THE INTERNATIONAL COUNCIL ON CLEAN TRANSPORTATION

Cost-benefit assessment of the China VI emission standard for new heavy-duty vehicles

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Keywords: Tailpipe emissions, air quality, human health, particulate matter, nitrogen oxides, remote emissions monitoring

1. BACKGROUND

In July 2018, the Ministry of Ecology and Environment (MEE)[1] released the final rule for the China VI emission standard for new heavy-duty vehicles (HDV), hereafter referred to as *China VI* (MEE, 2018a).

The China VI standard is among the world's most stringent HDV emission standards and combines best practices from both European and U.S. regulations. It applies to diesel and gas fueled (i.e., liquefied petroleum gas and natural gas) engines, and applies to M1 vehicles with a gross vehicle weight (GVW) of more than 3,500 kg, as well as M2, M3, N1, N2 and N3 vehicles.[2] The standard will be implemented in two phases. China VI-a, which is largely equivalent to Euro VI,[3] will take effect starting on July 1, 2019 for new gas fueled HDVs; on July 1, 2020 for new urban HDVs, for example, public buses, sanitation and postal trucks; and on July 1, 2021 for all new HDVs. China VI-b, which introduces slightly more stringent testing requirements and a remote emission monitoring system, will take effect beginning on January 1, 2021 for new gas fueled HDVs, then on July 1, 2023 for all new HDVs. According to the latest data released by MEE, the vehicles affected by the China VI standard account for more than 90% of particulate matter (PM) emissions and nearly 70% of nitrogen oxides (NO_x) emissions from China's on-road transportation fleet (MEE, 2018b).

In order to effectively reduce tailpipe emissions from HDVs, improve urban air quality, and protect human health, the China VI standard tightens emission limits of regulated air pollutants and adds provisions to enhance vehicle emissions compliance in real-world driving conditions. In particular, the standard

- reduces NO_x and PM emission limits by about 70% compared to the current China V standard;
- introduces particulate number (PN) limits;
- changes the test cycles from the European Steady-state Cycle (ESC) and European Transient Cycle (ETC) to the more representative and dynamic World Harmonized Stationary Cycle (WHSC)

1 Formerly the Ministry of Environmental Protection (MEP)

2 M1: passenger vehicles, no more than eight seats in addition to the driver's seat.
M2: passenger vehicles, more than eight seats in addition to the driver's seat, GVW not exceeding 5,000 kg.
M3: passenger vehicles, more than eight seats in addition to the driver's seat, GVW exceeding 5,000 kg.
N1: vehicles for the carriage of goods, GVM≤ 3,500 kg
N2: vehicles for the carriage of goods, 3,500 kg <GVW≤ 12,000 kg.
N3: vehicles for the carriage of goods, GVM>12,000 kg.

3 For more details about Euro VI emission standard, see Williams & Minjares (2016).

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and World Harmonized Transient Cycle (WHTC);

- introduces new World Harmonized Not-to-exceed (WNTE) test;
- extends durability requirements;
- introduces full-vehicle Portable Emission Measurement System (PEMS) testing for type test, new production and in-service conformity testing, largely adopting European PEMS requirements, with some modifications to address unique driving conditions in China;
- requires the installation of improved on-board diagnostic (OBD) systems and adopts U.S. anti-tampering provisions;
- requires the installation of remote emission management on-board terminals (remote OBD);
- introduces a multi-component compliance program involving agency- and manufacturer-run emission tests during preproduction, production, and in-use stages; and
- introduces an HDV emission warranty program, in which manufacturers are required to guarantee emission-control parts valid for a minimum distance traveled or service time.

Table 1 compares the China VI-a and China VI-b emission standards to the China V/Euro V and Euro VI standards. For more details, see the International Council for Clean Transportation policy update (ICCT, 2018).

Table 1. Comparison of China V/Euro V, Euro VI, China VI-a, and China VI-b HDV emission standards

