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Cost–benefit assessment of proposed China 6 emission standard for new light-duty vehicles

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1. BACKGROUND

In May 2016, the Chinese Ministry of Environmental Protection (MEP) released a proposal, "Limits and Measurement Methods for Emissions from Light-Duty Vehicles (LDVs)," also known as the China 6 emission standard for new LDVs (referred to hereafter as the *proposed standard* or *China 6 standard proposal*; China MEP, 2016).[1] The proposed standard is a major upgrade to previous standards in China, with integration of best practices from the latest emission regulations in the United States, the State of California, and the European Union.

This paper estimates health benefits and technology upgrade costs of the proposed standard and implementation timetable. The research focuses mainly on nationwide impacts but also separately analyzes China's three key regions: the so-called Jing-Jin-Ji (or JJJ) region (agglomeration surrounding the capital, including Beijing, Tianjin, and Hebei province); the Yangtze River delta (or YRD) region (including Shanghai, Jiangsu, and Zhejiang provinces); and Guangdong province.

The proposed standard includes tightened emission limits of regulated ambient air pollutants, as well as provisions that enable better emission compliance of vehicles in real-world driving conditions.

To control exhaust emissions, the proposed standard introduces two phases of fuel-neutral emission limits of carbon monoxide (CO), hydrocarbon (HC), non-methane hydrocarbons (NMHC), nitrogen oxides (NO_x), particulate matter (PM), and particulate number (PN). Figure 1 compares the emission limits (on a per-vehicle maximum limit basis) for non-methane organic gases (NMOG)+NO_x and PM from diesel and gasoline passenger cars (M1 category) under the Euro 6, US Tier 2, California LEV III, and the proposed Phases A and B of China 6 LDV emission standards. According to the proposed standard, China 6a and China 6b will be implemented nationwide in 2020 and 2023, respectively.

The proposed China 6 standard and the Euro 6 standard both adopt the Worldwide Harmonized Light Vehicles Test Cycle and Procedure (WLTC/P), which allows an apples-to-apples comparison of the stringency of both standards. Compared with Euro 6 standard requirements, China 6a tightens the CO limit for gasoline vehicles by 50%–67% depending on vehicle class; tightens the NO_x limit for diesel vehicles by 25%–33% depending on vehicle class; and tightens the PM limit for both gasoline and diesel vehicles by 10%. China 6b further tightens CO, NO_x, and PM limits by 50%, 40%, and 33%, respectively, compared with China 6a requirements (Figure 1).

1 The final rule of China 6 emission standard for new LDVs was released by China MEP on December 23, 2016, which will be implemented starting on July 1, 2020. Retrieved from http://www.mep.gov.cn/gkml/hbb/bgg/201612/t20161223_369497.htm

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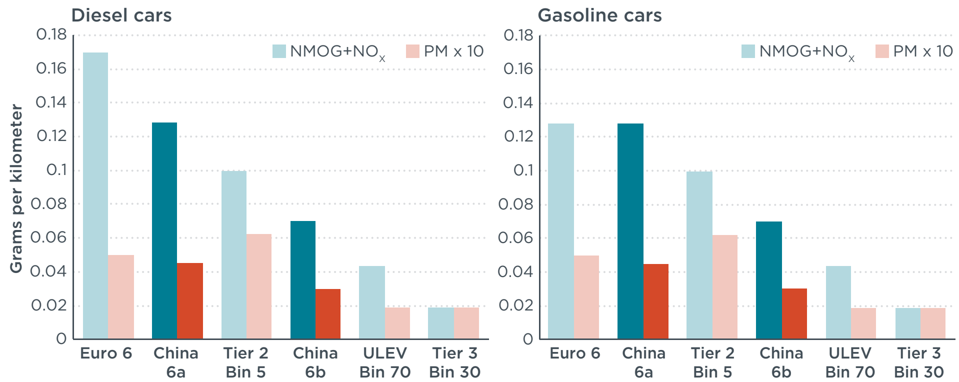
To control evaporative emissions, the proposed standard tightens the emission limit from 2 g under the Euro 6 requirement to 0.70 g per test using an innovative 2-day diurnal test procedure that better reflects the real-world operating condition compared with the European counterpart. Given the dramatically different test cycle and procedures, it is extremely difficult to provide an apples-to-apples comparison of the evaporative emission standards under various regulatory programs. Table 1 summarizes the current and proposed emission limits associated with different testing procedures in four markets—Europe, the United States, California, and China (proposed).

In addition, the China 6 proposal adopts a more dynamic/realistic Worldwide Harmonized Light Vehicles Test Cycle (WLTC), a Real-World Driving Emissions (RDE) testing requirement, and a stronger On-Board Diagnostics (OBD) provision similar to that adopted in California. These requirements play a critical role in strengthening in-use emission compliance of vehicles and contribute to emissions reductions during real-world driving.

We considered the above improvements when assessing the benefits of the China 6 standard proposal regarding emissions reduction. To model the potential impacts of the RDE and OBD provisions, we assumed a higher vehicle emission compliance ratio (that is, the share of new vehicles for a given production year that comply with the emission standards for the useful life among the entire new vehicle fleet of that year) compared with previous emission standards in China. To determine compliance ratios, we looked at in-use vehicle emission testing data, examined data from vehicle inspection and maintenance (I/M) programs in China, and consulted with the Vehicle Emission Control Center (VECC) of MEP. Detailed modeling scenarios are provided in Section 2.1.1 of this paper.

The cost–benefit assessment presented later in the paper provides modeled emission reductions of regulated ambient air pollutants; changes to $PM_{2.5}$ and ground-level ozone concentrations; and avoided health impacts, including premature deaths and hospitalization, attributable to implementation of the proposed standard. Social benefits are quantified by estimating the economic value of reductions in premature deaths and hospitalization attributable to improved urban air quality. Finally, the social value of public health benefits is compared with the costs associated with upgrading to cleaner vehicle technology.



1. Emissions limits are those for Type I test (regular temperature, cold start emission test).
2. For diesel LDVs, Europe and China regulate HC and NO_x, instead of NMOG+NO_x.
3. For gasoline LDVs, Europe and China regulate NMHC and NO_x, instead of NMOG+NO_x.
4. This analysis simply compares direct emission limits and does not take into consideration the differences in test cycles and procedures among various regulatory programs.

Figure 1. Comparison of NMOG+NO_x and PM emission limits for passenger cars under Euro 6, US Tier 2, California LEV III, and the proposed China 6 LDV emission standards.

Table 1. Comparison of evaporative emission standards for passenger cars under Euro 6, US Tier 2 (enhanced), California LEV III, and the proposed China 6 LDV emission standards (g/test).

Test procedure	Euro 6*	US Tier 2	LEV III	CHINA 6 A/B
24-hour diurnal + hot soak	2.0	/	/	/
48-hour diurnal + hot soak	/	0.65	0.35	0.70
72-hour diurnal + hot soak	/	0.50	0.30	/

*Changing from a 24-hour to a 48-hour diurnal test is under discussion in the Euro 6c proposal.

