Environmental Impact Assessment of Electro-Mobility in China

A Lifecycle Assessment

Final Draft for Comments

Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ) GmbH

On behalf of:

Federal Ministry for the Environment, Nature Conservation, Building and Nuclear Safety

of the Federal Republic of Germany

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Electro-Mobility and Climate Protection in China

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China's continuously growing traffic volume, especially in individual transport, not only causes environmental concerns, it also puts pressure on China to address its strong dependence on oil imports. The Chinese government promotes the development of electro-mobility and alternative fuels as a means to decrease this dependency and increase energy efficiency in the transport sector. With zero tailpipe emissions electric vehicles also have strong potential to improve urban air quality. To be sustainable electro-mobility needs to be both environmentally friendly and safe. This is not only about electric vehicles (EVs). It also means fostering renewable energies in the national grid and designing integrated strategies for charging and maintaining electric vehicles. Innovative recycling plans and new mobility concepts can enhance the environmental impact of electric vehicles over their life cycle.

The "Climate Protection and Electro-Mobility in China" Project is therefore structured into four components.

Component 1: Environmental Impacts of Electro-Mobility in China
Component 2: Electro-Mobility and Environmental Standards
Component 3: Battery-Recycling
Component 4: Electric Vehicles and Sustainable Mobility

Table of Contents

1. Methodology and Database 4

1.1. Methodological concept and system boundary 4

1.1.1. System boundaries 5

1.2. Fuel-cycle analysis 7

1.2.1. Fuel-cycle analysis methodology and fuel pathways simulation 7

1.2.2. Database of electricity generation sector 9

1.2.3. Database of fuel economy and air pollutant emissions on the vehicle operation stage 17

1.3. Material-cycle analysis 24

1.3.1. Vehicle components 24

1.3.2. Composition map for battery and other key parts 25

1.3.3. The energy consumption in the critical process of material production 28

1.3.4. The energy consumption in batteries module and the production of other key parts 31

1.3.5. Database on Key Time-series Parameters 31

1.4. Projection of light-duty vehicles population and fleet composition 32

1.4.1. Vehicle growth forecast model 32

1.4.2. Projection of light-duty vehicles 34

1.4.3. The analysis of fleet composition and travelled distance 35

2. The Well to Wheels Energy Consumption and Pollutant Emissions of LDPV 38

2.1. Well-to-wheel result in China 38

2.1.1. Energy use and CO_2 emissions 38

2.1.2. Air pollutant emissions 41

2.1.3. Discrepancy between the two scenarios 46

2.2. Well-to-wheel result in six regions 47

2.2.1. Energy use and CO_2 emissions 47

2.2.2. Air pollutants emissions 51

2.3. Specific case study for Beijing 58

2.3.1. The natural gas case for Beijing 59

2.3.2. The wind power case for Beijing 65

3. The Energy Consumption and CO2 Emissions of LDPV in Full Life Cycle 72
3.1. Vehicle-cycle result of LDPV 72
3.1.1. Energy use and CO_2 emissions for battery..72
3.1.2. Energy use and CO2 emissions for LDPV..77
3.2. Full life cycle result of LDPV 79
3.2.1. Full life cycle result in China..79
3.2.2. Full life cycle results in PRD and JJJ 81
4. Vehicle Fleet Energy Saving and Emissions Reductions 84
4.1. The current penetration status of HEV/PHEV/BEV in China.................................84
4.2. EV penetration scenarios 85
4.2.1. The benefit of energy consumption and CO_2 emissions for the fleet 87
4.2.2. The benefit of pollutant emissions for fleet 89
4.3. The benefit of energy saving and emissions reduction for fleet in three regions 91
4.3.1. The benefit of energy consumption and CO_2 emissions for fleet......................91
4.3.2. The benefit of pollutants emissions for fleet...94
5. Conclusions 97
5.1. Oil security..97
5.2. Fossil energy and CO_2 mitigation 97
5.3. Air pollutant emissions mitigation 98
6. Discussions and Outlook 100
6.1. Generation mix 100
6.2. Integration of renewable energy into electric vehicle charging systems 101
6.3. User acceptance 101
6.4. Limitations and unsolved issues of this study 102
7. Conclusion 104

Abstract

The Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ) GmbH and the China Automotive Technology and Research Center (CATARC) are jointly commissioned by the German Federal Ministry for the Environment, Nature Conservation, Building and Nuclear Safety (BMUB) and the Chinese Ministry of Science and Technology (MoST), to manage the implementation of the bilateral technical cooperation project on Electro-Mobility and Climate Protection in China.

One of the chief goals of the above project is to identify and analyse the climate and environmental effects of Electro-Mobility in China. Given this, GIZ commissioned the Tsinghua University to conduct a comprehensive life-cycle assessment to quantify the full lifecycle emissions of new energy vehicles (battery electric vehicles, plug-in hybrids and hybrids) and internal combustion engine vehicles (ICEV) within the 2030 timeframe.

To what extent electric vehicles (EVs) alleviate the environmentally harmful effects of motorised transport depends on a number of influencing factors. When driven in purely electric mode, EVs produce neither air pollutants nor greenhouse gases at the tailpipe. The environmental impacts of EVs consequently depend on upstream emissions such as the emissions during electricity production as well as the vehicle manufacturing and recycling process. Whether the electricity is generated from renewable energy or from fossil fuels is a key question in this regard. Additional factors play an important role. These include: the number of EVs in the market, vehicle kilometres travelled and their real-world power consumption, the resource efficiency of the production / recycling process and the type of vehicles they replace.

Through this project, a comprehensive database to forecast the growth of the passenger car stock, its relevant vehicle activity, energy and emission profiles were developed. The life-cycle energy and environmental impacts of different electro-mobility scenarios were fully assessed for six regional grid systems in China and compared with an alternative baseline scenario that relies exclusively on passenger cars with conventional propulsion systems. The impact of power generation on the environmental impact of electric vehicles is illustrated in a case study for Beijing, were an increasing power generation from natural gas and wind is hypothetically assumed.

By analysing respective greenhouse gas emissions and air pollutant emissions, the impact assessment stresses the importance of a high share of renewable energy sources in the grid mix. Particularly in regions where energy production heavily relies on carbon intensive coal based power generation, such as the Jing-Jin-Ji region, the life-cycle analysis concludes that the emission reduction effects of battery electric vehicles during the operation stage are offset by upstream emissions during energy production. Lifecycle CO_2 emissions were of special interest in the study. In the baseline year 2010 the lifecycle CO_2 emissions of an electric vehicle (EV) exceeded the emissions of an internal combustion engine vehicle (ICEV) (when using the national grid mix) not least due to significant energy demand in the vehicle cycle. This decreases in 2015 and is expected to continue to do so in successive years due to the higher share of renewable energies and cleaner coal-production technologies. By 2030 it is projected that an EV will produce 27% fewer emissions in the full lifecycle compared to an ICEV. Substantial CO2 mitigation effects for the entire vehicle fleet occur from 2020 onwards. By 2030 propulsion technologies could save between 40 and 47 million tonnes of CO2. Various air pollutants were analysed in the study. EV promotion significantly reduces volatile organic compound and carbon monoxide emissions, but may increase nitrogen oxide, sulphur dioxide and PM2.5 emissions significantly.

The lifecycle assessment (LCA) results show: Promoting e-mobility is not a low-hanging fruit in the battle against climate change. The long-term mitigation potential of EVs could come at significant abatement and environmental costs today. Nonetheless the LCA also shows that a low-carbon automotive sector is not attainable without EVs.

Abbreviations

ADR	Assembly, Displacement and Recycling
AER	All Electric Range
ANL	Argonne National Laboratory
BEV	Battery Electric Vehicle
CATARC	China Automotive Technology & Research Center
CO_2	Carbon Dioxide
EV	Electric Vehicle
GDP	Gross Domestic Product
GJ	Gigajoules
GHG	Greenhouse Gas
GREET	The Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation Model
HEV	Hybrid Electric Vehicle
ICEV	Internal Combustion Engine Vehicle
JJJ	Jing-Jin-Ji, region consisting of Beijing, Tianjin, and Hebei province
LCA	Life Cycle Assessment
LDPV	Light-Duty Passenger Vehicle
MJ	Megajoules
NCM	Ni-Co-Mn
PRD	Pearl-River Delta Region including Guangdong province
PHEV	Plug-in Electric Vehicle
PM2.5	Particulate Matter 2.5 micrograms per cubic meter
TTW	Tank to Wheel
US	United States
VECC	Vehicle Emission Control Center
WTT	Well to Tank
WTW	Well to Wheel
Yangtze-River-Delta	Yangtze-River-Delta Region consisting of Shanghai, Jiangsu and Zhejiang province

Introduction

Monetary purchasing subsidies, super credits, tax exemptions and local incentives for industry and consumers: China is sparing no efforts in its drive towards market expansion for e-mobility. The motives of China's industrial policy are straightforward, yet environmental protection as a driver is not equally unambiguous. Prevalent coal-fired electricity production is sparking doubts whether an electrification of motorised individual mobility has a positive impact on the climate. A Sino-German cooperation project addresses these issues by assessing the environmental impact of electric vehicles in China.

The purpose of this study was to

1. set up an appropriate Material Flow Model to calculate energy use, greenhouse gas (GHG) emissions and criteria air pollutant emissions of the vehicle fleet in China
2. to design a baseline to describe the reference scenario for the vehicle fleet in China and two alternative scenarios for electric vehicles (EVs)
3. to compare the environmental impacts for the different scenarios on the national level and on the level of selected regions for the applications of EVs in selected transport modes;
4. and to propose recommendations on how electro mobility in China can be introduced to contribute most to climate and environmental protection and forward these recommendations to the Chinese government to assist in the design of the regulatory framework.

Through this project, the comprehensive database to forecast the growth of the automobile stock and its relevant vehicle activity profiles (e.g., total vehicle kilometres travelled and energy and emission profiles were developed. The life-cycle energy and environmental impacts of different PHEV (Plug-in Hybrid Electric Vehicle)/BEV (Battery Electric Vehicle) scenarios were fully assessed for six regional grid systems in China and compared with the current ICEV (Internal Combustion Engine Vehicle)/HEV (Hybrid Electric Vehicle). The PHEV scenarios were examined with different technology designs, battery sizes and specific vehicle driving patterns. The development of natural gas and the wind power impact for EVs in Beijing were considered in this study as well.

1. Methodology and Database

1.1. Methodological concept and system boundary

A variety of models and assessment tools to allow the quantification and assessment of the climate and environmental impacts of a vehicle fleet in China were developed through this project. Figure 1 presents the fundamental methodology, tools and major data inputs and outputs for this project.