<table>
<tr><th colspan="2"></th><th>China V/
Euro V</th><th>Euro VI</th><th>China VI-a</th><th>China VI-b</th></tr>
<tr><td colspan="2">Engine test cycle</td><td>ESC, ETC</td><td colspan="3">WHSC, WHTC, WNTE</td></tr>
<tr><td rowspan="3">Emission limits on transient cycle</td><td>NO_x (g/kWh)</td><td>2</td><td colspan="3">0.46</td></tr>
<tr><td>PM (g/kWh)</td><td>0.03</td><td colspan="3">0.01</td></tr>
<tr><td>PN (#/kWh)</td><td>No limit</td><td colspan="3">6E+11</td></tr>
<tr><td colspan="2">PEMS test</td><td>No</td><td colspan="3">Yes</td></tr>
<tr><td rowspan="2">Emission limits for PEMS test</td><td>NO_x (g/kWh)</td><td>N/A</td><td colspan="3">0.69 (CF=1.5)</td></tr>
<tr><td>PN (#/kWh)</td><td>N/A</td><td>No limit</td><td>No limit</td><td>1.2E+12 (CF=2.0)</td></tr>
<tr><td colspan="2">Altitude boundary for PEMS test</td><td>N/A</td><td><1,700 m</td><td><1,700 m</td><td><2,400 m</td></tr>
<tr><td colspan="2">Payload for PEMS test</td><td>N/A</td><td>50%–100% (Euro VI-c and before)
10%–100% (Euro VI-d)</td><td>50%–100%</td><td>10%–100%</td></tr>
<tr><td colspan="2">OBD requirements</td><td>Euro V OBD</td><td>Euro VI OBD</td><td colspan="2">Euro VI OBD + U.S. anti-tampering provisions</td></tr>
<tr><td colspan="2">Remote OBD</td><td>No</td><td>No</td><td>No</td><td>Yes</td></tr>
<tr><td colspan="2">Emission durability periods for different vehicle categories</td><td>100,000 km/ 5 years
200,000 km/ 6 years
500,000 km/ 7 years</td><td>160,000 km/ 5 years
300,000 km/ 6 years
700,000 km/ 7 years</td><td colspan="2">200,000 km/5 years
300,000 km/6 years
700,000 km/7 years</td></tr>
<tr><td colspan="2">Emission warranty program</td><td>No</td><td>No</td><td colspan="2">Yes</td></tr>
</table>

This paper aims to provide a comprehensive assessment of the environmental and social benefits as well as compliance costs of implementing the China VI standard at national and subnational levels. First, we use state-of-the-art methods to estimate the emission reductions of regulated ambient air pollutants, the consequent changes in $PM_{2.5}$ and ground-level ozone concentrations, and avoided health impacts (including premature deaths and hospitalization) attributable to implementation of the China VI standard. Social benefits are then quantified by estimating the economic value of reductions in premature deaths resulting from improved urban air quality. Second, we evaluate the incremental technology costs of complying with the China VI standard based on the ICCT's HDV technology cost assessment model. Finally, we compare the social benefits and technology costs and get the benefit-cost ratio of the China VI standard. Our research focuses mainly on nationwide impacts but also separately analyzes China's three key regions: the so-called Jing-Jin-Ji (or JJJ) region, an agglomeration surrounding the nation's capital city, including Beijing, Tianjin and Hebei province; the Yangtze River Delta (or YRD) region, including Shanghai, Jiangsu, and Zhejiang provinces; and Guangdong province.

2. HEALTH BENEFIT EVALUATION

The China VI emission standards will lead to significant emission reductions of regulated ambient air pollutants. As a result, substantial benefits will be achieved in air quality and human health. This section introduces the methodology we used to model the impacts of China VI on emissions, air quality, and human health (Section 2.1) and presents the modeling results (Section 2.2) in detail.

Table 2. Implementation dates of China VI standard modeled under the China VI scenario

Region	Vehicle type	Emission standard applied	Implementation date
JJJ region	All new HDVs	China VI-b	January 1, 2019
YRD region			
Guangdong			
Nationwide	New gas fueled HDVs	China VI-a	July 1, 2019
	New urban HDVs	China VI-a	July 1, 2020
	New gas fueled HDVs	China VI-b	January 1, 2021
	All new HDVs	China VI-a	July 1, 2021
	All new HDVs	China VI-b	July 1, 2023

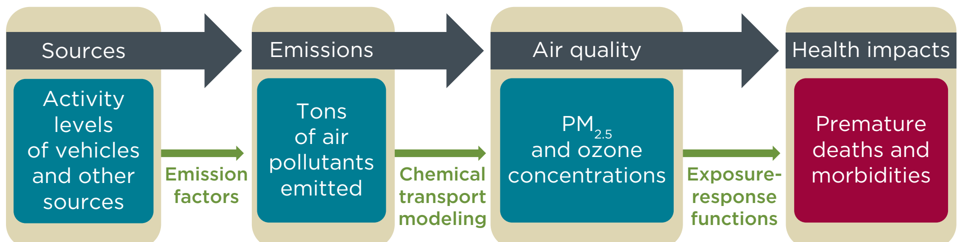

Figure 1. Conceptual framework for evaluating the health impacts attributable to $PM_{2.5}$ and ozone exposure.