2. HEALTH BENEFIT EVALUATION

2.1 METHODOLOGY

2.1.1 Technology Roadmap

This analysis focuses on on-road LDVs in China, including vehicles fueled by gasoline, diesel, liquefied petroleum gas (LPG), and natural gas in the categories of M1, M2, N1, and N2, not exceeding 2,610 kg of vehicle reference mass for type approval, as well as the same categories of vehicles not exceeding 2,840 kg of reference mass for extended type approval. To evaluate the impacts of the proposed standard for LDVs on emissions, air quality, and human health, we examined two scenarios:

1. **Business As Usual (BAU)**: The BAU scenario maintains China 5 LDV emission standards nationwide and in all major cities throughout the study period (2016–2030).

2. **China 6**: This scenario assumes that China 6 LDV emission standards are implemented first in key regions, followed by nationwide implementation, as described in Table 2.

The three key regions (i.e., JJJ, YRD, and Guangdong) are expected to implement China 6 emission standards ahead of the national timeline based on previous patterns and on the State Council's air-quality improvement plan. Table 2 provides the implementation timeline assumed under the China 6 scenario. The three keys regions will skip China 6a and go straight to China 6b, while the rest of China will implement China 6a beginning on January 1, 2020, and upgrade to China 6b 3 years later. Table 3 specifies emissions limits under the China 6a and 6b standards.

For each scenario, we used a three-step approach to evaluate the health impacts resulting from $PM_{2.5}$ and ozone exposure, as described conceptually in Figure 2. Step 1 models the emission amounts of regulated pollutants from vehicles and other emission sources using various emission inventory models. Step 2 translates the emission amounts into ambient $PM_{2.5}$ and ozone concentrations using a chemical transport model. Step 3 estimates the health impacts in terms of morbidities and premature deaths by applying exposure-response functions for specific health outcomes in combination with demographic and health incidence data. The following subsections will elaborate the methodology and results for each step. Comparing the modeled health impacts results of the BAU and China 6 scenarios can help to determine the health benefits of the proposed standard.

Table 2. Implementation dates of China 6 LDV emission standards modeled under the China 6 scenario.

Region	Emission limits applied (for Type I test)	Implementation date
Beijing	China 6b	January 1, 2018
Shanghai	China 6b	April 1, 2018
Rest of JJJ, rest of YRD, and Guangdong	China 6b	July 1, 2018
Nationwide	China 6a	January 1, 2020
Nationwide	China 6b	January 1, 2023

Table 3. Emissions limits under the proposed China 6a and 6b standards for the Type I test.

Vehicle Type/ Test weight class		CO (g/km)	THC (g/km)	NMHC (g/km)	NO_x (g/km)	N_2O (g/km)	PM (g/km)	PN (#/km)
China 6a								
I	All	0.5	0.1	0.068	0.06	0.128	0.0045	6×10^{11}
II	~1305	0.5	0.1	0.068	0.06	0.128	0.0045	6×10^{11}
	1305~1760	0.63	0.13	0.09	0.075	0.165	0.0045	6×10^{11}
	>1760	0.74	0.16	0.108	0.082	0.19	0.0045	6×10^{11}
China 6b								
I	All	0.5	0.05	0.035	0.035	0.07	0.003	6×10^{11}
II	~1305	0.5	0.05	0.035	0.035	0.07	0.003	6×10^{11}
	1305~1760	0.63	0.065	0.045	0.045	0.09	0.003	6×10^{11}
	>1760	0.74	0.08	0.055	0.05	0.105	0.003	6×10^{11}

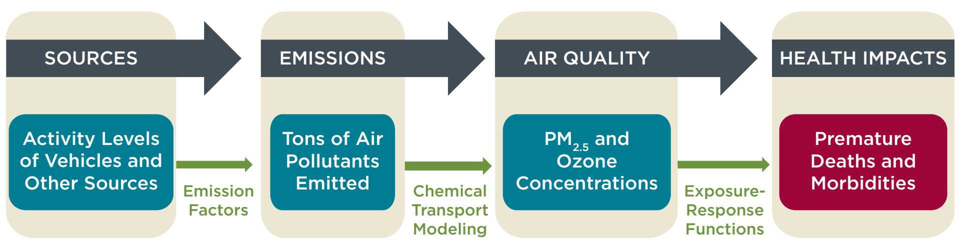


Figure 2. Conceptual framework for evaluating the health impacts attributable to $PM_{2.5}$ and ozone exposure.

2.1.2 Emissions Modeling Methodology

We used the China Mobile Source Emission Inventory Model (Façanha, Blumberg, & Miller, 2012; Shao & Wagner, 2015; Yang, Wang, Shao, & Muncrief, 2015) to calculate tailpipe emissions from LDVs for the two scenarios considered. By applying localized input parameters related to vehicle technology, annual vehicle sales, vehicle survival rates, vehicle kilometers traveled (VKT), efficiency, and fuel share, the model provides the tailpipe emissions of seven conventional air pollutants (i.e., CO, SO_2, NO_X, HC, $PM_{2.5}$, BC, and OC) emitted from LDVs.

The China Mobile Source Emission Inventory Model is under consistent improvement to ensure that it can best capture the policy efforts and fleet emissions in China. The model was updated for this analysis with a new set of emission factors, which were developed with inputs from real-world testing conducted by Tsinghua University and VECC and are recommended by MEP. In-use compliance was also modeled, with improvements assumed under more stringent standards. The modeling scenarios used the percentages in Table 4 to describe the share of vehicles using the emissions factors described above. For the remaining vehicles (15% for the earliest emissions standards and dropping to just 2% for China 6), our modeling assumed that these vehicles were high emitters. High emitters with China 0-3 control technologies were assigned the same emission factors as uncontrolled vehicles. High emitters with China 4-6 control technology were assigned the same emission factors as China 1 vehicles.

Table 4. In-use compliance rates of vehicles with different control technologies modeled under both scenarios.

	CHINA 0-3	China 4-5	China 6
Compliance ratio (%)	85	90	98

We evaluated evaporative emissions from LDVs using an external model developed by Ingevity Corporation with inputs from the Motor Vehicle Emission Simulator (MOVES) 2014 (U.S. Environmental Protection Agency [U.S. EPA], 2014). Ingevity's model calculates evaporative HC emissions during refueling and the rest of vehicle use (including diurnal, hot soak, permeation, and running losses) using information on Chinese fuels characteristics, ambient temperatures, latitudes, and altitudes. Evaporative emission factors for China are described in Table 5. We applied emissions factors to a simple fleet turnover model using the same vehicle inputs and assumptions developed for the China Mobile Source Emission Inventory Model.

Table 5. Evaporative emissions factors for China.