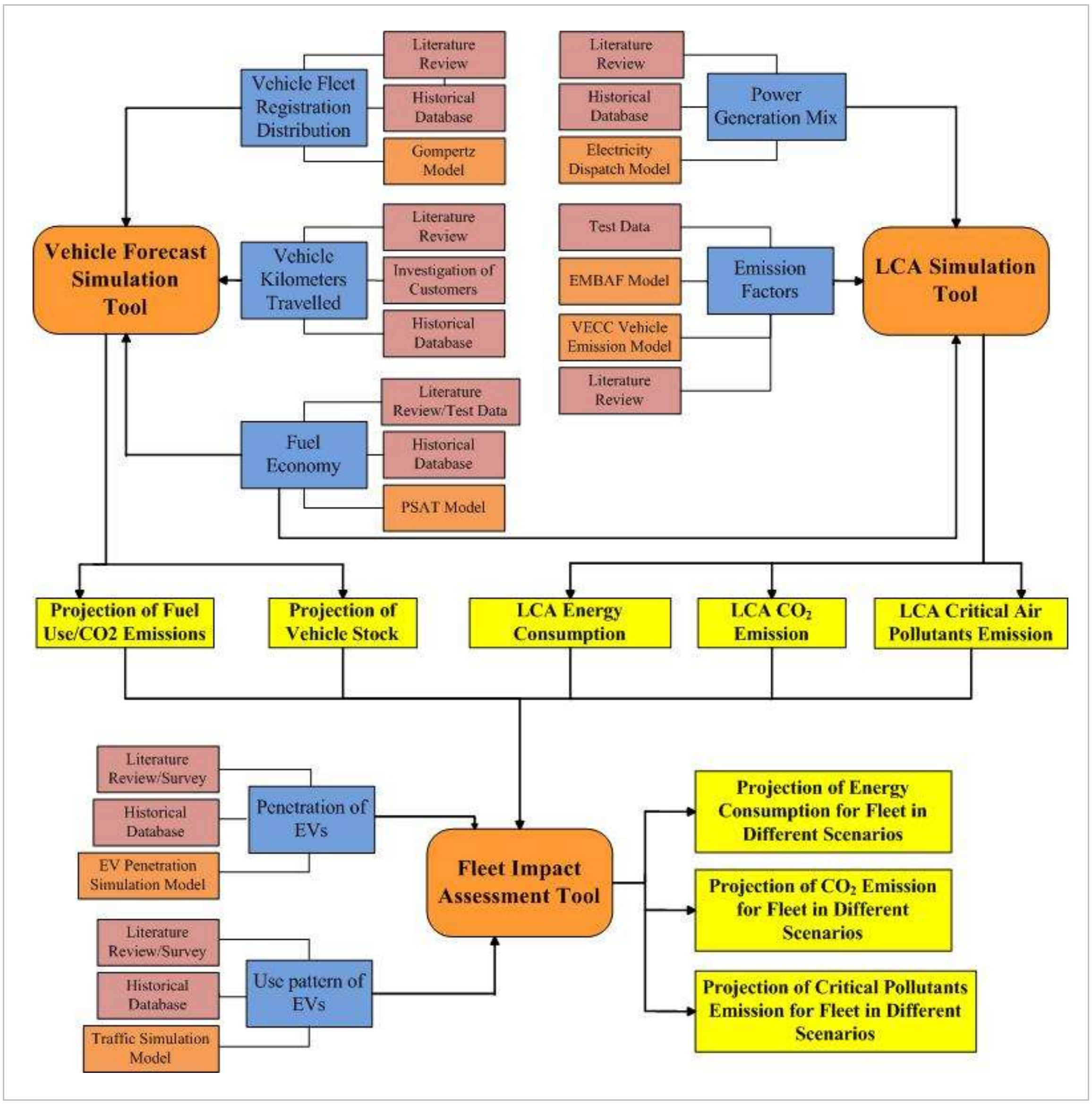

Figure 1: Logistics of methodology fundamentals, tools and major data inputs/outputs

Three major tools -developed by Tsinghua University- were applied, upgraded and updated through this project. They are the Vehicle Fleet Forecast Simulation, the Life-cycle assessment (LCA) Energy and Environment Simulation, and Fleet-based Energy and Environment Impact Assessment Tool (see Figure 1).

The analytical steps are as follows:

5. Set up the scope of vehicle technologies, fuel pathways, time frame, and location for LCA and vehicle forecast simulation;
6. Collect fundamental data to develop the database for LCA, such as power generation mix and vehicle emission test data; and for vehicle forecast simulation, such as vehicle fleet registration records, vehicle kilometres travelled, and fuel economy of different vehicle technologies;
7. Set up various scenarios for vehicle technology shares, generation mix, penetration of electric vehicles, and use patterns based on historical data and literature review;
8. Generate parameters and make necessary assumptions for model simulation;
9. Combine the results of LCA simulation tool and vehicle forecast simulation tool with the penetration and use pattern of EVs to obtain the projection of energy consumption, CO_2 emissions, and critical pollutant emissions for fleet in different scenarios.

1.1.1. System boundaries

The system boundaries include five parts: LCA Boundaries, Technology Boundaries, Regional Boundaries, Temporal Boundaries and Impact Categories.

1.1.1.1. LCA boundaries

The LCA simulation tool is used to simulate and compare the energy consumption, GHG emission factors and major criteria air pollutant emission factors of various electric vehicle technologies (e.g., PHEV, BEV) with conventional gasoline ICEV and conventional HEV. Figure 2 illustrates the major processes included in the fuel cycle (well-to-wheels) analysis and vehicle cycle analysis. In the fuel cycle (well to wheels: WTW) analysis, we focus on the WTT (well-to-tank) stage, which includes pathways, power generation mix, generating efficiency and the TTW (tank-to-wheels) stage, which includes fuel economy and criteria air pollutant emission factors. In the vehicle cycle, we calculate the vehicle life cycle energy consumption and CO_2 emissions. For each auto-material, the analysis includes the elements of mining, refining and processing of raw ore, producing, assembling of parts and vehicle, as well as waste treatment and recycling.

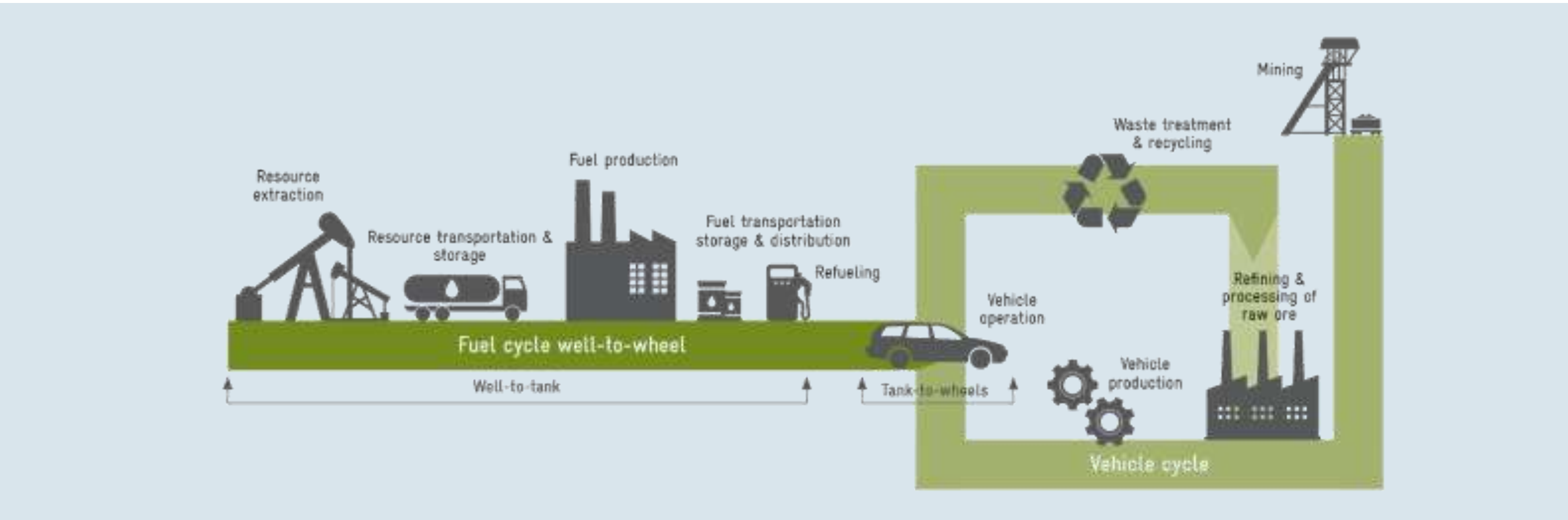

Figure 2: Well-to-Wheel Life Cycle Assessment

In the WTT share of the fuel cycle, the focus is on energy consumption and pollutant emissions in the fuel pathways and process of colliery, power plants or gas station construction are neglected. Only combustion emissions are calculated; some pollutants associated with the mining or other non-combustion activities are neglected (e.g., $PM_{2.5}$). In the TTW stage, the focus is on primary pollutant emissions, and secondary emissions or emission concentrations are neglected in this assessment.

In the vehicle cycle, mass components and the producing processes of each vehicle part (key materials), especially of vehicle batteries are analysed. The material upstream production data is taken from each industry or typical companies in mainland China. This life cycle study only covers the direct life cycle processes and factors based on vehicle, comprising fuels and vehicle materials. Indirect factors, such as charging station, power plant and boiler construction are neglected.

The GREET2012 (GREET1-2012 for WTW analysis and GREET2-2012 for vehicle cycle analysis) model developed by Argonne National Laboratory (ANL) were applied as a platform, and all major input parameters have been updated with the most recent data from Chinese-specific databases mentioned above, to calculate the life cycle energy consumption, CO_2 emissions and criteria air pollutant emissions of PHEV and BEV relative to conventional gasoline vehicle or HEV.

1.1.1.2. Technology boundaries

1. Several technologies, such as Hybrid Electrical Vehicle (HEV), Plug-in Hybrid Electric Vehicle (PHEV) and Battery Electric Vehicles (BEV) are considered. PHEVs with an all-electric range (AER) of 15 km to 50 km were chosen.
2. In this study, the focus is on light duty gasoline vehicle (LDGV). Not considered are heavy duty diesel vehicles or motorcycles.

1.1.1.3. Regional boundaries

We assessed the climate and environmental impacts not only on the national level for China, but also for comparison of different regions of China due to unique regional characteristics, e.g., six regional grid systems. The provinces included in the regions are shown in Table 1.

Table 1 Six regional grid systems and the provinces included

Enterprises	Area	Provinces included
State Grid	North China	Beijing, Tianjin, Hebei, Shanxi, Shandong, Inner Mongolia
	Northeast China	Liaoning, Jilin, Heilongjiang
	East China	Shanghai, Jiangsu, Zhejiang, Anhui, Fujian
	Middle China	Henan, Hubei, Hunan, Jiangxi, Chongqing, Sichuan
	Northwest China	Shaanxi, Gansu, Qinghai, Ningxia, Xinjiang
Southern Power Grid	South China	Guangdong, Guangxi, Yunnan, Guizhou, Hainan

1.1.1.4. Temporal boundaries

The time span for this project is set from 2015 to 2030. We evaluate energy consumption, greenhouse gas (GHG) emissions and criteria air pollutants emissions every five years.

1.1.1.5. Impact categories

1. Energy: Fossil energy use and petroleum energy use
2. CO_2 emissions.
3. Criteria air pollutant emissions: VOC, CO, NO_X, $PM_{2.5}$, and SO_2.

Note: We only evaluate criteria air pollutant emissions in the fuel cycle. The criteria air pollutant emissions database in the vehicle cycle is being developed and is not considered here.

1.2. Fuel-cycle analysis

In the fuel cycle, we simulated fuel pathways, including gasoline and electricity to obtain the detailed data of each fuel process. Then, we focus on the WTT stage, and consider factors such as electricity mix, generation efficiency and pollution emissions from power plants. Electricity mix is the key parameter to evaluate the life cycle results, so we pay attention to the mix diversity of different regions and project the future change in those regions. In the TTW stage, we analyse the current situation of fuel economy and pollutant emissions and predict the future scenarios.

1.2.1. Fuel-cycle analysis methodology and fuel pathways simulation

In this part, we simulated gasoline and electricity fuel pathways for ICEV, HEV, PHEV and BEVs. The gasoline pathway includes the oil extraction, storage, refining and fuel transportation. The electricity pathway concentrates on the thermal power generation.

1.2.1.1. Gasoline pathway

Table 11Figure 3Figure 3 Table 11Table 11represents the gasoline pathway and includes fuel distribution. In this process, we focus on the crude recovery efficiency, gasoline refining efficiency, share and distance of various fuel-transport modes such as tanker, railway, barge, and pipeline. Local oil recovery efficiency is obtained from the China Energy Statistical Yearbook. Based on the proportion of imported oil, we can calculate national oil recovery efficiency. Gasoline refining efficiency is obtained from published literature. Transportation mode's proportion and distance is obtained from the China transportation yearbook and other literature. The oil extraction and gasoline refining efficiency and transportation modes data are listed in Table 2: .

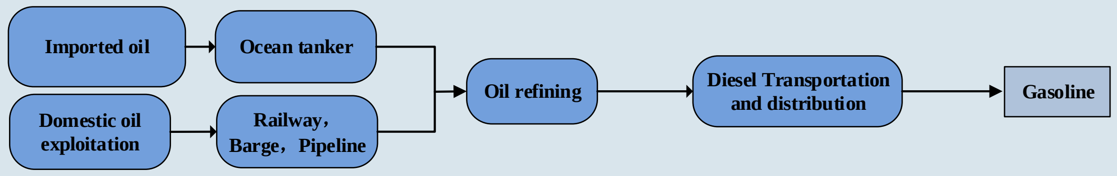

Figure 3: Gasoline pathway

1.2.1.2. Electricity pathway

Figure 4 represents the electricity pathway. The major electricity power generation are coal power, natural gas power and hydro power in China. In 2010, the coal-fire power takes 79% in the power generation mix, followed by hydro power (16%). Other power generation take a small proportion. The coal mining efficiency and transportation modes data are listed in Table 2: .