2.1 METHODOLODY

This section introduces the methods we applied in this paper to estimate the emission reductions, air quality improvements, and avoided health impacts such as premature deaths and morbidities attributable to the implementation of China VI.

2.1.1 Technology roadmap

To evaluate the impacts of China VI on emissions, air quality, and human health, we examined two scenarios:

1. Business As Usual (BAU): The BAU scenario assumes China VI is not implemented in China throughout the study period (2015–2030).

2. China VI: This scenario assumes China VI applies to new HDVs following the schedule described in Table 2. The three key regions (JJJ, YRD, and Guangdong) are assumed to adopt China VI ahead of the national timeline based on the State Council's Three-Year Action Plan to Defend Blue Sky (China State Council, 2018) and previous patterns. Historically, these regions have advanced schedules for adopting previous emission standards, including the China 6 emission standard for light-duty vehicles (LDVs).[4]

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 1. 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 premature deaths and morbidities 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. The health benefits of the China VI standard are equal to the difference between the modeled health impacts results of the BAU and China VI scenarios.

4 For more details about the China 6 LDV emission standard, see ICCT (2017).

2.1.2 Emission modeling methodology

The China Mobile Source Emission Inventory Model (Façanha, Blumberg, & Miller, 2012; Shao & Wagner, 2015; Yang, Wang, Shao, & Muncrief, 2015) was used to estimate tailpipe emissions from HDVs 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 calculates the tailpipe emissions of conventional air pollutants emitted from vehicles. Seven conventional air pollutants are taken into consideration, including carbon monoxide (CO), sulfur dioxide (SO_2), nitrogen oxides (NO_x), hydrocarbon (HC), particulate matter (PM), black carbon (BC) and organic carbon (OC). The model is updated regularly to ensure that it accurately reflects the policy efforts and fleet emissions in China. For this analysis, it was updated with a new set of HDV emission factors, which were developed with inputs from real-world testing conducted by Tsinghua University and the Vehicle Emission Control Center (VECC)-MEE.

The improvements in the China VI standard, including lower emission limits, more representative test cycles, and stricter compliance and enforcement requirements, were all taken into consideration when modeling emissions from HDVs under the China VI scenario. Particularly, to model the potential impacts of provisions related to compliance and enforcement, for example PEMS testing and improved OBD systems, we defined a parameter called vehicle emission compliance ratio. That is defined as 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. We assumed higher compliance ratios correspond to more stringent compliance and enforcement requirements. In this study, we looked at the existing in-use vehicle emission testing data and consulted with VECC to determine the compliance ratios for pre-China I through China VI-b HDV emission standards.

To fulfill the data requirements of air quality modeling, we also estimated emissions from LDVs as well as non-transport emission sources. For LDVs, we used the method previously introduced in Cui et al. (2017) to calculate the fleet emissions. The China 6 LDV emission standard was taken into consideration in our calculation. For other anthropogenic emission sources, for example, coal burning, we used the emission inventory developed by Tsinghua University (GBD MAPS Working Group, 2016). For natural sources, we used the Model of Emissions of Gases and Aerosols from Nature (MEGAN) described in Guenther et al. (2006). To analyze the impacts of emission reduction of HDVs on air quality and human health due to the implementation of China VI emission standards, emissions from LDVs and non-transport emission sources are the same in the BAU and China VI scenarios.

2.1.3 Air quality modeling methodology

Based on the emission amounts calculated in Section 2.1.2, we conducted air quality simulation to estimate annual average $PM_{2.5}$ concentrations, both primary and secondary formation, and annual average daily 1-hour peak ozone concentrations in 2030 for the two scenarios considered. This was done using the offline-coupled Weather Research and Forecasting (WRF) model (Michalakes et al., 2001) and the Community Multi-Scale Air Quality (CMAQ) model (Byun, Ching, Novak, & Young, 1998).

The WRF model is a new generation meso-scale numerical weather prediction system designed to serve a wide range of meteorological applications. WRF has been widely used to provide meteorological input fields necessary for chemical transport models. In this analysis, we applied WRF v 3.8.1 to generate meteorological fields with 23 vertical layers. Initial and boundary conditions were taken from the U.S. National Centers for Environmental Prediction Final Analysis (NCEP-FNL).[5] Land use/land cover and topographical data were taken from a default WRF input dataset.