Scenario	Refueling	Other Vehicle Use
BAU	0.089 g/km	0.446 g/km
China 6	0.002 g/km	0.018 g/km

To fulfill the data requirements of air-quality modeling, we also estimated emissions from other types of vehicles as well as non-transport emission sources. Among them, we derived emissions from natural sources from the Model of Emissions of Gases and Aerosols from Nature (MEGAN; Guenther et al., 2006), while we calculated emissions from anthropogenic sources using the Multi-Resolution Emission Inventory for China (MEIC; He, 2012). Specifically, for anthropogenic sources, we assumed that a variety of policies would be implemented as required under the 2013 Air Pollution Prevention and Control Action Plan (China State Council, 2013). These policies include accelerated utilization of clean energy, strict control over growth of high-polluting and high-energy-consuming industries, improvements in productivity, and reductions in excess capacity. End-of-pipe measures include desulfurization facilities at all coal-fired power plants, sintering machine and pellet production facilities of iron-steel manufacturing centers, and catalytic cracking units at oil refineries and non-ferrous metal smelting facilities. Coal-fired boilers greater than 20 tons per hour (t/h) are assumed to install desulfurization facilities. Additional assumptions were that denitrification facilities will be installed at all coal-fired power units, except at circulating fluidized boilers; new dry-cement kilns will install NO_x combustion technology and denitrification facilities; dust-removal facilities will be upgraded at coal-fired boilers and industrial furnaces; and VOC will be comprehensively controlled at petrochemical, chemical, paint, packaging, and printing industries, alongside vapor recovery at oil-storage facilities and service stations and promotion and use of water-based paints and low-volatile solvents. To analyze the impacts of emission reduction of LDVs on air quality and human health due to the implementation of China 6 emission standards, emissions from other vehicle types and non-transport emission sources were held constant in the BAU and China 6 scenarios.

2.1.3 Air-Quality Modeling Methodology

Based on the emission amounts calculated in Section 2.1.2, we conducted chemical transport model simulation to estimate annual average $PM_{2.5}$ (both primary and secondary formation) concentrations and seasonal (April–September) average daily 1-hour peak ozone concentrations in 2030 for the two scenarios considered.

As a new generation meso-scale numerical weather prediction system designed to serve a wide range of meteorological applications, the offline-coupled Weather Research and Forecasting (WRF) model (Michalakes et al., 2001) has been widely used to provide meteorological input fields necessary for chemical transport models. In this analysis, we applied WRF v3.5.1 to generate meteorological fields at a 36-km horizontal grid resolution with 23 vertical layers. Initial and boundary conditions were taken from the U.S. National Centers for Environmental Prediction Final Analysis (NCEP-FNL)[2]. Land use/land cover and topographical data were taken from a default WRF input dataset.

As a 3-D Eulerian atmospheric chemistry and transport modeling system designed to simulate multiple pollutants from city to trans-continental scales, the Community Multi-Scale Air Quality (CMAQ) Modeling System (Byun, Ching, Novak, & Young, 1998) has been widely used to simulate ambient $PM_{2.5}$ and ozone concentrations. This study used CMAQ v5.0.1 released in July 2012. This version of CMAQ model contains an updated carbon bond gas-phase mechanism with new toluene chemistry, a new aerosol module (AERO6), and ISORROPIA v2.1 inorganic chemistry functionality (Fountoukis & Nenes, 2007). Vertically, 14 layers are included from surface to the tropopause (approximately 16 km). The first layer is 38 m high.

The modeling domain in this analysis covered all of China with a grid resolution of 36 km x 36 km. For $PM_{2.5}$, we ran simulations for January, April, July, and October, respectively, with 2 weeks of spin-up time each. For ozone, we ran simulations from April to September, also with 2 weeks of spin-up each time. Then, we obtained annual average $PM_{2.5}$ and seasonal (April to September) average daily 1-hour peak ozone concentrations for each grid in the modeling domain by computing the arithmetic averages of the values in these four (for $PM_{2.5}$) or six (for ozone) representative months.

2 The NCEP FNL data are on 1-degree by 1-degree grids prepared operationally every six hours. Retrieved from https://rda.ucar.edu/datasets/ds083.2/

2.1.4 Health Modeling Methodology

Exposures to $PM_{2.5}$ and ozone pollution will result in both acute (short-term) and chronic (long-term) health impacts. This study focused on the latter because long-term health impacts are more inclusive and typically account for short-term impacts. For each scenario, we quantified the long-term morbidities and premature deaths caused by $PM_{2.5}$ and ozone exposures in 2030, based on the gridded pollutant concentration values estimated in Section 2.1.3.

We applied the GBD 2010 integrated exposure-response (IER) functions developed by Burnett, Pope, Ezzati, Olives, Lim, & Mehta (2014) to estimate the gridded premature deaths of four major diseases attributable to $PM_{2.5}$ pollution, including ischemic heart disease (IHD), stroke, chronic obstructive pulmonary disease (COPD), and lung cancer. For both scenarios, we first calculated the relative risk (RR) of mortality of each disease caused by $PM_{2.5}$ exposure using Equation 1. The four diseases share the same mathematical function; however, the values of C_0, α, γ, and δ for different diseases vary, as shown in Table 6. Second, based on Equation 2, we converted the calculated RR value to attributable fraction (AF), which represents the percentage of baseline death rate owning to $PM_{2.5}$ exposure. Last, we estimated the $PM_{2.5}$-caused premature deaths (PD) of each disease using Equation 3. The baseline death rates of the four diseases considered are listed in Table 6. The 2010 population data was taken from the LandScan global population database.

$$RR(C) = \begin{cases} 1+\alpha\left(1-e^{-\gamma(C-C_0)^{\delta}}\right), & C > C_0 \\ 1, & else \end{cases}$$

(Equation 1)

$$AF = \frac{RR-1}{RR}$$

(Equation 2)

$$PD = AF \times BDR \times P$$

(Equation 3)

Where:

- C = annual average $PM_{2.5}$ concentrations in the target grid (μg/m³);
- C_0 = low-concentration threshold (LCT) below which $PM_{2.5}$ has no effect on mortality (μg/m³);
- α, γ, and δ = parameters used to determine the shape of the concentration-response curve;
- BDR = baseline death rate of disease in China; and
- P = exposed population in the target grid.

The cardiovascular and respiratory hospitalizations attributable to $PM_{2.5}$ exposure were estimated based on the log-linear exposure-response function used in Europe-HRAPIE project (World Health Organization [WHO], 2013). For both scenarios, we first calculated the relative risk (RR) of cardiovascular and respiratory hospitalization due to $PM_{2.5}$ exposure using Equation 4. The β values for the

Table 6. Values of key parameters used to estimate the premature deaths of four major diseases due to $PM_{2.5}$ exposure.

Disease	α	γ	δ	C_0	BDR
IHD	0.843	0.0724	0.544	6.96	0.000707
Stroke	1.01	0.0164	1.14	8.38	0.00129
COPD	18.3	0.000932	0.682	7.17	0.000696
LC	159	0.000119	0.735	7.24	0.000383

two diseases considered are shown in Table 7. Second, based on Equation 2, we converted the calculated RR value to attributable fraction (AF), which represents the percentage of baseline hospitalization rate owning to $PM_{2.5}$ exposure. Last, we estimated the $PM_{2.5}$-caused hospitalizations (H) of each disease using Equation 5. The baseline hospitalization rates of the two diseases considered are listed in Table 7. The 2010 population data was also taken from the LandScan global population database.

$$RR(C) = e^{\beta C}$$

(Equation 4)

$$H = AF \times BHR \times P$$

(Equation 5)

Where:

- C = annual average $PM_{2.5}$ concentrations in the target grid (μg/m³),
- β = parameter used to determine the shape of the concentration-response curve,
- BHR =baseline hospitalization rate of disease in China, and
- P = exposed population in the target grid.