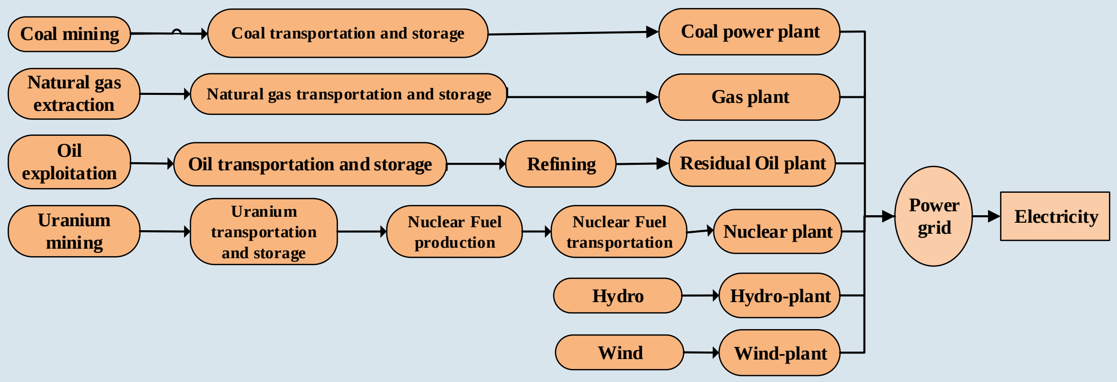

Figure 4: Electricity pathway

Table 2: Energy efficiency, shares of process fuels and transportation modes for two pathways

		Crude oil	Coal	Gasoline
Efficiency		Recovery	Mining	Refining
		Domestic: 91% Oversea: 98%	97.0%	91.0%
Shares of process fuels	Crude oil	22%		3%
	Residual oil	1%		5%
	Diesel fuel	9%	3%	1%
	Natural gas	44%	0%	9%
	Coal	4%	79%	22%

	Electricity	14%		15%		15%	
	Refinery still gas	2%				34%	
	Others and Loss	4%		2%		11%	
Transportation and Distribution		Share	Distance/km	Share	Distance/km	Share	Distance/km
	Ocean Tanker	44%	13900				
	Pipeline	22%	2500			20%	373
	Rail	29%	940	35%	640	40%	559
	Barge	16%	700	12%	1250	25%	746
	Truck			53%	180		

1.2.2. Database of electricity generation sector

Generation mix, efficiency and emission factors of power plants are key parameters for analysing life cycle energy consumption, CO_2 emissions and pollutant emissions of electric vehicles. Based on the electric power yearbook and the projections of electricity generation mix by researchers, this chapter developed the projections of generation mix, efficiency and emission database on national and regional levels. We selected six regions in China: Jing-Jin-Ji (JJJ), Yangtze River Delta (YRD), Pearl River Delta (PRD), Northeast China, Central China and Northwest China.

1.2.2.1. Average generation mix in China and six regions

Over the past 30 years, the electric power sector experienced a rapid increase in China. Figure 5 presents the electricity production of China from 1980 to 2010. The electricity generation during 1980 was 300 billion kWh, which climbed to 4,200 billion kWh in 2010, over a 14 times increased in 30 years and with an average annual growth rate of 9%. According to the projections by senior experts in the power sector, the total generation of China will be 7 to 11 thousand billion kWh in 2030.

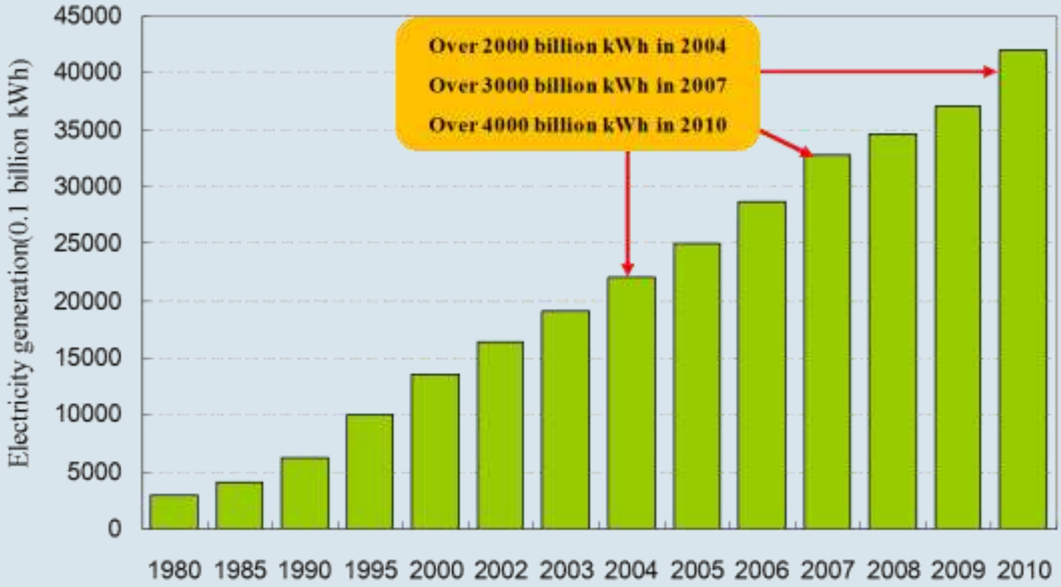

Figure 5: Electricity generation development of China, 1980-2010

When measured by generating capacity, electric power resources in China are quite unevenly distributed. Most large power plants are located in the central and eastern areas of China which are relatively developed regions. While provinces located in the western part like Xinjiang, Qinghai, and Gansu have large land areas, they do not have many power plants.

We divided China's power grid system into six independent regional power grids, excluding Tibet, according to the State Grid and the Southern Power Grid administration districts, which are the main state-owned power grid enterprises, as shown in Table 1.

In 2010, the average electricity generation mix on the national and regional levels in China are shown in Figure 6. On the national average level, coal power contributes 79% of total power generation, while hydro power occupies 16% and the share of nuclear, natural gas and other power sources are less than 2%. Further, considering the regional stage, the generation mix differs significantly because of the uneven distribution of natural resources. Coal resources are intensive in North China so the contribution ratio of coal-based generation to total power in that area reaches beyond 95%. Coal-based generation only accounts for 60% of total generation mix in South China, where hydro power takes up 25%. Similar conditions occur in Middle China and Northwest China, where major rivers are located in these regions. Nuclear power is concentrated in South and East China, both accounting for 4% while wind power is mostly located in the north, making up 5% of the total mix in Northeast China. The development of the electric vehicle and its environmental impacts are strongly related to the regional distribution of power generation.

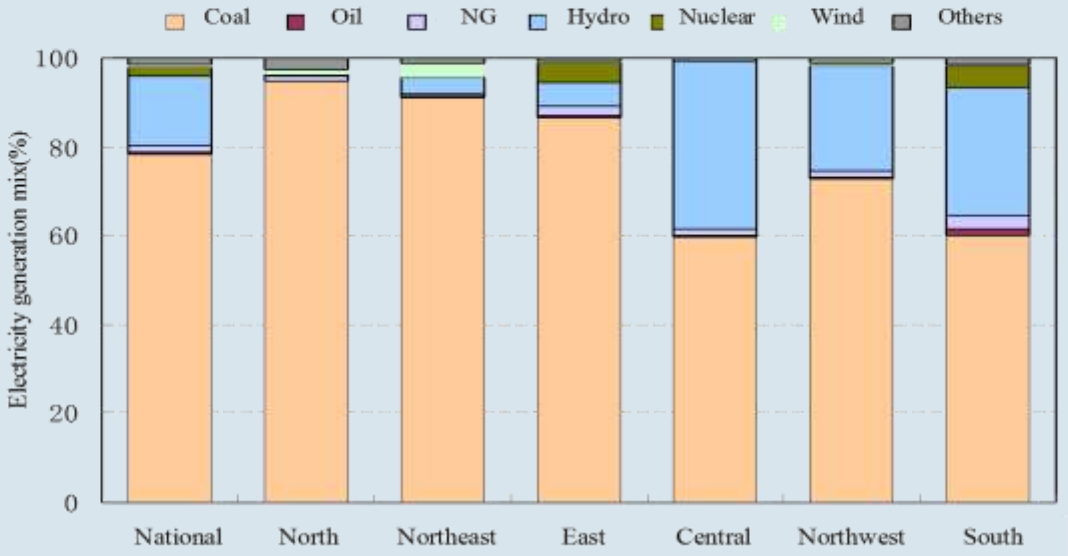

Figure 6: Electricity generation mix at national and regional levels in China, 2010

The national average generation mix from 2005 to 2010 were obtained from the China electric power yearbook, while the projection of the future generation mix from our literature review has significant uncertainties, as shown in Figure 7.

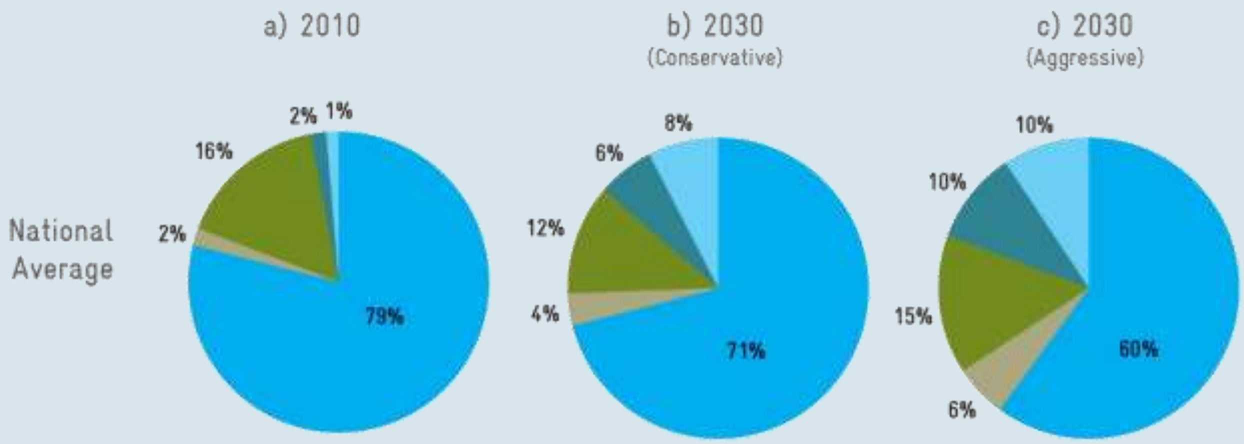

Figure 7: Generation mix in China, 2010-2030

Based on literature research, this study set two different scenarios for the average generation mix of China in 2030, as shown in Table 3. Compared to the ambitious scenario, the conservative scenario forecasts that the coal-based electricity share would remain relatively high and the promotion of clean energy such as hydropower, nuclear and wind power generation would be moderate.

Table 3: Projection of national average generation mix in China, 2030

Scenarios	Researches	Coal	Hydro	Nuclear	Natural gas	Wind
Conservative	Wu (2007)	71.1%	12.4%	8.7%	4.7%	3.5%
	Jiang (2011Baseline)	71.5%	14.0%	5.2%	1.7%	
	IEA (International Energy Agency) (2007Reference)	78.0%				
	CEC (2012)				4.0%	5.0%
	This study	**71.0%**	**12.0%**	**6.0%**	**3.5%**	**5.0%**
Ambitious	CAE (2011)	50.0%	7.0%	7.0%	7.0%	
	IEA (2007High)	64.0%	17.0%	6.0%	6.0%	
	Wu (2012)	55.9%	13.9%	18.1%	5.0%	5.9%
	This study	**60.0%**	**15.0%**	**10.0%**	**5.5%**	**6.0%**

On the regional level, we also forecasted two scenarios: conservative and ambitious. The trends of coal-based and natural gas electricity generation are basically consistent with the national level. For the other generation resources, the projection was based on the primary energy structure in those regions.