The CMAQ model is a 3-D Eulerian atmospheric chemistry and transport modeling system designed to simulate multiple pollutants from city to transcontinental scales. CMAQ has been widely used to simulate ambient $PM_{2.5}$ and ozone concentrations, using emission inventory and meteorological fields generated by the WRF model as inputs. This study used the latest CMAQ v 5.2, choosing CB05 as the gas-phase mechanism and AERO6 as the aerosol module. Vertically, 14 layers were included from surface to the tropopause, covering approximately 16 km.

The modeling domain covered all of China and part of East Asia with a grid resolution of 36 km by 36 km. For both $PM_{2.5}$ and ozone, we ran simulations for January, April, July, and October with three days of spin-up time each.[6] We then obtained annual average $PM_{2.5}$ and annual 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 representative months.

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. 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 et al. (2014) to estimate the gridded premature deaths of four major diseases

5 The NCEP FNL data are on 1-degree by 1-degree grids prepared operationally every six hours.

6 The three-day spin-up period is designed to minimize the influence of the initial condition used as the start of the simulation.

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_o, α, γ and δ for different diseases vary, as shown in Table 3. Second, based on Equation 2, we converted the calculated RR value to attributable fraction (AF), which represents the percentage of baseline death rate attributable to $PM_{2.5}$ exposure. Lastly, we estimated the $PM_{2.5}$-attributable premature deaths (PD) of each disease using Equation 3.

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

$$AF = \frac{RR - 1}{RR} \quad (2)$$

$$PD = AF \times BDR \times P \quad (3)$$

Where:

- C = annual average $PM_{2.5}$ concentrations in the target grid (µg/m³);
- C_o = 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 from 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 the 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 two diseases considered are shown in Table 4. Second, based on Equation 2, we converted the calculated RR value to attributable fraction (AF), which represents the percentage of baseline hospitalization rate attributable to $PM_{2.5}$ exposure. Lastly, we estimated the $PM_{2.5}$-attributable hospitalizations (H) of each disease using Equation 5.

Table 3. Values of key parameters used to estimate the premature deaths from four major diseases due to $PM_{2.5}$ exposure.

Disease	α	γ	δ	C_o	*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
Lung Cancer	159	0.000119	0.735	7.24	0.000383

Table 4. 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

$$RR(C) = e^{\beta C} \quad (4)$$

$$H = AF \times BHR \times P \quad (5)$$

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.

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, ε, C_o, and baseline death rate (BDR) of respiratory diseases were set as 0.0017, 33.3 ppb, and 0.000696, respectively.

$$RR(C) = e^{\varepsilon(C-C_o)} \quad (6)$$

Where:

- C = annual average daily 1-hour peak ozone concentrations in the target grid (ppb),
- C_o = low-concentration threshold (LCT) below which ozone has no effect on mortality (ppb), and
- ε = 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 ε, C_o, and baseline hospitalization rate (BHR) of respiratory diseases were set as 0.00026, 35 ppb, and 0.00325, respectively.

$$RR(C) = e^{\tau(C-C_o)} \quad (7)$$

Where:

C = annual average daily 1-hour peak ozone concentrations in the target grid (ppb),

C_o = 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.

For all the diseases mentioned above, we derived the 2013 BDR and BHR from the 2014 China Health and Family Planning Yearbook (National Health and Family Planning Commission, 2015) and assumed constant BDRs and BHRs over the period 2013 to 2030. We got the 2010 population data from the LandScan global population database (Dobson et al., 2000) and assumed the gridded population to be unchanged from 2010 to 2030. As a result of these two assumptions, the health impacts estimated in this analysis are conservative.

By comparing the health impacts modeling results of both scenarios, we obtained the health benefits—which is to say, premature deaths and morbidities avoided—attributable to the China VI standard.

2.2 MODELING RESULTS

Using the modeling methods introduced in Section 2.1, we evaluated the impacts of China VI on emissions, air quality, and human health in the entire nation and three key regions. This section presents the modeling results in detail.

Table 5. Projected emission reduction from HDV sector in 2030 compared with the BAU

	CO	HC	NO_x	PM
Emission Reduction (thousand metric tons)				
China	1,327	86	4,512	159
JJJ	156	10	510	18
YRD	230	15	772	28
Guangdong	101	6	353	13
Percentage Reduction (%)				
China	56	55	86	82
JJJ	60	61	88	85
YRD	58	57	88	84
Guangdong	55	51	88	84