Table 7. Values of key parameters used to estimate the cardiovascular and respiratory hospitalizations due to $PM_{2.5}$ exposure.

Disease	β	*BHR*
Cardiovascular diseases	0.0009059	0.00797
Respiratory diseases	0.001882	0.00325

To estimate the premature deaths from respiratory diseases attributable to ozone exposure, we applied the log-linear function developed by Jerrett et al. (2009). For both scenarios, we first used Equation 6 to evaluate the relative risk of death from respiratory diseases due to ozone exposure. Then, we inserted the calculated RR values into Equations 2 and 3 to estimate the premature deaths caused by exposure to ozone pollution. In this analysis, ξ, C_0, and baseline death rate of respiratory diseases were set as 0.0017, 33.3 ppb, and 0.000696, respectively.

$$RR(C) = e^{\xi (C - C_0)}$$

(Equation 6)

Where:

- C = seasonal average daily 1-hour peak ozone concentrations in the target grid (ppb),
- C_0 = low-concentration threshold (LCT) below which ozone has no effect on mortality (ppb), and
- ξ = parameter used to determine the shape of the concentration–response curve.

We estimated respiratory hospitalizations due to ozone exposure using the log-linear function developed by Orru et al. (2013). For both scenarios, we first calculated the relative risk of respiratory hospitalization caused by ozone exposure using Equation 7. Then, we inserted the calculated RR values into Equations 2 and 5 to evaluate the respiratory hospitalizations attributable to ozone exposure. The ξ, C_0, and baseline death rate of respiratory diseases were set as 0.00026, 35 ppb, and 0.00325, respectively.

$$RR(C) = e^{\tau (C - C_0)}$$

Where:

- C = seasonal average daily 1-hour peak ozone concentrations in the target grid (ppb),
- C_0 = low-concentration threshold (LCT) below which ozone has no effect on morbidity (ppb), and
- τ = parameter used to determine the shape of the concentration-response curve.

By comparing the health impacts modeling results of both scenarios, we finally obtained the health benefits (i.e., morbidities and premature deaths avoided) attributable to the proposed China 6 standards.

2.2 MODELING RESULTS

2.2.1 Emissions Impacts

Table 8 provides the projected reductions (absolute and percent) of six conventional air pollutants from LDVs in China and three key regions in 2030 with implementation of China 6. It is clear that emissions of all of the air pollutants are greatly

Table 8. Projected emission reduction from LDV sector in 2030.

	CO	NO_x	$PM_{2.5}$	HC			BC	OC
				Tailpipe	Evap.	Total		
Emission Reduction (thousand tons)								
China	3,396	1,001	25.92	550	3,633	4,184	5.62	2.00
JJJ	431	107	3.15	70.16	477	547	0.56	0.21
YRD	705	202	5.42	117	746	863	1.11	0.40
Guangdong	401	172	3.59	70.86	341	412	1.04	0.33
Percentage Reduction (%)								
China	42	74	54	59	90	84	65	34
JJJ	43	76	56	62	90	85	63	31
YRD	43	77	57	62	90	85	66	34
Guangdong	42	79	61	63	90	84	73	43

reduced by introducing China 6 LDV emission standards within the timeline considered in this analysis. The emissions reductions for NO_X and HC, two important gaseous precursors of both $PM_{2.5}$ and ozone, reach 74% and 84%, respectively, at the national level. The evaporative HC emissions are controlled most effectively, with a reduction rate of 90% at the national level, due to the implementation of evaporation-related control strategies proposed under China 6 vehicle emission standards. Figures 3 to 6 depict the annual emissions of $PM_{2.5}$ and NO_X from LDVs under BAU and China 6 scenarios from 2010 to 2030, at both national and sub-national levels.

Table 9 provides the projected emissions of six conventional air pollutants from four major economic sectors in China under BAU and China 6 scenarios. As a result of implementation of the proposed China 6 standards, the economy-wide emissions of CO, NO_X, $PM_{2.5}$, HC, BC, and OC in 2030 decrease by 3.0%, 6.0%, 0.6%, 17.3%, 0.9%, and 0.2%, respectively.

2.2.2 Air-Quality Impacts

In 2015, the estimated annual population-weighted average concentration of $PM_{2.5}$ nationwide was 50.2 μg/m³ (or 16.2 μg/m³ without population weighting). Among three key regions, the JJJ region generated the highest concentrations of $PM_{2.5}$ (68.3 μg/m³), followed by the YRD region (65.4 μg/m³) and Guangdong (25.2 μg/m³). These simulation results are in agreement with air-quality monitoring data (Greenpeace, 2015) showing that JJJ and YRD were not in compliance with national Class 2 air-quality standards (35 μg/m³) for annual $PM_{2.5}$ (China MEP, 2012) in 2015. These numbers also demonstrate that the national population was exposed on average to a level of $PM_{2.5}$ that exceeded national Class 2 standards.

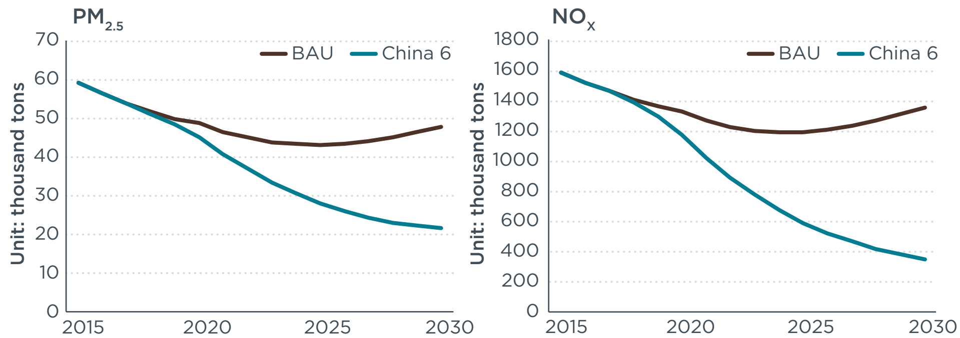


Figure 3. Emission modeling results for $PM_{2.5}$ and NO_X at the national level.

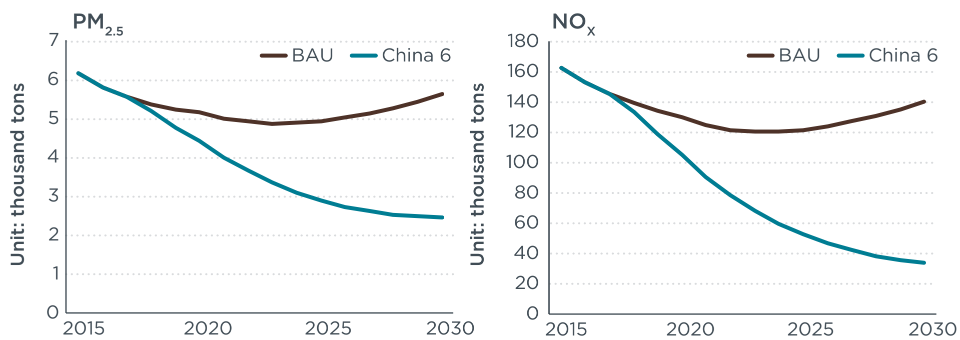


Figure 4. Emission modeling results for $PM_{2.5}$ and NO_X in the Jing-Jin-Ji region.