As Figure 17 shows, coal-based power accounts for 88% and 77% of the total generation mix in JJJ in the conservative and ambitious scenario in 2030, respectively. In YRD, the share of coal-based electricity generation will be 76% and 64% in the conservative and ambitious scenario, respectively. While in PRD where coal is less used (55%) although used more than that of the present US national average mix (45%), nearly 40% of the generation mix is from hydro and nuclear power in the ambitious scenario. In Northeast, the share of coal power takes 81% and 70% under the conservative and ambitious scenario by 2030, as high as in JJJ. In Central China, the hydro power accounts about 40%, the coal power takes 52% and 42% under the conservative and ambitious scenario by 2030. For Northwest China, the share of coal power takes 66% and 55% under the conservative and ambitious scenario by 2030.

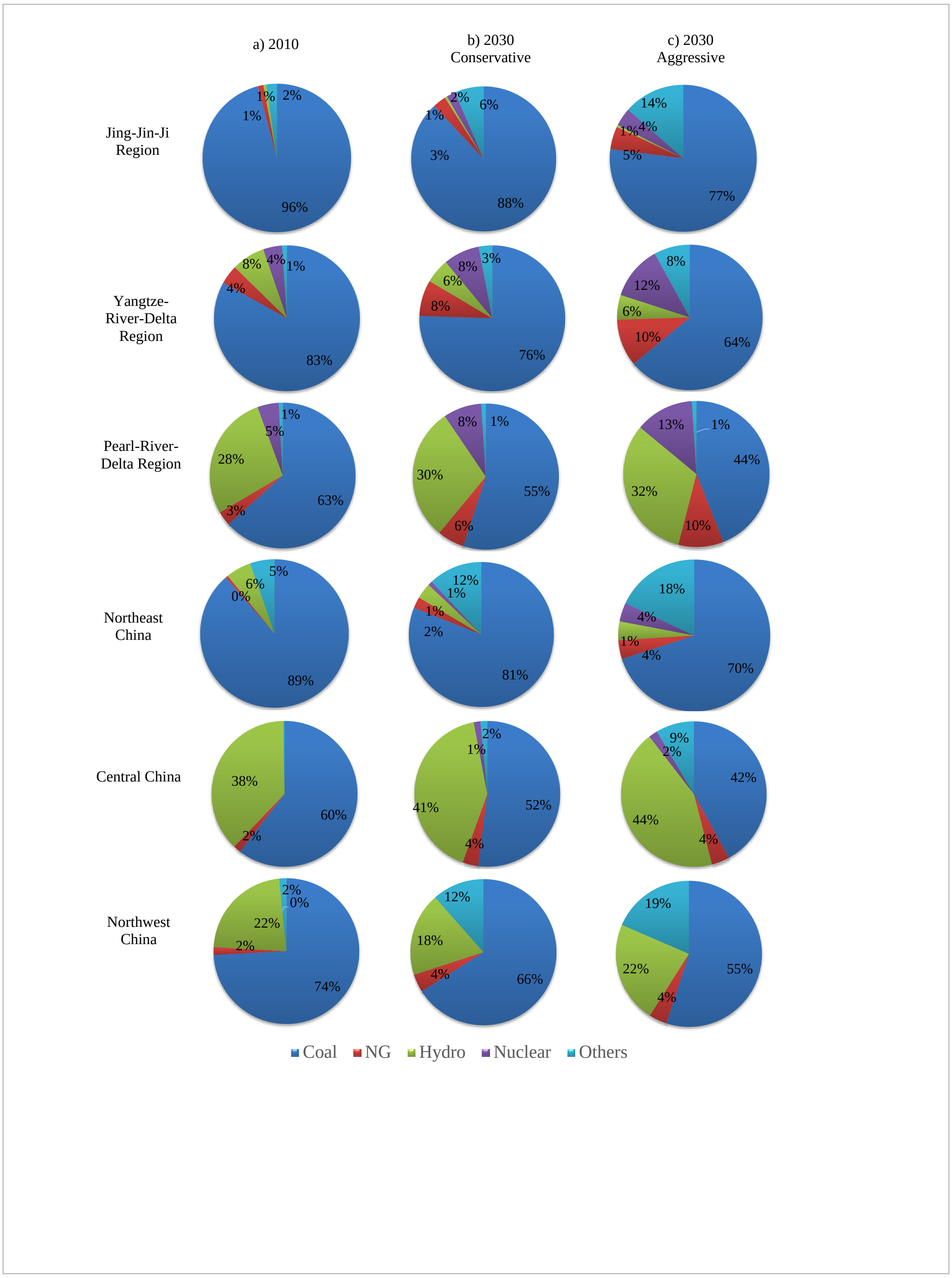

Figure 8: Generation mix in six regions, 2010-2030

1.2.2.2. Electricity generation efficiency for power plants in China

Electricity generating efficiency can significantly impact the energy consumption and CO_2 emissions of thermal power plants. This section concentrated on the generating efficiency of coal-fired power plants in China, which can be calculated by equation 1-1:

$$\omega = \frac{P \times m}{T \times n} \times 100\% \qquad (1\text{-}1)$$

Where: ω = the energy efficiency of coal in unit of %;

P = the coal-fired electricity generation in unit of kWh;

M = the conversion factor from electricity to heat in J/kWh;

T = the total consumption of coal used for electricity generating in unit of kg coal-equivalent;

n = the average low calorific value of standard coal in unit of J/kg.

There is much room for improvement of the electricity generating efficiency of China. Both the middle and long term program of energy conservation and the National policy for the energy saving technology explicitly proposed the developing direction and technology selection of the future electricity generating technology of China. This included a gradual closure of the electricity generation units with medium and small generation capacity, vigorously development of SC (super critical) and USC (ultra super critical) units with capacity of more than 300MW, and promotion of high-efficiency and clean coal-fired units and large combined cycle units such as Integrated Gasification Combined Cycle (IGCC). It was reported by IEA that the generating efficiency of SC and SSC units are 39% and 42%, respectively, which are similar to the projection by several Chinese domestic researchers. As the development continues of the power industry and generating technology in China, SC and SSC units will dominate the electricity generating technology market. Referring to the IEA's report, this study forecasted the electricity generation from SC and USC would account for 55% of total generation from coal-fired power plants in China by 2030. For the IGCC technology, this study adopted the assumption by IEA, and forecasted the share of IGCC in total coal-fired generation will be 10% in 2030, with the default generating efficiency in GREET 2012 model.

The average generating efficiency of China's coal-fired power plants in 2030 is estimated to be 40% based on the projection of market share and efficiency of each generating technology, which has a remarkable improvement when compared to the current 34% average efficiency. This study forecasted the generating efficiency for the years between 2010 and 2030 by linear interpolation method. For the regional generating efficiency, this study assumed the efficiency is equal to the national average level without considering the regional discrepancy.

1.2.2.3. Emission factors of air pollutants from power plants in China

The equation for calculating the projected emission factors of air pollutants emitted from thermal power plants is as below:

Actual emission factors = Emission factors without control×Penetration rate of control technology×(1-removal efficiency of control technology)

The control technologies for CO and VOC emitted from thermal power plants were not included because their emissions are relatively small. This study adopted the equation above to forecast the emission factors of SO_2, PM, and NO_X. The emission factors without control refer to the emission factors under no control technologies, which are strongly related to fuel type and combustor structure. This study chose the emission factors in 2005 as baseline. At first, the total emissions from power plants and total electricity generation in 2005 were used to calculate the average emission factors in 2005 in units of g/kWh. Then the average emission factor without control of each pollutant can be figured out by using the penetration rate and removal efficiency of each control technology in 2005. It was assumed that the emission factors without control would not change with time. Finally, the actual emission factor from control of SO_2, PM, and NO_X from power plants in 2010-2030 are calculated by using the emission factors without control, projection of penetration rate and removal efficiency of control technologies in specific model year.

1. **Control Technologies of SO_2**

   Desulfurization before combustion and flue gas desulfurization (FGD) are two main classifications for the control technologies of the power plant emissions of SO_2. The former (pre-combustion desulfurization) cannot meet the current stringent emission control requirements because of limited removal efficiency. FGD technologies are divided into wet, dry, and semi-dry sub-categories according to the process characteristics. Among them, wet FGD has the best desulfurization efficiency, which is the major applied technology in the FGD projects in power plants currently.

2. **Control Technologies of PM**

   Control technologies for power plant emissions of PM differ with different kinds of boilers: the control technologies for grated-fired furnace include wet scrubber (WET) and cyclone (CYC), which provide relatively modest control; the control technologies for pulverized coal furnace are fabric filter (FF), electrostatic precipitator (ESP), and WET.

3. **Control Technologies of NO_X**

   The main control technologies for power plant emissions of NO_X are low nitrogen burning (LNB) and flue gas denitration. LNB technologies including low nitrogen combustor, air staging combustion technology and re-burning technology controls the production of NO_X during the combustion process, with NO_X removal efficiency between 30% and 40%. This broadly applied technology is technically simple and cost-effective because the combustion system and boiler structure does not have to be changed; the combustor is replaced. SCR is the most popular technology applied around the world currently among flue gas denitration technologies, with more than 90% removal rate. However, the SCR has high investment and operation costs, which makes it difficult to widely apply to existing power plants. Based on literature research, this study forecasted the future penetration rate and removal efficiency of different NO_X control technologies for power plants.

4. **Projected Results of Emission Control Technologies for Power Plants in China**

   By using the forecasted penetration rate and removal efficiency of the control technologies in Table 4 (developed by Tsinghua University), the power plant emission factors for the next 20 years were calculated, which is shown in Figure 9.

Table 4: Removal efficiency and penetration rate of emission control technologies of power plants in China, 2010-2030

		Removal efficiency of control technologies		**Penetration rate of control technologies**				
		Equipment	Removal efficiency	2010	2015	2020	2025	2030
SO_2	Sulfur removal	FGD	90%	80%	90%	95%	98%	100%
		NON	0%	20%	10%	5%	2%	0%
NO_X	Nitrogen removal	LNB	55%	71%	60%	40%	30%	20%
		SCR	85%	5%	20%	55%	65%	75%
		NON	0%	24%	20%	5%	5%	5%
$PM_{2.5}$	Dust removal for pulverized coal furnace	FF	99.5%	5%	10%	15%	20%	30%
		ESP	93.6%	95%	90%	85%	80%	70%
		WET	50.0%	0%	0%	0%	0%	0%
	Dust removal for pulverized coal furnace	WET	50.0%	60%	100%	100%	100%	100%
		CYC	10.0%	40%	0%	0%	0%	0%

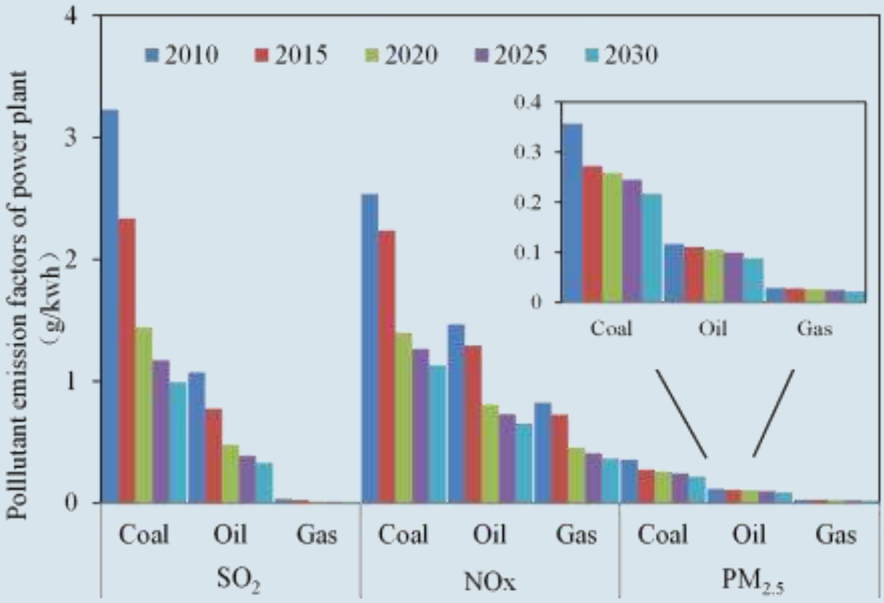

Figure 9: Emission factors of air pollutants from power plants in China, 2010-2030

1.2.3. Database of fuel economy and air pollutant emissions on the vehicle operation stage

1.2.3.1. Fuel economy of vehicle driving stage

1. **Fuel economy for conventional light duty gasoline vehicles**

The indicator for the vehicle fuel economy is the fuel consumption per hundred kilometres travelled in units of L/100 km in China. In 2004, China started to develop fuel consumption standards for light duty passenger vehicles (LDPVs). Subsequently, a series of standards were developed and implemented, which are shown in Table 5.