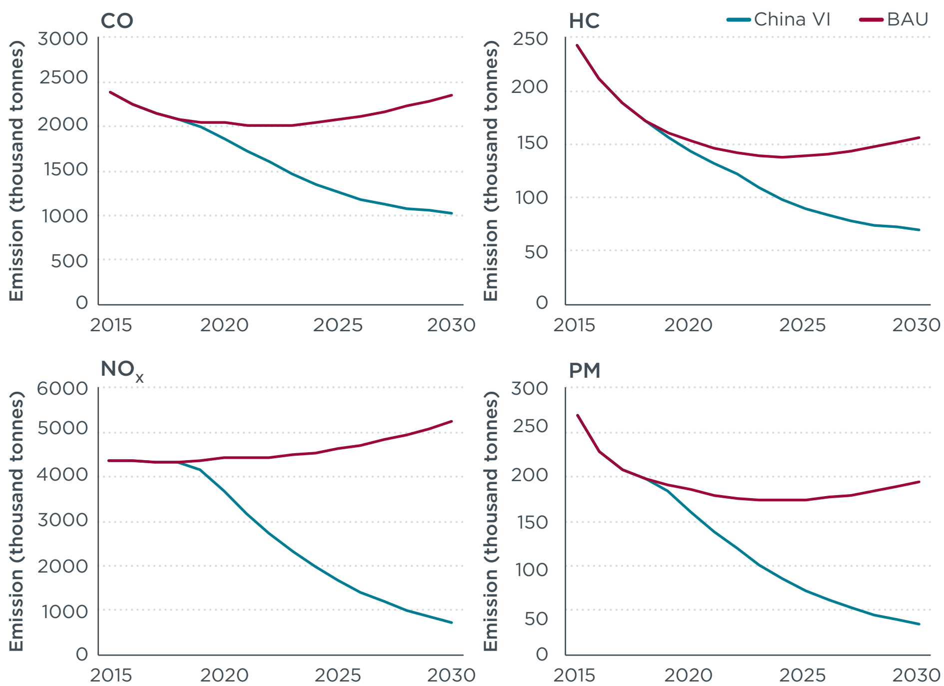

Figure 2. Emission modeling results for CO, HC, NO_x and PM from HDVs in China.

2.2.1 *Emission impacts*

Table 5 provides the projected reductions, both in absolute terms and percentages, of four major conventional air pollutants from HDVs in the entire nation and three key regions in 2030 attributable to the implementation of China VI. It is clear that emissions of all the air pollutants are greatly reduced by introducing the China VI standard within the timeline considered in this analysis. Specifically, the emission reductions for NO_x, and PM reach 86% and 82%, respectively, at the national level. In the three key regions, the effects of China VI are sometimes even more prominent. Figure 2 depicts the annual emissions of each conventional air pollutant from China's HDV fleet under BAU and China VI scenarios from 2015 to 2030. Over this period, the accumulated emission reductions of CO, HC, NO_x and PM in China are 8667, 526, 31158, and 1079 thousand metric tons, respectively.

2.2.2 Air quality impacts

Assuming major sectors, such as the power sector and the industrial sector, continue to meet State Council mandates for air pollution control, national Class 2 air-quality standards (MEE, 2012) are likely to be achieved in 2030 even before considering the air quality benefits of China VI. However, both at the national level and within key regions, levels of $PM_{2.5}$ exposure will still fail to meet national Class 1 standards, which are more in line with U.S. EPA and WHO air-quality guidelines but are currently applied in China only in special regions, such as national parks.

According to our analysis, the China VI standard will ensure compliance with national Class 2 air-quality standards and move national air quality toward Class 1 levels. Through implementation of China VI, concentrations of both $PM_{2.5}$ and ozone in 2030 will further decline from the BAU scenario, as shown in Table 6. At the national level, average population-weighted exposure to annual average $PM_{2.5}$ will fall by 1.04 µg/m^3, a reduction of 5% from the BAU scenario. Larger reductions will occur in the JJJ region in terms of both absolute values and percentages, where diesel vehicles are the third largest contributor to local $PM_{2.5}$ pollution (Xue, 2018). Similarly, the annual average daily 1-hour peak ozone concentration will fall nationwide by 0.93 ppb (population-weighted), equal to a decline of 2.1% from BAU. Larger absolute reductions are projected to occur in the YRD and JJJ regions.