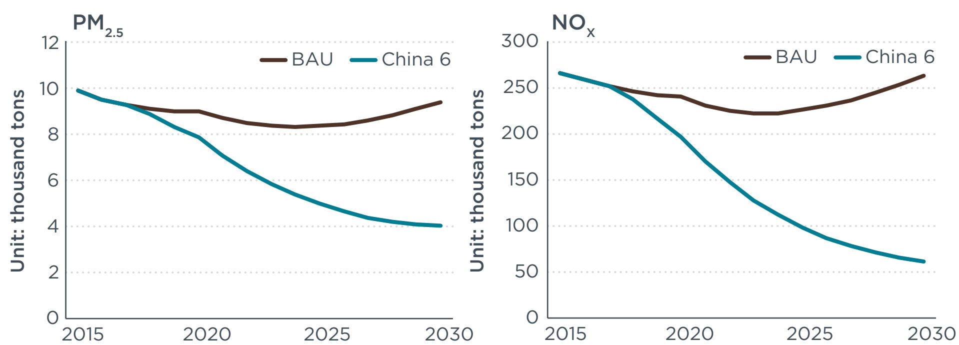


Figure 5. Emission modeling results for $PM_{2.5}$ and NO_X in the YRD region.

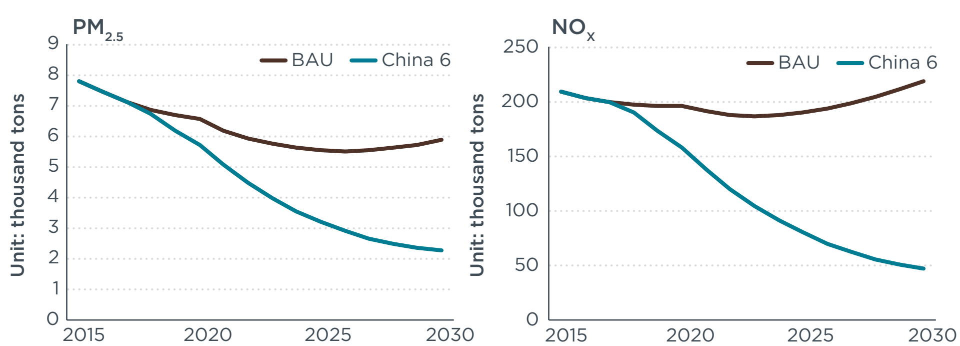


Figure 6. Emission modeling results for $PM_{2.5}$ and NO_X in the Guangdong region.

This study did not generate 8-hour ozone concentration data, so we are unable to compare ozone air-quality status in 2015 against national air-quality standards for ozone.

Under the BAU scenario in 2030, reductions are expected in population-weighted average exposure to both $PM_{2.5}$ and ozone. At the national scale, this study projects a 40% reduction in exposure to annual average $PM_{2.5}$ concentrations and a smaller 6% reduction in ozone exposure. Of the three key regions, the largest reductions in $PM_{2.5}$ exposure will occur in YRD (39%), followed by JJJ (36%) and Guangdong (32%). A 5% reduction in ozone exposure will occur in both JJJ and YRD, but no reduction in exposure is projected in Guangdong. These reductions reflect a BAU scenario that assumes the successful reduction of air pollutants to meet State Council mandates for air pollution control nationwide and in key regions with an emphasis on power and industrial sector emissions (also refer to the description in Section 2.1.2, Emissions Modeling Methodology). By 2030, this study assumes that JJJ and YRD will be in compliance with Class 2 national ambient air-quality standards for $PM_{2.5}$ even without adoption of the China 6 standards under a BAU scenario. National population-weighed exposure will also fall below national Class 2 standards. Both at the national level and within key regions, levels of $PM_{2.5}$ exposure will still exceed Class 1 standards, which are nearly in agreement with U.S. EPA and WHO air-quality guidelines but are currently applied in China only in special regions, such as national parks.

A China 6 standard will ensure compliance with national Class 2 air-quality standards and move national air quality toward Class 1 levels. As a result of implementation of China 6, concentrations of both $PM_{2.5}$ and ozone will decline from the 2030 BAU scenario (see Table 10). At the national level, average population-weighted exposure to annual average $PM_{2.5}$ will fall by 1.1 μg/m³, a reduction of 3.5% from the BAU scenario. Larger reductions will occur in JJJ and YRD, whereas smaller reductions will occur in Guangdong. Similarly, the seasonal average daily 1-hour average peak ozone concentration will fall nationwide by 2.1 ppb (population-weighted), equal to a decline of 3.3% from BAU. Larger reductions will occur in JJJ and YRD, whereas small reductions will occur in Guangdong.

Table 9. 2030 modeled emissions for all sectors in China (thousand tons).

	CO	NO_x	$PM_{2.5}$	HC	BC	OC
BAU						
Power	3,890	3,400	333	103	1	0.3
Residential	35,980	554	1,060	2,380	206	628
Transportation	16,730	5,770	210	7,530	286	93
Industry	55,740	6,950	2,490	14,230	127	114
Economy-wide	112,350	16,670	4,090	24,240	619	834
China 6						
Power	3,890	3,400	333	103	1	0.3
Residential	35,980	554	1,060	2,380	206	628
Transportation	13,334	4,769	184	3,346	280	91
Industry	55,740	6,950	2,490	14,230	127	114
Economy-wide	108,960	15,680	4,070	20,050	613	832
Reduction from BAU to China 6						
Thousand Tons	3,396	1,001	25.9	4,184	5.6	2.0
% Reduction	3.0	6.0	0.6	17.3	0.9	0.2

Table 10. Effect of China 6 on $PM_{2.5}$ and ozone concentrations in 2030 in relation to the BAU scenario.*

	Annual Average $PM_{2.5}$		Seasonal average daily 1-hOUr Peak Ozone	
	μ g/m³	% Change	ppb	% Change
JJJ	−2.0	−4.6%	-3.7	−5.7%
YRD	−1.6	−3.9%	-3.0	−4.5%
Guangdong	−0.4	−2.5%	−2.0	−3.0%
China	−1.1	−3.5%	−2.1	−3.3%

*Concentration values are population-weighted.

2.2.3 Health Benefits

According to our analysis, in 2030, implementation of China 6 will avoid more than 21,700 premature deaths and more than 28,500 hospitalizations nationwide compared to the BAU scenario, as a result of reduced exposure to both $PM_{2.5}$ and ozone. Most of these avoided health impacts will be attributable to reductions in exposure to ambient $PM_{2.5}$ concentrations. Reductions in ozone exposure will account for approximately 24% of avoided premature deaths and 14% of avoided morbidities. Table 11 provides avoided premature deaths and morbidities for $PM_{2.5}$ and ozone in China and in three key regions.

At the provincial level, the largest absolute reductions in premature deaths will occur in the Henan, Shandong, Hebei, Jiangsu, Anhui, Hubei, and Sichuan provinces, where more than 1,000 annual premature deaths may be avoided in 2030. The greatest relative reductions from BAU (on a percentage basis) may occur in the northeastern provinces, including Heilongjiang, Liaoning, and Jilin (Figure 7).