Table 5: Chinese fuel consumption standards for new passenger cars

Vehicle curb mass(kg)	Standards for regular vehicles(L/100km)			Standards for special-featured vehicles(L/100km)		
	Phase I	Phase II	Phase III	Phase I	Phase II	Phase III
CM≤750	7.2	6.2	5.2	7.6	6.6	5.6
750<CM≤865	7.2	6.5	5.5	7.6	6.9	5.9
865<CM≤980	7.7	7.0	5.8	8.2	7.4	6.2
980<CM≤1090	8.3	7.5	6.1	8.8	8.0	6.5
1090<CM≤1205	8.9	8.1	6.5	9.4	8.6	6.8
1205<CM≤1320	9.5	8.6	6.9	10.1	9.1	7.2
1320<CM≤1430	10.1	9.2	7.3	10.7	9.8	7.6
1430<CM≤1540	10.7	9.7	7.7	11.3	10.3	8.0
1540<CM≤1660	11.3	10.2	8.1	12.0	10.8	8.4
1660<CM≤1770	11.9	10.7	8.5	12.6	11.3	8.8
1770<CM≤1880	12.4	11.1	8.9	13.1	11.8	9.2
1880<CM≤2000	12.8	11.5	9.3	13.6	12.2	9.6
2000<CM≤2110	13.2	11.9	9.7	14.0	12.6	10.1
2110<CM≤2280	13.7	12.3	10.1	14.5	13.0	10.6
2280<CM≤2510	14.6	13.1	10.8	15.5	13.9	11.2
2510<CM	15.5	13.9	11.5	16.1	14.7	11.9

Wang et al. demonstrated that the average fuel economy of light duty passenger cars in China was 8.1 L/100 km in 2006, which was lower by 12% than that of 2002. In the assessment report for the implementation of the National fuel consumption standard for light duty passenger cars, the average fuel economy of 2002 and 2006 was 9.1 L/100 km and 8.1 L/100 km, respectively, which was calculated through the corporate-average fuel consumption (CAFC) of 34 manufacturers and their corresponding sales numbers. Wagner and Huo's studies pointed out that the fuel economy of LDPV improved to 7.8-7.9 L/100 km in 2009. The research report of the CAFC development of the Chinese passenger car manufacturers showed the fuel economy was 7.8 L/100 km in 2009, which was almost equal to the result of the two former research investigations. This study used the sales number of the top 60 vehicle models and their fuel economy publicized by the Ministry of Industry and Information Technology (MIIT) to calculate the sales-weighted average fuel economy from the top 10 to the top 60 in 2010. As the LDPV market share increased with more high-sale vehicle models added, the average fuel economy became steady at 7.54 L/100 km after the top 50, which is shown in Figure 10. Therefore, this study adopted 7.54 L/100 km as the average fuel economy of China's newly sold LDPVs in 2010.

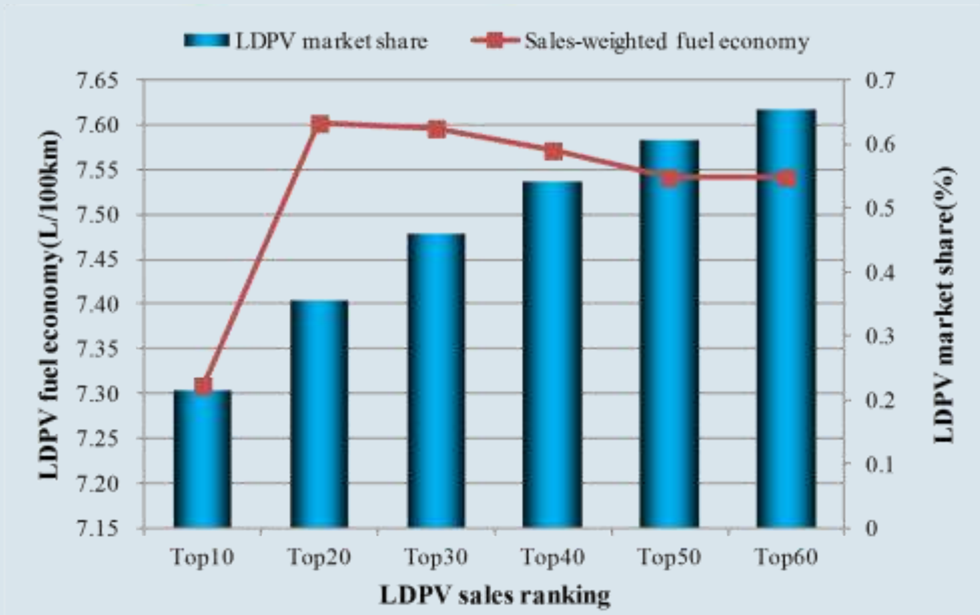

Figure 10: Sales-weighted fuel economy and market share of the top 60th sales of LDPV models in China

In December 2012, the Chinese Phase III fuel consumption standard of new passenger vehicles was implemented, which is more stringent than the former standards. The gradually tightened standards are shown in Table 5. The target limit of average fuel economy of conventional LDPV is 6.9 L/100 km in the Planning for the development of energy-saving and new energy automobile industry 2012, which was issued by the State Council of China. The energy-saving effects and targets of passenger cars projected by the Society of Automotive Engineers in China in the Strategy research of Chinese automotive energy-saving are shown in Table 6.

Two scenarios were projected because the energy-saving effects of the advanced technologies are intervals, not specific values: the conservative scenario employed the decreasing rate from 2010 to 2015, and extended it to 5.6 L/100 km in 2030 with steady decrease; the other scenario used the average value of the intervals to calculate the recommended fuel economy of 2020 and 2030, which are 5.6 and 4.8 L/100 km, respectively.

Table 6: Energy saving effect and penetration target of energy saving technologies of passenger car

Main technologies		Energy saving effects	Current situation	Targets in 2020
Advanced engine technologies	Turbocharger gasoline engine	1.8% ~ 4.8%	7%	40%
	Gasoline Direct Injection	10% ~ 20%	7%	40%
	Variable valve timing	2% ~ 3%	25%	100%
	Variable valve lift	1% ~ 3%	25%	100%
	Lower engine friction loss	2% ~ 5%	Quite low	50%
	Idle stop	5% ~ 8%	Quite low	100%
	Cylinder fuel-cut technologies	3.9% ~ 5.5%	0%	2%
Advanced transmission technologies	Lifting component performance	3% ~ 5%	Quite low	50%
	Multiple gears	1.4% ~ 3.4%	6MT: 1% 6/7/8AT: 12%	100%
	DCT	2.7% ~ 7.5%	2%	25%
Other advanced technologies	Continuously variable transmission	0.7% ~ 2.0%	4%	5%
	Lightweight	2% ~ 8%	Quite low	Average weight lighted: 15%
	Hybridization (exclude idle stop)	10% ~ 40%	Quite low	20%
	Electric power steering	1% ~ 2%	Quite low	100%
	Monitoring system to lower air resistance	2% ~ 3%	Quite low	75%
	Low rolling-resistance tire	1% ~ 2%	Quite low	100%
Small displacement passenger vehicle		20%	65%	1.6L and even lower: 80%
Diesel vehicle		20%	0.80%	20%
Green driving/green maintenance		15%	5%	75%
Smart transport		15%		

The fuel consumption values mentioned above are all based on laboratory testing under specific driving cycles, which are often below the real-world driving fuel consumption. Lin and Huo's studies illustrated that the fuel economy under real-world condition is lower by about 15% than that under laboratory testing conditions. Therefore, the two projections of LDPV fuel economy from 2010 to 2030 are shown in Table 7, with real-world correction.

Table 7: Projection of fuel economy of light duty passenger vehicles in China

Fuel economy in L/100km	2010	2015	2020	2025	2030
Conservative	8.7	7.9	7.3	6.6	6.4
Recommended	8.7	7.9	6.5	6.0	5.5

2. Fuel economy improvements for new energy vehicles

For HEV, PHEV and BEV, fuel economy was expressed by the improvement rate compared to the conventional vehicles. Through researching the HEV models sold in the US market, Bennion et al.'s study showed the fuel economy of HEV was higher by 37%-42% than conventional vehicles, which is very close to the default 40% of GREET2012 model. Therefore, our study developed two scenarios for the fuel economy of HEV in China: the conservative scenario set the improvement rate of HEV at 30% compared to corresponding conventional vehicles; and the recommended scenario at 50% improvement.

For BEVs, this study also developed two scenarios for the improvement rate compared to conventional vehicles. As shown in Figure 11, the conservative scenario considered the improvement rate is 225% based on comparisons between BEVs and the corresponding ICEVs of three brands. Taking the uncertainty of future policy into account, the improvement rate of the recommended scenario is 250%, which is the average of the conservative scenario and the default 275% of GREET2012 model.

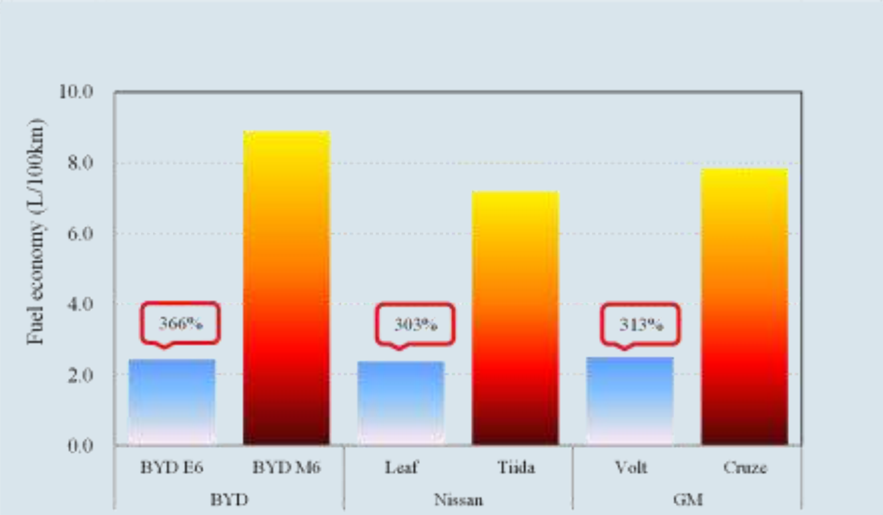

Figure 11: Comparisons of fuel economy between BEVs and the corresponding ICEVs, 2010

The fuel economy of PHEV is strongly related to its all-electric range (AER). The higher AER, the longer electricity-driving distance and less consumption of petroleum fuels. In addition, the daily travel mileage also significantly impacts the fuel economy of PHEVs. Shorter daily travel means more distances can be covered by electricity-driving pattern, leading to better combined fuel economy. After investigating the user driving patterns in several cities of China, the daily travel distance of LDPV of this study is defined as 50 km, so we select PHEV50 (AER=50 km) as one PHEV model. On the other hand, typical trips in the urban areas from home to the work place are usually around 15-20 km, which makes PHEV15 (AER=15 km) as the other PHEV option in this study. The driving strategy of a PHEV depends on the state of charge (SOC), which means the remaining capacity of battery. When the SOC is high, the PHEV will be mostly driven in the charge-depleting (CD) mode, in this case the electricity from battery is the primary energy for propulsion and the petroleum fuel is used as complement for fast accelerations. If the SOC is relatively low, the PHEV will be driven substantially on charge-sustaining (CS) mode; in this case the internal combustion engine is used to drive the vehicle primarily, just like an HEV. This study employed the results of Elgowainy's research on PHEV15 and PHEV50 due to the lack of domestic research on fuel economy of PHEVs. Similar to the HEV and BEV cases, the fuel economy improvement rate of PHEVs has two scenarios. For the conservative scenario of PHEV50, the improvement rate is 170% and 10% on CD and CS mode, respectively. In the recommended scenario, the improvement rate is 190% and 30% on CD mode and CS mode, respectively. For the PHEV15, the improvement rate on CD mode and CS mode is 80% and 50%, respectively, in the conservative scenario. In the recommended scenario, the improvement rate on CD mode and CS mode is 90% and 70%, respectively. For the PHEV50, the improvement rate is 170% / 190% on CD mode and 10% / 30% on CS mode under conservative scenario and recommended scenario, respectively. In addition, the improvement rates already included a real-world correcting factor of 0.7. The fuel economy of ICEV, HEV, PHEV and BEV are shown in Figure 12.

