would result from the parameter uncertainty in GEMM alone, which is the only source of uncertainty considered in many previous studies (2, 7).

Ideally, we would have conducted a formal analysis of the propagation of uncertainty through our complete set of models (activity => emissions => concentrations => exposure => health impacts). Unfortunately, the computing time necessary to conduct such an analysis was prohibitive.

Nonetheless we were able to develop a sense of the uncertainty in our estimates by relying on a simplified approximate form of our model. This simple analysis is conducted at the national level and provides no finer geographic resolution and is limited to the impacts of fine particles, which are responsible for more than 90% of the mortality impact seen in the full model.

Because, in the region of exposure of interest, GEMM exhibits an approximately linear relationship between relative risk and exposure concentration and because, due to the use of the "fractional reduction" approach, our model behaves as if there is a proportional relationship between reductions in emissions and reductions of ambient concentrations, the mortality impact of a policy which reduces emissions can be well approximated by:

$$\Delta M = Mo \times \tilde{\beta} \times \widetilde{iF} \times \widetilde{EF} \times \varepsilon \times A \quad [9]$$

where ΔM is the reduction in the deaths attributable due to the policy, Mo is the baseline mortality ($P \times I_{hat}$), $\tilde{\beta}$ is the slope of the GEMM exposure–response function (dRR/dC), $\widetilde{iF}$ is the pseudo intake fraction giving the change in ambient concentration corresponding to a unit increase in emissions (dC/dE), $\widetilde{EF}$ is the emissions factor giving the increase in emissions per unit increase in vehicle activity, ε is the efficiency of the control strategy (E_1/E_2), and A is a measure of the vehicle activity ($VP \times VKT$). If we assume that Mo, ε, and A are well known and can be treated as constants and that $\tilde{\beta}$, $\widetilde{iF}$, and $\widetilde{EF}$ are uncertain and can be represented as lognormals with medians βo, iFo, and EFo and geometric SDs, $\sigma g\ \beta$, $\sigma g\ iF$, and $\sigma g\ EF$, then ΔM is lognormal with a median of $Mo \times \beta o \times iFo \times EFo \times \varepsilon \times A$ and a geometric SD of exp (sqrt ($\ln^2 (\sigma g\ \beta) + \ln^2 (\sigma g\ iF) + \ln^2 (\sigma g\ EF)$)).

Using the characterizations of uncertainty in each of the parameters given above, i.e., $\sigma g\ \beta \sim 1.05$ (consistent with CI 95 of roughly $\pm 10\%$), $\sigma g\ iF \sim 1.10$ ($\pm 20\%$), and $\sigma g\ EF \sim 1.18$ ($\pm 40\%$), we estimate $\sigma g\ \Delta M$ as 1.22 which would imply a CI 95 from roughly 2/3 to 3/2 of the central estimate. Applying these to the central estimate of a reduction (at the national level) of 460 thousand deaths attributable to air pollution (derived using the full model) we develop a rough CI 95 from 310 to 680 thousand deaths.

Limitations. This study and most population health risk analysis ignores the potential differential toxicity of $PM_{2.5}$ from various emission sources (58) and assumes the health impact of ambient $PM_{2.5}$ depends only on the magnitude of mass concentration. Recent research has suggested stronger associations between asthma morbidity and air pollutant exposures in areas with heavy traffic versus other pollution sources, which implies vehicular emissions may be more harmful to public health (59). If $PM_{2.5}$ from vehicular sources is more toxic per mass unit than average, this leads to underestimating the health benefits attributed to vehicle emissions controls.

The relatively coarse spatial resolution of this study could not capture the exposure to higher air pollution levels in some microenvironments of on-road and near-road conditions and might average out the spatial difference and lead to bias in our assessment (60). For example, the estimate of reduced $PM_{2.5}$ concentrations in central Beijing in this study is nearly 10 $\mu g/m^3$ higher at the spatial resolution of 4 km $\times$ 4 km compared to that at 36 km $\times$ 36 km. This also leads to a higher estimate of health benefit from vehicle emissions control for central Beijing based on 4 km $\times$ 4 km resolution.

In this study, we focused on the long-term effects of mitigating $PM_{2.5}$ and O_3 pollution and associated reduction in mortality attributable to air pollution due to the implementation of the control programs. We note that elevated ambient air pollution, notably $PM_{2.5}$ and NO_2, would also be expected to increase short-term (e.g., daily) morbidity and mortality (e.g., restricted activity days, missed days of school, new cases of asthma, exacerbation of respiratory symptoms, emergency department admission, hospitalizations, and deaths). Estimates of these impacts of short-term elevations in air pollution are not included in our analysis (see details in *SI Appendix, Note S3.1*). While typically the contributions of these acute effects to monetized estimates of damage are small in comparison with the monetized benefits of reducing chronic mortality attributable to air pollution, they may be of great concern to the public and to local policymakers and deserve to be considered carefully in any policy analysis.

Emission regulations might have affected vehicle fuel usage and the emissions along the fuel supply chain, which are not included in our estimation either. A comprehensive assessment of the costs and benefits of vehicle emission control measures in China, including mortality and morbidity from both acute and chronic exposures to air pollution, should be performed in the future. This would provide a more complete basis for policy decision making.

Data Availability. All study data are included in the article and *SI Appendix*.

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