Compared to a 2030 BAU emissions scenario for all sources (including power, industry, and residential), implementation of China 6 will reduce total premature deaths from $PM_{2.5}$ by 2% and from ozone by 5% nationwide. The percentage of reductions in $PM_{2.5}$-related premature deaths in three key regions—JJJ, YRD, and Guangdong—will be equivalent (2%). Ozone-related premature deaths will be reduced by 9% in JJJ, 6% in YRD, and 4% in Guangdong. Figure 8 shows expected premature deaths from $PM_{2.5}$ and ozone in China under a China 6 scenario.

Table 11. Avoided health impacts from China 6 implementation in 2030 relative to BAU.

	Avoided premature deaths			Avoided hospital admissions		
	$PM_{2.5}$	O_3	Total	$PM_{2.5}$	O_3	Total
JJJ	1,761	981	2,742	7,295	717	8,012
YRD	1,809	595	2,404	2,160	426	2,586
Guangdong	669	211	880	471	155	626
China	16,386	5,368	21,754	24,559	4,000	28,559

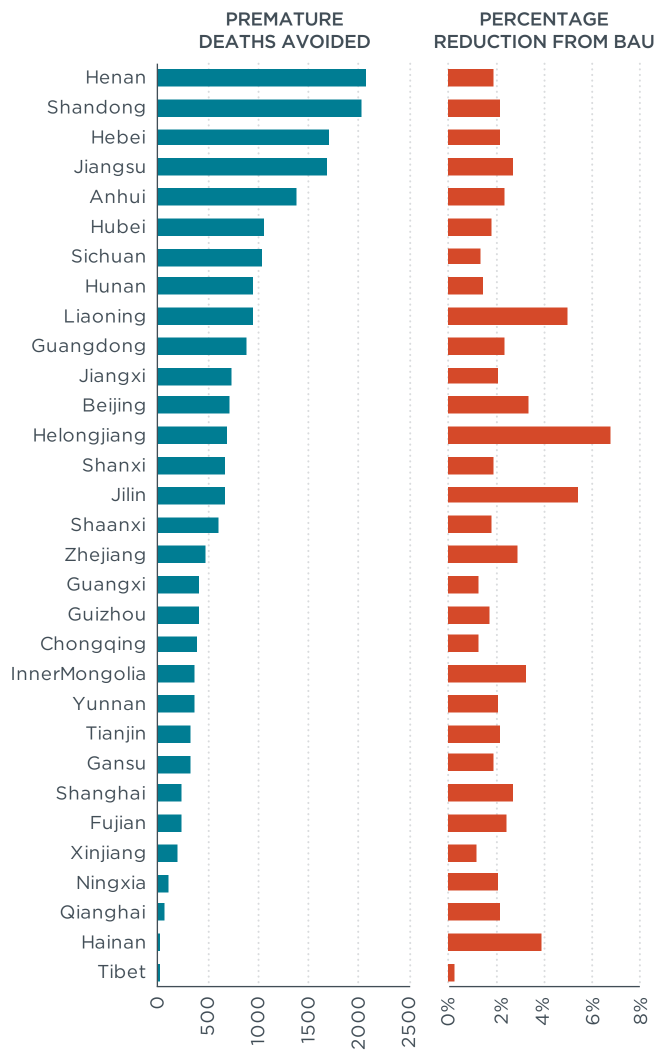


Figure 7. Avoided premature deaths and percentage reduction from BAU by province in 2030.

3. TECHNOLOGY COST ASSESSMENT

Posada, Bandivadekar, & German (2012) estimated the costs associated with the introduction of advanced vehicle emission-control technologies required to meet the more stringent emission standards for LDVs. For this current cost–benefit analysis of China 6 implementation, the costs of technologies already in the market were updated and cost items relevant to the requirements specified by China 6a and China 6b were added.

Cost estimates were developed for gasoline and diesel vehicles. Gasoline engines power 98% of passenger vehicles and almost half of the light commercial vehicle fleet in China. For gasoline vehicles, the assessment focused on estimating the cost of catalysts required by three-way catalysts, the cost of gasoline particle filters, and the cost of evaporative emission-control technologies. For diesel vehicles, the assessment focused on the cost of selective

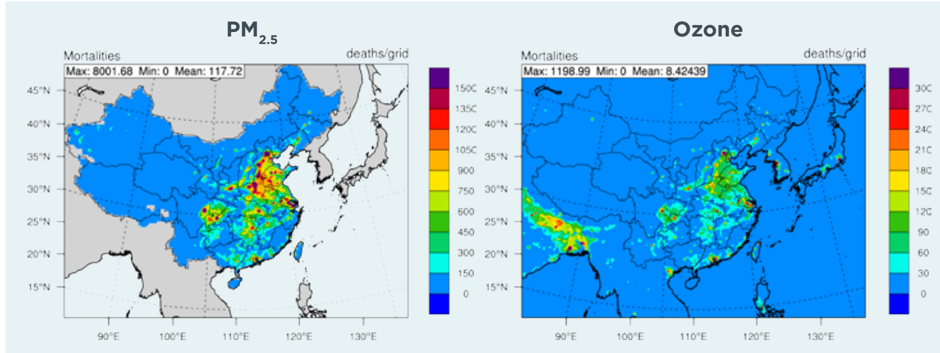


Figure 8. $PM_{2.5}$ and ozone-related premature deaths under a China 6 scenario.

Table 12. Incremental cost to comply with the proposed China 6 standard based on an average car (@1.7-L engine).*

	China 5 to 6a		China 5 to 6b	
	PV	**LCV**	**PV**	**LCV**
Gasoline	$131	$123	$138	$128
Diesel	$648	$678	$699	$729

*Costs reflect the incremental cost to manufacturers, not the price increment paid by the consumer.

catalytic reduction (SCR) systems required for NO_x control to meet the stringent Euro 6a and 6b proposed standards and the OBD requirements.

To evaluate the incremental costs of China 6 vehicle and emission-control technology, we updated the cost-modeling methodology developed by Posada et al. (2012) with information from recent regulatory documents and peer-reviewed journal articles. Regulatory impact assessments used for this cost assessment include the impact assessment developed for the U.S. EPA Tier 3 regulation and the California Air Resources Board (CARB) impact assessment for LEV III regulations. Costs for evaporative emissions control technologies were based on previous U.S. EPA rulemaking support documents with costs updated to the end of 2015. More than a dozen peer-reviewed papers published in the last 5 years were summarized to get an updated version of the costs incurred by these emission regulations; a list is presented in Annex A.

Table 12 summarizes the estimated vehicle and engine costs assuming an average vehicle with a 1.7-liter engine. The compliance cost for gasoline vehicles is much lower than for diesel vehicles. Most of the cost for gasoline vehicle compliance comes from the adoption of gasoline particulate filters for direct injected engines[3] (called *gasoline direct injection*, or GDI) required to meet the PN targets and from evaporative emission control technology. The literature search concluded that a very small fraction of the cost is incurred by additional catalysts required to reach the most stringent NO_x and NMHC targets. For diesel vehicles, 67% of compliance costs (approximately $430) are due to SCR technologies, which are needed to meet the NO_x standards for diesel vehicles; the remainder are incurred by changes to the air-fuel management system, OBD requirements, and engine calibrations.