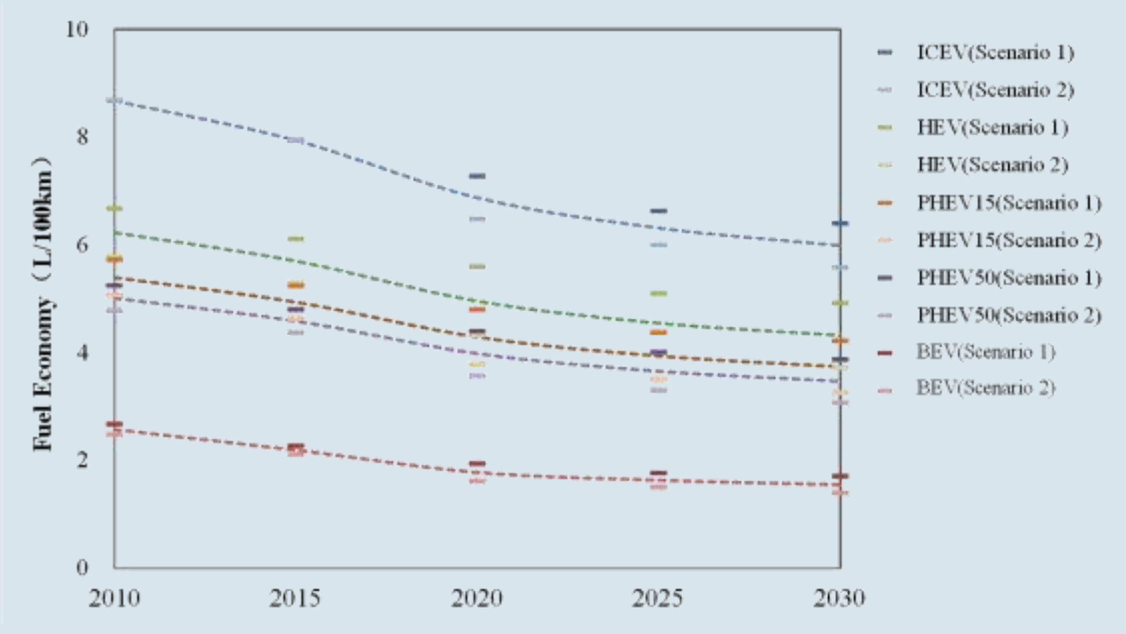

Figure 12: Fuel economy projections of ICEV, HEV, PHEV, and BEV, 2010-2030

To combine the energy consumption of CD and CS mode, we need to assume the share of CD mode as a proportion of the total daily travel mileage, hereafter refers to as the "utility factor"

(UF). In this study, we adopted the default UF values of GREET model for different types of PHEVs, which are based on the 2001 NHTS (National Household Travel Survey) data of the United States. Because the average annual VMT (Vehicle Mileage Travelled) in China (~19,000 km) is close to that of the United States (~12,000 mi), according to our previous studies.

1.2.3.2. Emission database on vehicle driving stage

1. Emission factors for conventional light duty gasoline vehicles

The estimation of the emission factors of conventional light duty gasoline vehicles was based on long-term and large-scale collection of laboratory and real-world testing data, including certified testing data of about 1,500 new vehicles, dynamometer testing data of 200 in-use vehicles, emission testing data under different driving conditions of 400 tests, and real-world emission testing data of 50 trips.

At first, the zero-mile emission factors obtained from the certified data of new vehicles and testing data of in-use vehicles were combined to obtain the impacts of vehicle miles travelled on vehicle emissions. Then, we analysed and estimated the modification factor for vehicle emission degradation to obtain the basic emission factors. Based on the analysis of testing emission data under different driving speeds, we established the speed-correcting curve for emission factors.

In addition, the emissions of light duty vehicles can be affected by other factors, such as fuel quality, ambient temperature, proportion of high emission vehicles, and use pattern of air conditioning. These influential factors correspond to different correcting modules to the basic emission factors, among which some correcting factors referred to the results of previous domestic and foreign research on urban vehicle emission factor models, such as MOBILE, COPERT and MOVES.

The equations for the calculation of light duty vehicle emission factors are:

$$EF_{total} = EF_{run} + EF_{cold\ start} + EFeva \qquad (1\text{-}2)$$

$$EF_{run} = (ZML_{NEDC} + DR \bullet M)\ C_S \bullet C_A \qquad (1\text{-}3)$$

Equation 1-2 refers to the integrated emission factor, which is the sum of operation emissions, cold start emissions, and volatile emissions.
Equation 1-3 refers to the operation emission factor, where ZML_{NEDC} is the zero-mile emission level, DR is the degradation rate of vehicle emissions, M is the cumulative travel distance, C_S is the correcting factor of average speed, and C_A is the integrated correcting factor including use of air conditioning, fuel quality, environmental temperature, and proportion of high emission vehicles.

Figure 13 shows the integrated emission factors of light duty gasoline vehicle (LDGV) under different control levels. For convenient comparison, the cumulative mileages and the average speeds of LDGVs were all set to 100,000 km and 25 km/h, respectively, with vehicle emission control level consistent with its fuel quality standard. In addition, it was assumed that the improvement rate of the unimplemented China VI compared to China V standard is equal to the improvement rate of Euro VI to Euro V. More accurate emission factors will be updated in the upcoming studies.

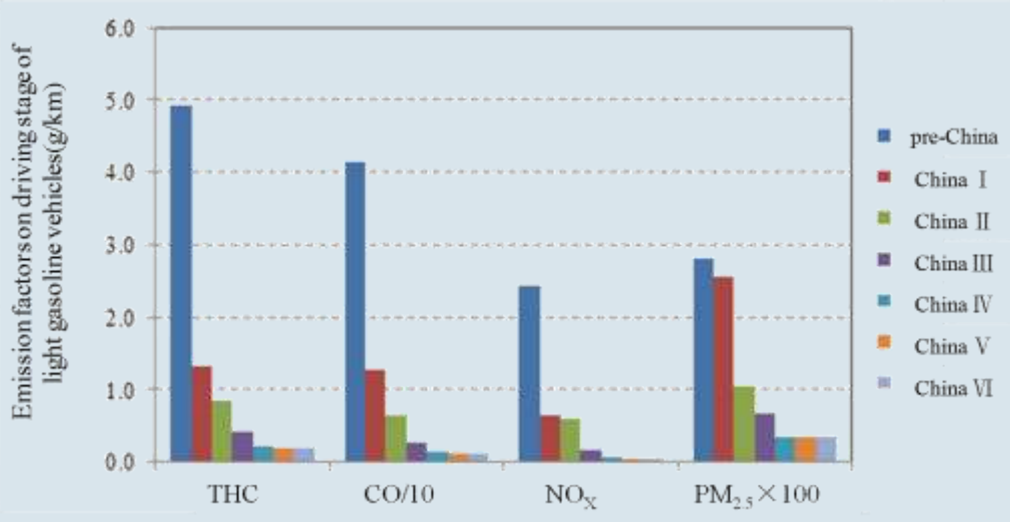

Figure 13: Emission factors on driving stage of light duty gasoline vehicles

2. Emission factor for new energy vehicles

The tests conducted in Macau showed a remarkable emission reduction of HEVs compared to conventional gasoline cars under China IV standard. The reduction rates of VOC, CO, NO_X, and $PM_{2.5}$ are all between 60% and 70%, as in the ambitious scenario for the emission reduction of HEV. While the conservative scenario considered the reduction rates between 15% and 30% referring to the emission reduction rates of China V LDGVs compared to China IV LDGVs. For PHEV, the CD mode is similar to BEV, which has no emissions in the driving stage; while the emission reduction rate of CS mode is equal to HEV. Then, the reduction rates on the CD mode and CS mode are combined by using a mileage split factor to address the weighted-average emission reduction rate for PHEVs. For BEV, there is no emission during vehicle driving stage. Figure 14 presents the emission reduction rates of HEV, and PHEV compared to conventional China IV LDGVs, as a baseline.

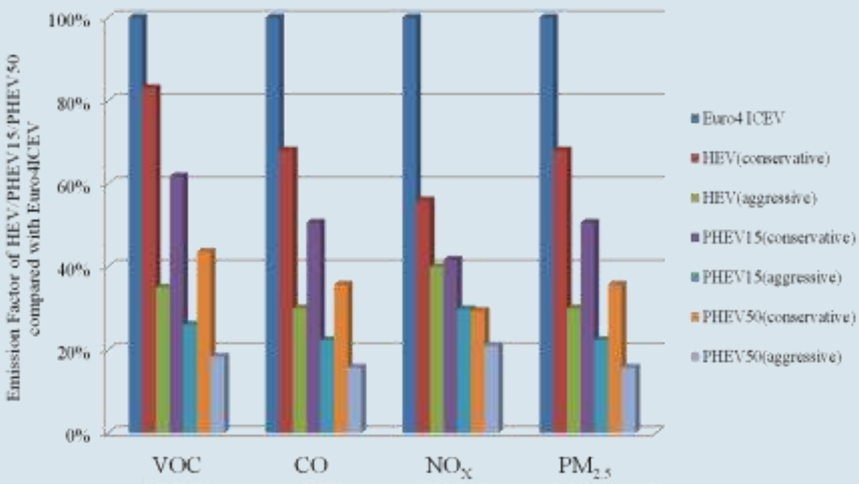

Figure 14: Emission reduction rate of HEV and PHEV compared with light duty gasoline vehicles under China IV emission standard

1.3. Material-cycle analysis

1.3.1. Vehicle components

In this study, vehicles are divided into three major parts, 1) fixed parts, 2) core parts, and 3) changing parts. The battery module, as defined, is a core part of electric vehicles and used as low voltage power in ICEVs. All other relevant electric modules (e.g., controller), and engine and transmission are called changing parts. This is because during the vehicle electrification, these vehicle components will change substantially. Body and chassis are fixed parts for ICEV and all electric vehicles. Figure 15 describes the major vehicle components of a BEV (35 kWh NCM battery module as an example), PHEV50 (AER=50km), HEV and ICEV. The fixed parts are 948 kg in total. The component maps of key parts are listed in Table 8, in which ferrous (steel and cast iron) and aluminium are the most used materials.

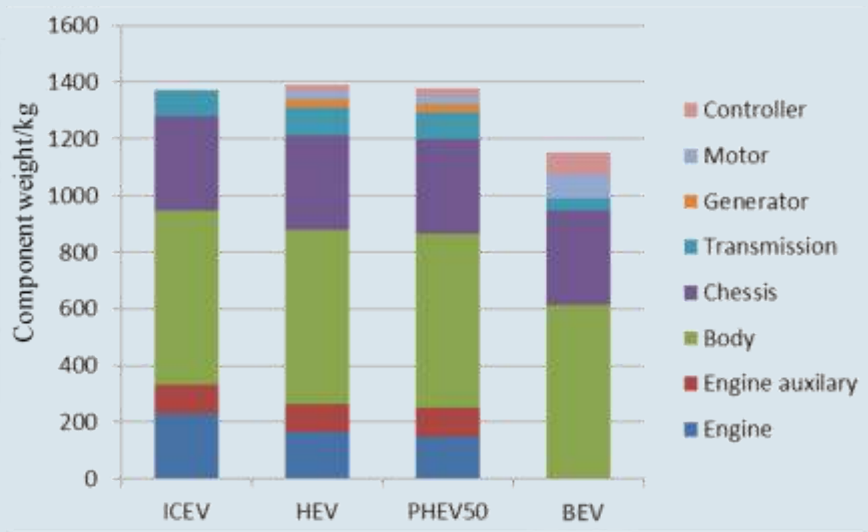

Figure 15: Weight share of major components for different vehicle fleets (except batteries)

Table 8: Component compositions of each part of vehicle

	Engine	Engine auxiliary	Motor	Body	Chassis	Controller	Transmission (ICEV)	Transmission (BEV)
Steel	17%	91%	36%	68%	84%	5%	30%	61%
Cast iron	41%	0%			7%		30%	19%
Plastic	11%	5%		18%	2%	24%	5%	
Aluminium	25%	0%	36%	1%	1%	47%	30%	20%
Rubber	4%	0%		1%	4%	4%	5%	
Copper	2%	4%	28%	2%	1%	8%		
Glass				7%				
Others				4%	1%	12%		

Power battery design is largely dependent on energy demand, or AER of different vehicle fleets. We use two typical AERs of 50 km and 150 km for PHEV and BEV, respectively, which referred to the Chinese local electric vehicles parameters. HEVs are equipped by parallel power flow structure, in which electricity goes to motor prior to the battery which means lower battery demand. If equipped with Li-ion batteries, the curb weight would add 300-500 kg to achieve 35 kWh power demands, as seen in Figure 16. Pb-ac battery is only used as low voltage power for auxiliary systems, which weighted for 20 kg. Ni-MH battery needs 900 kg to meet the same power demand to Li-ion.