Table 6. Effect of China VI on $PM_{2.5}$ and ozone concentrations in 2030 compared with the BAU scenario[7]

	Annual average $PM_{2.5}$		Annual average daily 1-hour peak ozone	
	µg/m³	% Change	ppb	% Change
China	–1.04	–5.0	–0.93	–2.1
JJJ	–1.46	–5.2	–0.99	–2.0
YRD	–1.04	–3.9	–1.46	–3.1
Guangdong	–0.65	–4.8	–0.74	–1.7

2.2.3 Health benefits

According to our analysis, in 2030, the implementation of China VI will avoid more than 29,200 premature deaths and more than 17,300 hospitalizations nationwide, compared to the BAU scenario, as a result of reduced exposure to both $PM_{2.5}$ and ozone pollutions. 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 4.5% of avoided premature deaths and 5.5% of avoided morbidities. Table 7 provides avoided premature deaths and morbidities for $PM_{2.5}$ and ozone in China and in three key regions. Because we did not take population growth and BDR/BHR increase due to an aging population into consideration, the health benefits estimated in this analysis are conservative.

Table 7. Avoided health impacts from China VI implementation in 2030 compared to BAU

	Avoided premature deaths			Avoided hospital admissions		
	$PM_{2.5}$	O_3	Total	$PM_{2.5}$	O_3	Total
China	27,887	1,320	29,207	16,392	962	17,354
JJJ	2,319	103	2,422	1,685	75	1,760
YRD	2,680	222	2,902	1,749	162	1,911
Guangdong	1,948	75	2,023	755	55	810

3. TECHNOLOGY COST ASSESSMENT

The technology required to meet the China VI standards for HDVs has associated costs of manufacturing that include the physical parts and cost associated with developing that technology. The cost values presented in this section build upon the study by Posada, Chambliss, and Blumberg (2016) on the cost of emission reduction technologies for HDVs. The China VI technology cost analysis presented here updates that work with cost items relevant to the Chinese heavy-duty fleet, especially in terms of engine size profiles.

The emission control technology cost assessment focuses on diesel powertrains. The costs assessed include in-cylinder technologies to control engine-out emissions, the aftertreatment technologies that act on the exhaust stream, and OBD systems. Engine-out emissions are reduced by adjusting the temperature and air/fuel (A/F) balance within the engine, using improvements to fuel injection and air handling and employing exhaust gas recirculation (EGR). Aftertreatment systems include selective catalytic reduction (SCR) systems with ammonia as the reducing agent to control NO_x and diesel oxidation catalysts (DOCs) and diesel particulate filters (DPFs) to control PM. OBD systems detect malfunctions in emission control systems and their components, preventing high emissions to go unchecked.

7 Concentration values are population-weighted.

To evaluate the incremental costs of China VI emission control technology for HDVs, we updated the cost modeling methodology developed by Posada et al. (2016) with information from recent regulatory documents and peer-reviewed journal articles. The cost of emission control technology developed by Posada et al. (2016) covers the manufacturing cost associated technology steps required to reach Euro III, Euro IV, Euro V and Euro VI emission standards. The methodology to develop those cost estimates was: (a) identification of the technology pathways that reached commercial deployment in Europe, (b) estimation of emission control component costs for in-cylinder control (e.g., EGR systems) and aftertreatment systems (i.e., DOC, DPF, SCR), and OBD, and (c) integrating the costs for the different technology pathways. Component cost structures were developed based on public information from several regulatory impact assessments developed by the U.S. Environmental Protection Agency (EPA) HDV greenhouse gas standards regulation (U.S. EPA, 2011) and the comprehensive EPA 2010 rulemaking, which covered the regulatory stages for the US 2007 and US 2010 emission standards for new engines used in heavy-duty highway vehicles (U.S. EPA, 2000); the dollar values used for each emission control component were updated with the latest cost of materials for the 2016 work by Posada et al. and again for this 2018 work. This component cost update is required for several reasons: manufacturers improve their designs and reduce the use of key components, such as platinum for DPFs, and the price of those components has changed in the past five years. All of those changes are accounted for in this China VI emission control cost assessment.

Table 8. Engine sizes representative of HDV applications in China

Application	Engine size, L	HDV category
Urban Bus	4.8	Light heavy-duty (LHD)
Delivery Truck	6.7	Medium heavy-duty (MHD)
Coach bus	6.7	Medium heavy-duty (MHD)
Dump truck utility	8.4	Heavy heavy-duty (HHD)
Long-haul tractor Trailer	10.1	Heavy heavy-duty (HHD)

Table 9. Incremental costs to comply with the China VI standards for HDVs

Incremental Costs	LHD	MHD	HHD	HHD
	4.8L	**6.7L**	**8.4L**	**10.1L**
China V to VI	$1,425	$1,727	$2,001	$2,274

The costs of some in-cylinder and most aftertreatment control technologies are proportional to engine size. Estimating the China VI emission control technology costs required defining engine sizes representative of engines found in Chinese HDVs. The engine sizes used in this analysis are listed in Table 8.