Incremental costs are multiplied by the number of projected vehicles sold for each vehicle and fuel type in order to estimate the aggregate cost to manufacturers of meeting the China 6 standards. The incremental costs borne in transitioning to the proposed China 6 standard are very reasonable and the benefits are significant, as shown in Figure 9.

4. COMBINED ANALYSIS: COST–BENEFIT RATIOS

Reducing vehicle pollutant emissions yields corresponding improvements in ambient air quality, which has broad positive effects on the environment and public health. This analysis compares the incremental technology costs of complying with the China 6 standard with the economic value of avoided premature death in a single year (2030).

4.1 METHODOLOGY

For costs, as previously mentioned, we mainly assessed the manufacturer costs in vehicle technology upgrades for complying with the proposed China 6 standard. However, we did not include fuel cost in this analysis. Vehicle technology costs were calculated by estimating the per-vehicle incremental cost of technology needed to meet China 6 compared with China 5, multiplying these incremental costs by the number of vehicles sold in each calendar year. Annual cost reduction due to technology learning and increased production volume is assumed to be 3% from 2015–2020, 2% from 2020–2025, and 1% from 2025–2030 based on CARB estimates.

The health benefits stemming from the introduction of advanced vehicle

3 This study assumes that GDI technology will cover the entire gasoline passenger car fleet in 2020 and beyond. This may lead to an overestimation of the technology cost.

emissions are quantified in terms of the economic value of avoided premature deaths (not including avoided hospitalization), both from lower exposure to ambient $PM_{2.5}$ and ozone concentrations. The economic benefits of reductions in premature mortality are calculated based on *value of statistical life* (VSL), an indicator of willingness to pay to avoid health impacts.[4] This analysis did not monetize the benefits from reduced morbidity; therefore, our results should be considered conservative.

Ideally, estimates of VSL should be based on local empirical studies that reflect a combination of stated preference and revealed preference methods; however, in countries where sufficient empirical data are not available, estimates can be adjusted from other countries using a "benefit-transfer" approach (Minjares et al., 2014). In the absence of sufficient empirical evidence in China, we applied the benefit-transfer approach as described in Miller, Blumberg, & Sharpe (2014). The key assumption of this approach is that differences in per capita income are the most important determinants of differences in willingness to pay for mortality risk reduction between populations. For analyses of environmental policies in the United States, the U.S. EPA recommends using a central VSL estimate of $7.4 million (2006 USD) adjusted to the year of analysis. The corresponding value in 2015 USD is $8.75 million.

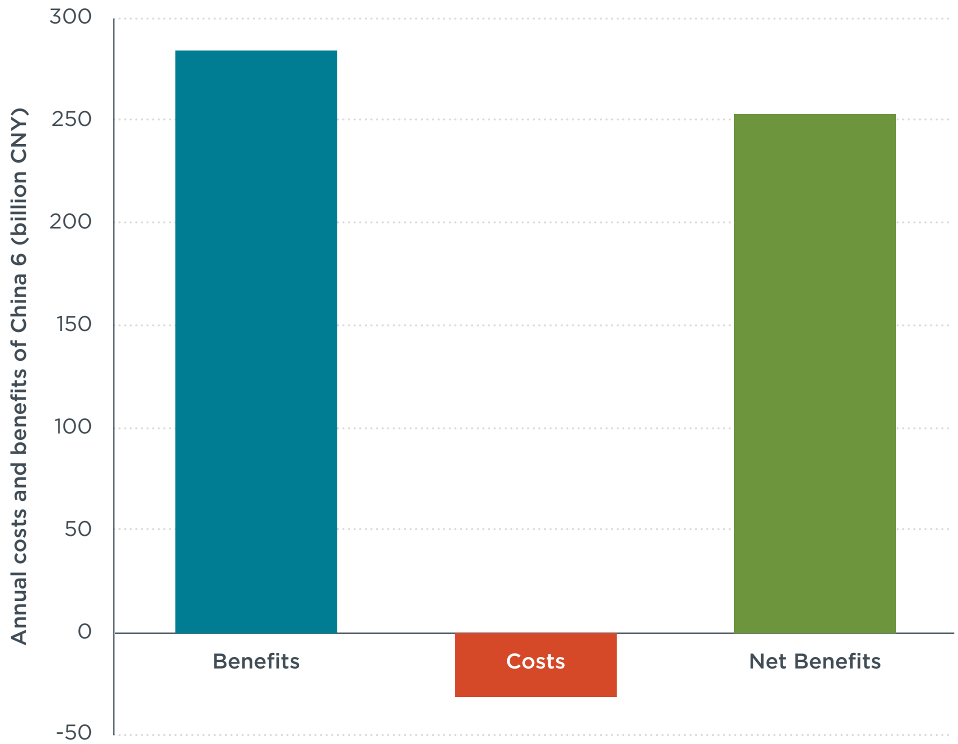


Figure 9. Costs and benefits of China 6 vehicle emission control in 2030.

Using income elasticity of 1.0 (Minjares et al., 2014) and the ratio of per capita income in China and the United States, we derived an estimate of the VSL of China to be $1.89 million in 2015 and $1.95 million in 2030 according to projected growth in per capita income.

Combining these quantified benefits with the costs, we examine the cost-effectiveness of the China 6 standard in 2030, about 12 years after standard phase-in and 7 years of full implementation.

4.2 RESULT

Costs and benefits are reported in currency units of 2015 USD and in Chinese *yuan* (CNY).[5] The adoption of a China 6 standard yields tremendous economic benefits over the mid-term (in 2030). We focus on 2030 considering the timetable of China's ambient air-quality standards and clean air action plans[6]. The cost-effectiveness of the standard will continue to scale up in the longer term as China 6 vehicles account for a greater share of vehicle activity. Figure 9 shows the costs and benefits arising from the proposed China 6 standard scenario in 2030. Most of the benefits come from curtailing the incidence of premature mortality as a result of $PM_{2.5}$ and ground-level ozone exposure in urban areas.

The total $PM_{2.5}$ and ozone-related health benefits from implementing China 6 are valued at USD 42.4 billion (CNY 284.3 billion) at a cost of USD 4.8 billion (CNY 31.8 billion) in

4 VSL is an economic concept. It is used to estimate "how much people are willing to pay for small reductions in their risks of dying from adverse health conditions that may be caused by environmental pollution" (U.S. EPA, 2014). It is used mostly in regulatory impact analyses and scientific research. It summarizes the value society places on preventing death for any particular person. This study begins with a suggested VSL value from the U.S. EPA (derived from economic studies of willingness-to-pay) that is then adjusted for China based on relative differences in gross national income.

5 Currency conversion rate between USD and CNY is assumed to be 6.7033:1 as of exchange rate on July 11, 2016. All monetary values are rounded to the first decimal.