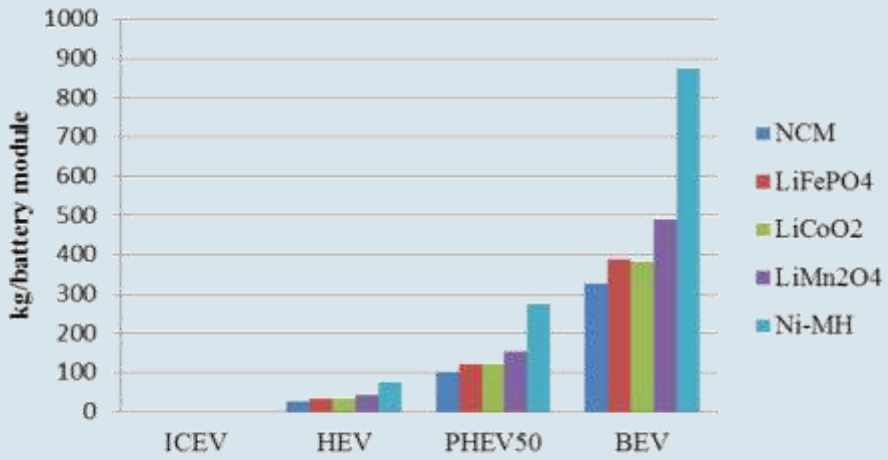

Figure 16: Battery energy demand of different vehicle fleets, 2010

1.3.2. Composition map for battery and other key parts

This research focuses on four major Li-ion batteries: $LiCoO_2$, $LiMn_2O_4$, $LiFePO_4$ and Ni-Co-Mn (NCM). For comparison, Ni-MH battery and Pb-Ac battery are also investigated. Generally, the Li-ion battery cell consists of cathode, anode, electrolyte, conducting wire (copper or aluminium) and cell cover, as shown in Figure 17. Compositions of all major subparts of a typical Li-ion battery cell are presented in Figure 18 and Figure 19. In this study, all subparts are assumed unchanged among these four Li-ion batteries except the cathode.

The material composition database for the four Li-ion batteries in China is developed. In general, the lithium oxide contributes 15.8-18.1% of total cell weight depending on the type of Li-ion battery, while key metals in cathode (including nickel, cobalt, magnesium, iron) contribute 14.3-24.4%, and aluminium, applied in conductors and covers usually shares 17.6% (shown in Figure 20).

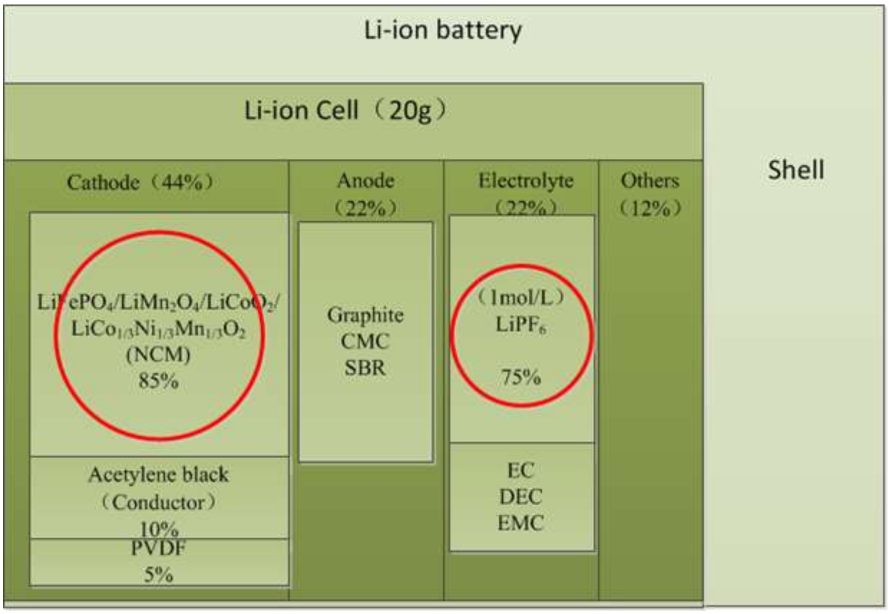

Figure 17: Major components and materials in a typical Li-ion battery

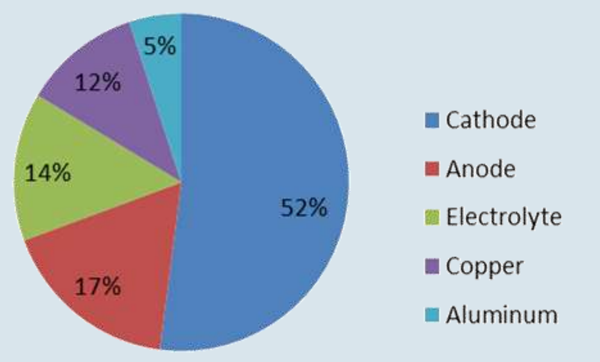

Figure 18: Component composition of a typical Li-ion battery cell

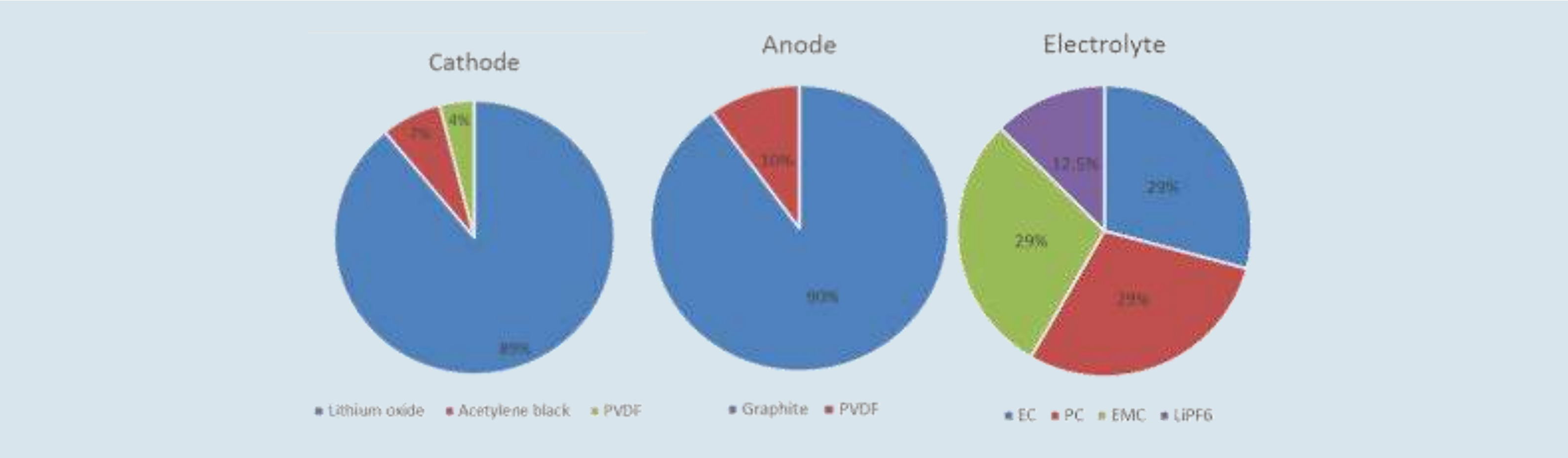

Figure 19: Material composition of major subparts in a Li-ion battery cell

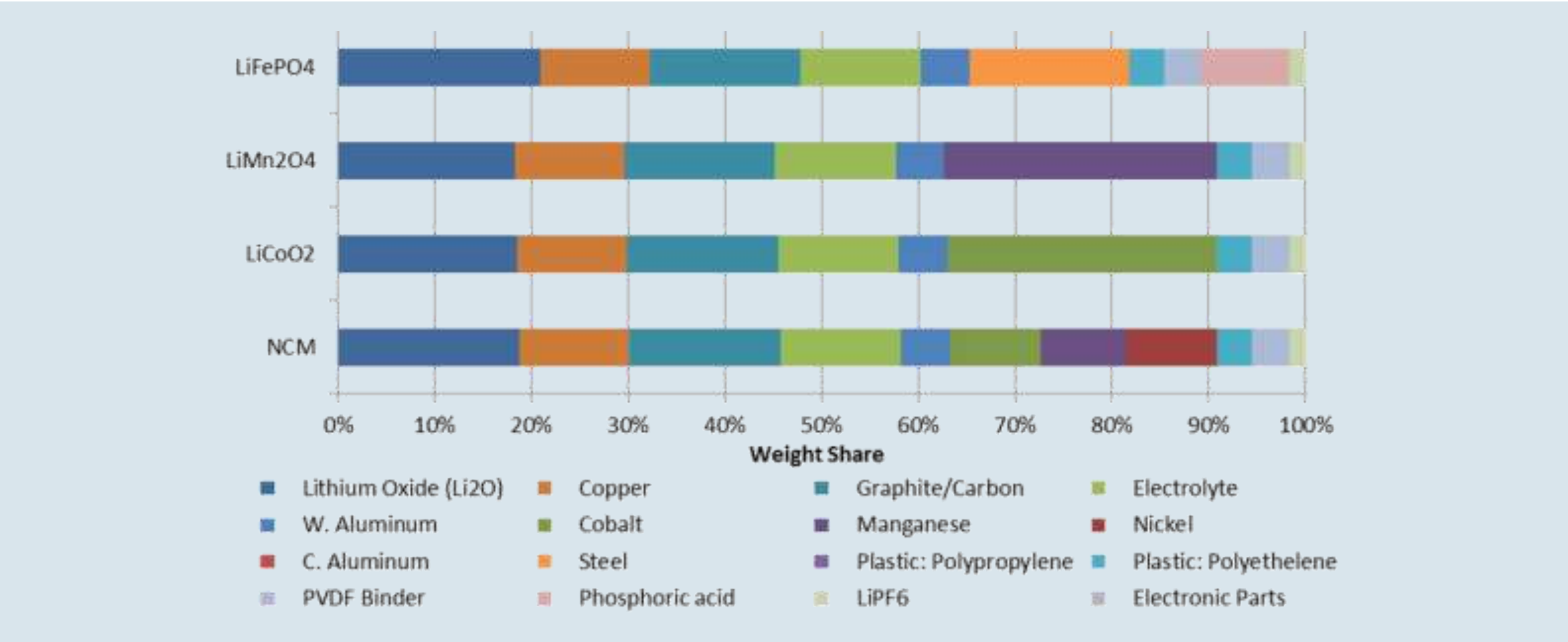

Figure 20: Material composition of four different Li-ion battery cells

The material composition of Ni-MH and Pb-ac batteries are shown in Figure 21. Data of the Ni-MH battery is from the test results of Toyota Prius in North America, in which nickel is the key material with 28% of the total weight. 69% of the Pb-Ac battery consists of lead.