Table 9 summarizes the estimated HDV emission control incremental costs with respect to the China V standard. Most of the compliance costs come from the adoption of DPFs to meet PN and PM standards, upgraded SCR systems with zeolite-based catalytic systems that deliver better NO_x control during urban driving conditions (which Euro/China V SCR technology does poorly), and the OBD requirements. OBD costs for China VI were obtained from U.S. regulatory documents, and estimated at approximately $425 (Posada et al., 2016). China V OBD was estimated to be less than half the cost of the China VI version, as the China VI technology has broader monitoring and sensing requirements for a larger number of emission control technologies than the very basic China V OBD versions.

More cost details can be found in the Appendix. To give an example, for in-cylinder systems, variable geometry turbochargers (VGTs) are needed for China VI; this technology was estimated to add $185 to the conventional turbocharger technology costs for the HDD engines. The most important cost item, the DPF, was estimated to be within a range of $530 to $1,000 based on engine size. Note that the single most expensive emission control system, the SCR ($2,200–$3,240), has been in use since China IV, and thus the impact on China VI costs is limited. This is justified because the benefit analysis covers only the China VI emission improvement over the baseline standard.

4. COMBINED ANALYIS: 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 VI standard with the economic value of avoided premature deaths in a single year (2030).

4.1 METHODOLOGY

For costs, as introduced in Section 3, we mainly assessed the manufacturer costs in vehicle technology upgrades for complying with the China VI standard, but not the manufacturer markups, costs associated with increased testing requirements, or fuel and other costs during operation. Vehicle technology costs were calculated by estimating the per vehicle incremental cost of technology needed to meet China VI compared to China V, and 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 California Air Resources Board (CARB) estimates.

The health benefits stemming from the introduction of China VI are quantified in terms of the economic value of avoided premature deaths, not including avoided hospitalization, attributable to lower exposure to ambient $PM_{2.5}$ and ozone concentrations. The economic benefits of reduction in premature mortality are calculated based on value of a statistical life (VSL), an indicator of willingness to pay to avoid health impacts. 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 method; 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, and Sharp (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 U.S. dollars) adjusted to the year of analysis (U.S. EPA, 2010). The corresponding value in 2015 U.S. dollars is $8.75 million. Using income elasticity of 1.0 (Minjares et al., 2014) and the ratio of per capita incomes 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 VI standard in 2030, about 11 years after standard phase-in and 7 years of full implementation. We focus on 2030 considering the timetable of China’s ambient air-quality standards and clean air action plans.[8] By design, this choice of time frame results in a conservatively low estimate of the net benefits of China VI, because air quality and health benefits will continue to increase after 2030 as China VI vehicles make up a growing portion of the in-use fleet.

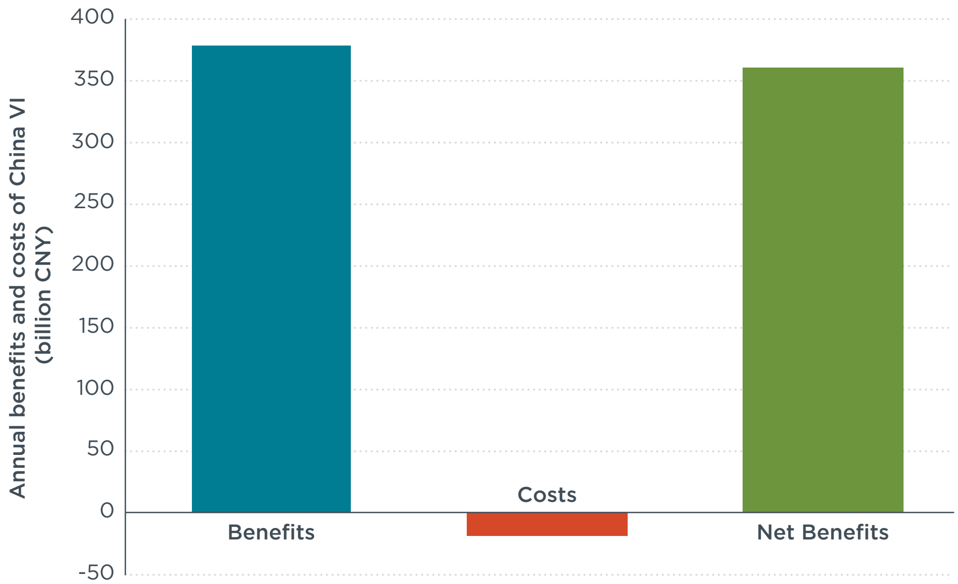

Figure 3 Annual benefits and costs of China VI vehicle emission control in 2030.