6 The target of Chinese government on air pollution control, as was indicated in 2013 by Zhou Shengxian, then Environment Minister of China, is to have all the cities in compliance with national Class 2 air-quality standards by 2030.

the year 2030, with a benefit-to-cost ratio of 8.9:1 and annual net benefit of USD 37.7 billion (CNY 252.4 billion). Table 13 specifies the value of avoided premature deaths, technology costs, net benefits (by subtracting cost from benefit), and benefit-to-cost ratio for the three key regions, the rest of China, and the entire nation. Among the three key regions, the JJJ region demonstrates the greatest benefit-to-cost ratio, at 9.7:1.

Because the estimate of VSL is strongly affected by the choice of income elasticity, we also consider sensitivity analysis with two alternative elasticities: 0.5 and 2.0. The net benefits are still positive across this range (Table 14).

Table 13. Costs and benefits of China 6 LDV standard in 2030 (billion).

Region	Social benefit of avoided premature deaths USD (CNY)	Incremental vehicle technology cost USD (CNY)	Annual net benefits USD (CNY)	Benefit-cost ratio
JJJ	5.3 (35.8)	0.6 (3.7)	4.8 (32.1)	9.7:1
YRD	4.7 (31.4)	1 (6.6)	3.7 (24.9)	4.8:1
Guangdong	1.7 (11.5)	0.5 (3.5)	1.2 (8)	3.3:1
Rest of China	30.7 (205.5)	2.7 (18.1)	28 (187.4)	11.4:1
Nationwide	42.4 (284.3)	4.8 (31.8)	37.7 (252.4)	8.9:1

Table 14. Costs and benefits using different elasticity values at the national level (billion).

Elasticity	VSL Million USD (CNY)	Social benefit of avoided premature deaths USD (CNY)	Incremental vehicle technology cost USD (CNY)	Annual net benefits USD (CNY)	Benefit-cost ratio
1	2.0 (13.4)	42.4 (284.3)	4.8 (31.8)	37.7 (252.4)	8.9:1
0.5	4.1 (27.5)	89.8 (602.2)	4.8 (31.8)	85.1 (570.3)	18.9:1
2	0.4 (2.7)	9.4 (63.3)	4.8 (31.8)	4.7 (31.4)	2.0:1

5. CONCLUSION

The proposed standard is among the world's most stringent and combines best practices from both European and U.S. regulations. It is expected to help reduce emissions from four major ambient air pollutants—CO, HC, NO_X, and PM—by approximately 3,396, 4,184, 1,001, and 26 thousand metric tons, respectively, in 2030. These emission reductions would help decrease the national annual average $PM_{2.5}$ and ground-level ozone pollution concentrations by 1.1 $\mu g/m^3$ and 2.1 ppb, respectively, in 2030. The improved air quality would curtail the incidence of premature mortality caused by $PM_{2.5}$ and ground-level ozone exposure in urban areas. As a result, a total of 21,754 premature deaths and 28,559 hospital admissions would be avoided in 2030. The total health benefits from implementing the proposed standard are valued at USD 42.4 billion (CNY 284.3 billion) at a technology upgrade cost of USD 4.8 billion (CNY 31.8 billion) in 2030, with a benefit-to-cost ratio of 8.9:1 and annual net benefit of USD 37.7 billion (CNY 252.4 billion), indicating that it is a very cost-effective standard.

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Annex A. Papers used to update emission-control system costs.

Area covered	Paper
PGM loading LEV III- SULEV 30	Yi, S., Kim, H., Quelhas, S., Giler, C., Dang, D., & Kim, S. (2014). Investigation of a catalyst and engine management solution to meet LEV III - SULEV with reduced PGM. SAE Technical Paper 2014-01-1506. doi:10.4271/2014-01-1506
Catalyst size LEV III- SULEV 30	Kakema, T., Suehiro, Y., Matsuzono, Y., Narishige, T., & Hashimoto, M. (2015). Development of Pd-only catalyst for LEV III and SULEV30. SAE Technical Paper 2015-01-1003. doi:10.4271/2015-01-1003
Advanced TWC PGM use	Miura, M., Aoki, Y., Kabashima, N., Fujiwara, T., Tanabe, T., Morikawa, A., Ori, H., … & Matsui, S. (2015). Development of advanced three-way catalyst with improved NO_x conversion. SAE Technical Paper 2015-01-1005. doi:10.4271/2015-01-1005
PGM loading LEV III- SULEV 30	Ball, D., Zammit, M., Wuttke, J., & Buitrago, C. (2011). Investigation of LEV-III aftertreatment designs. SAE International Journal of Fuels and Lubricants, 4(2), 1–8. doi:10.4271/2011-01-0301.
PGM loading LEV III- SULEV 30	Craig, A., Warkins, J., Aravelli, K., Moser, D., Yang, L., Ball, D., Tao, T., & Ross, D. (2016). Low cost LEV-III, Tier-III emission solutions with particulate control using advanced catalysts and substrates. SAE International Journal of Engines, 9(2), 1276–1288. doi:10.4271/2016-01-0925
PGM loading LEV III- SULEV 30	Ball, D., & Moser, D. (2012). Cold start calibration of current PZEV vehicles and the impact of LEV-III emission regulations. SAE Technical Paper 2012-01-1245. doi:10.4271/2012-01-1245.
GDI TWC and GPF technology	Richter, J., Klingmann, R., Spiess, S., & Wong, K. (2012). Application of catalyzed gasoline particulate filters to GDI vehicles. SAE International Journal of Engines, 5(3), 1361–1370. doi:10.4271/2012-01-1244
GDI TWC and GPF technology	Ito, Y., Shimoda, T., Aoki, T., Yuuki, K., Sakamoto, H., Kato, K., Thier, D., … & Vogt, C. (2015). Next generation of ceramic wall flow gasoline particulate filter with integrated three way catalyst. SAE Technical Paper 2015-01-1073. doi:10.4271/2015-01-1073
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TWC substrate design	Schoenhaber, J., Richter, J., Despres, J., Schmidt, M., Spiess, S., & Roesch, M. (2015). Advanced TWC technology to cover future emission legislations. SAE Technical Paper 2015-01-0999. doi:10.4271/2015-01-0999
TWC substrate design	Tanner, C., Twiggs, K., Tao, T., Bronfenbrenner, D., Matsuzono, Y., Otsuka, S., Suehiro, Y., & Koyama, H. (2015). High porosity substrates for fast-light-off applications. SAE Technical Paper 2015-01-1009. doi:10.4271/2015-01-1009
Tier 3 Cost for gasoline and diesel vehicles	U.S Environmental Protection Agency. (2014). Control of Air Pollution from Motor Vehicles: Tier 3 Motor Vehicle Emission and Fuel Standards Final Rule. Regulatory Impact Analysis. EPA-420-R-14-005. Washington, D.C.: U.S. Environmental Protection Agency.
LEV III Cost for gasoline and diesel vehicles	California Environmental Protection Agency Air Resources Board. (2011). Initial statement of reasons for proposed rulemaking, public hearing to consider the “LEV III” amendments to the California greenhouse gas and criteria pollutant exhaust and evaporative emission standards and test procedures and to the on-board diagnostic system requirements for passenger cars, light-duty trucks, and medium-duty vehicles, and to the evaporative emission requirements for heavy-duty vehicles. Staff report. Retrieved from https://www.arb.ca.gov/regact/2012/leviiighg2012/leviiighg2012.htm