4.2 RESULT

Costs and benefits are reported in currency units of 2015 U.S. dollars. and in Chinese yuan.[9] The adoption of the China VI standard yields tremendous economic benefits with relatively low costs over the mid-term (in 2030). Figure 3 shows the annual costs and benefits of the China VI standard in 2030. Most of the benefits come from curtailing the incidence of premature deaths as a result of $PM_{2.5}$ exposure in urban areas.

8 The target of the 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.

9 Currency conversion rate between U.S. dollars and Chinese yuan is assumed to be 6.6401 as of exchange rate on September 26, 2017.

The total $PM_{2.5}$ and ozone-related health benefits from implementing China VI are valued at $57 billion (378 billion yuan) at a cost of $2.8 billion (18.4 billion yuan) in the year 2030, with a benefit-to-cost ratio of 21:1 and annual net benefit of $54 billion. Table 10 specifies the value of avoided premature deaths, technology costs, net benefits (by subtracting cost from benefit), and benefit-to-cost ratio for the entire nation and the three key regions. Among the three key regions, Guangdong province demonstrates the greatest benefit-to-cost ratio, at 19:1. These benefit-cost ratios are extremely favorable; from an economic perspective, a policy is considered worthwhile as long as the benefit-cost ratio is greater than one.

Table 10. Costs and benefits of China VI HDV standard in 2030

	Social benefit of avoided premature deaths in billions of $ (Yuan)	Incremental vehicle technology cost in billions of $ (Yuan)	Annual net benefits in billions of $ (Yuan)	Benefit-cost ratio
China	57 (378)	2.8 (18.4)	54 (360)	21:1
JJJ	4.7 (31.4)	0.3 (2.0)	4.4 (29.4)	16:1
YRD	5.7 (37.6)	0.5 (3.1)	5.2 (34.5)	12:1
Guangdong	3.9 (26.2)	0.2 (1.4)	3.7 (24.8)	19:1

5. Conclusion

The China VI standard is among the world's most stringent HDV emission standards and combines best practices from both European and U.S. regulations. It will be a key pathway to clean up diesel emissions and is therefore a critical step toward winning the war against air pollution in China. According to our analysis, the China VI standard will reduce emissions of four major ambient air pollutants—including CO, HC, $NO_{X,}$ and PM—by approximately 1,327 thousand metric tons, 86 thousand metric tons, 4,512 thousand metric tons, and 159 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 1.04 $\mu g/m^3$ and 0.93 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, especially in urban areas. As a result, at least 29,200 premature deaths and 17,350 hospital admissions would be avoided annually in 2030. The health benefits from implementing China VI are valued at $57 billion (378 billion yuan) at a technology upgrade cost of $2.8 billion (18.4 billion yuan) annually in the year of 2030, with a benefit-to-cost ratio of 21:1 and annual net benefit of $54 billion (360 billion yuan), indicating that it is a very cost-effective standard.

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Appendix

INCREMENTAL COSTS TO COMPLY WITH THE CHINA VI STANDARDS, WITH RESPECT TO CHINA V, BY TECHNOLOGY[10]

Technology	LHD	MHD	HHD	HHD
	4.8L	6.7L	8.4L	10.1L
1. A/F control & engine-out emissions				
Fuel system 2100–2200 bar — 50% of cost[10]	$24	$36	$55	$73
Turbocharger — 50% of cost	$0	$0	$0	$0
VGT (extra cost with respect to turbocharger) — 50% of cost	$128	$151	$169	$185
EGR system	$0	$0	$0	$0
EGR cooling	$0	$0	$0	$0
Combustion improvements	$55	$55	$55	$55
Cost of A/F control and engine-out emissions	$207	$242	$279	$313
2. Aftertreatment systems				
DOC	$0	$0	$0	$0
DPF	$525	$691	$841	$991
SCR	$446	$546	$634	$722
Closed crankcase filtration	$0	$0	$0	$0
Cost of aftertreatment systems	$971	$1,237	$1,475	$1,713
3. Total cost of hardware	$1,178	$1,479	$1,754	$2,026
4. OBD and sensors	$213	$213	$213	$213
5. Fixed costs (R&D)	$35	$35	$35	$35
6. Total incremental cost of emission control technologies	$1,426	$1,727	$2,002	$2,274

10 For components that, in addition to emission control, serve other purposes such as performance improvement or basic functioning, only 50% of the cost is considered in this analysis.