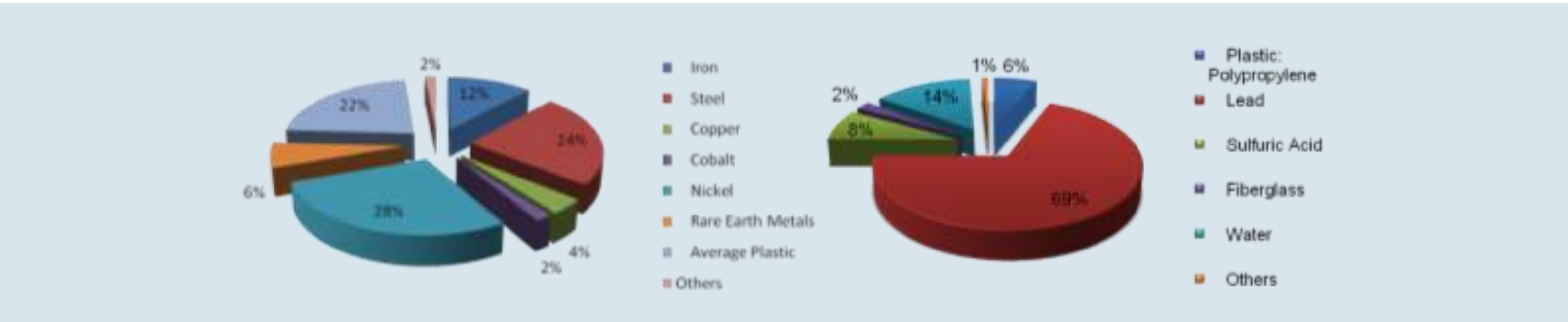

Figure 21: Material composition of Ni-MH (Left) and Pb-Ac (Right) batteries

1.3.3. The energy consumption in the critical process of material production

Based on the battery and vehicle component classification, the energy intensity data of typical key materials during the whole production processes was collected, including ore mining and dressing, transportation and storage, virgin and recycled material refinery. Figure 22, Figure 23, Figure 24 and Figure 25 shows the typical material flows and energy distribution of lithium, nickel, copper and aluminium, respectively. Within each material industry, different ores have different mining and refinery pathways. For example, lithium-containing lake brine is plentiful in China and lithium production from lake brine could save large amounts of energy compared to spodumene. Nickel, copper and aluminium have both virgin and recycled sources in China and maximizing recycled sources could save energy and ore deposits. Coal and electricity are the main fuels for material upstream production and the roasting process is often the most energy intensive compared to others.

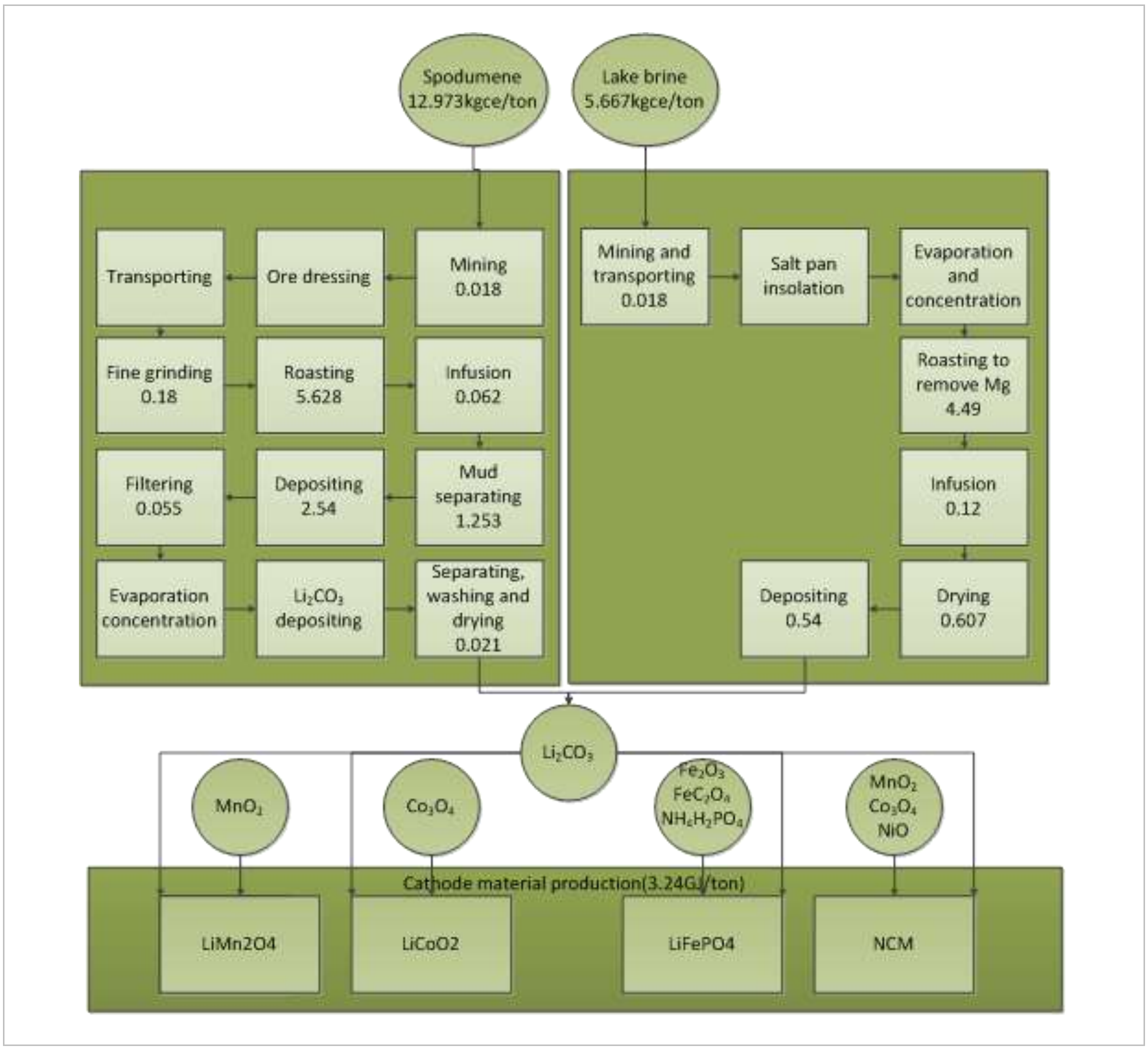

Figure 22: Energy and material flow of lithium compounds in China

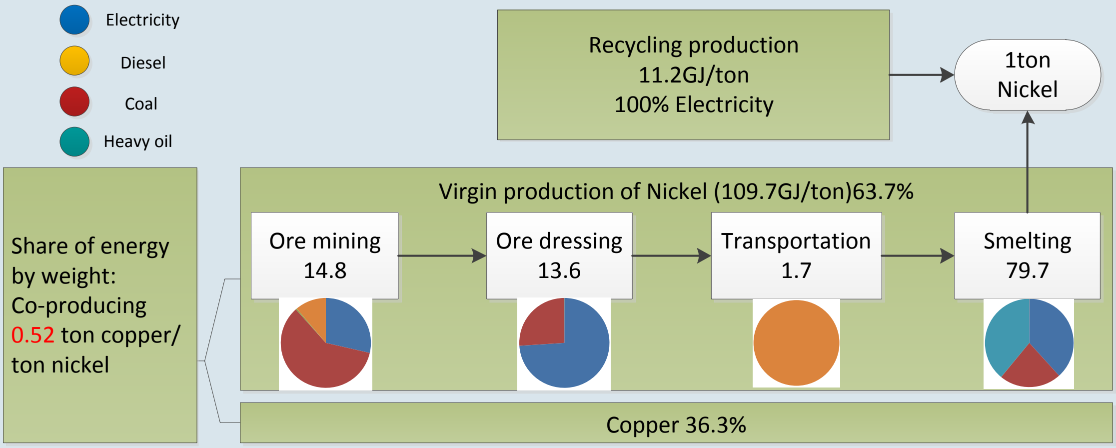

Figure 23: Energy and material flow of nickel compounds in China

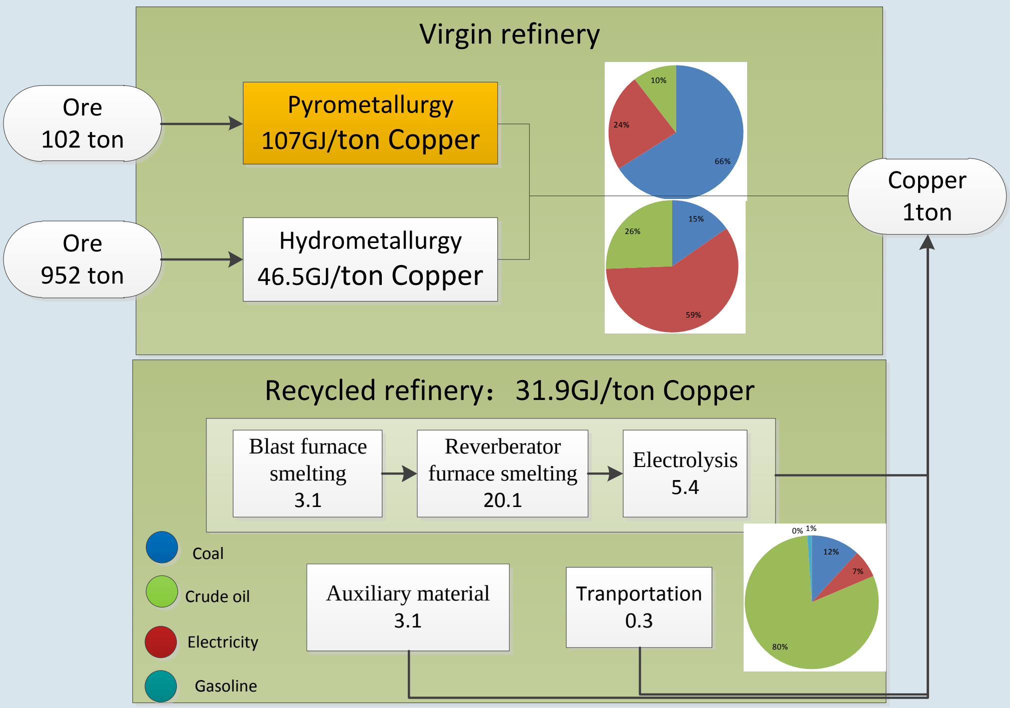

Figure 24: Energy and material flow of copper compounds in China

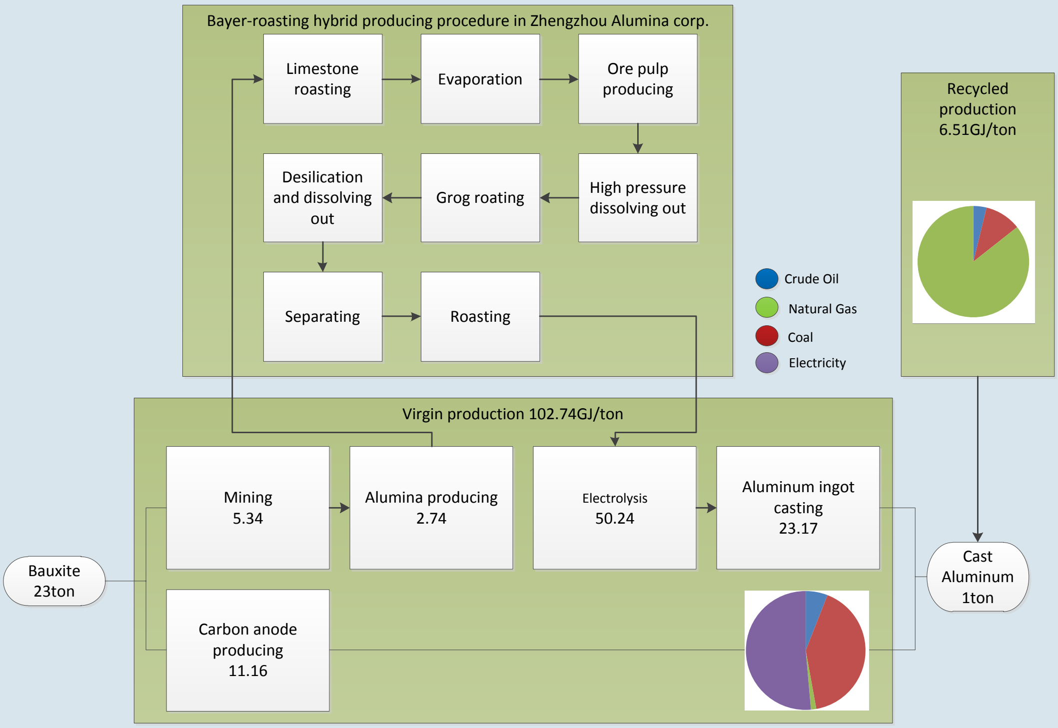

Figure 25: Energy and material flow of aluminium compounds in China

Based on the battery and vehicle components classification, the energy intensity data of typical key materials were calculated for ore mining, refining of virgin material, and recycled material refining. As illustrated in Figure 26, the energy intensity data for virgin lithium, cobalt and aluminium production are high, all exceeding 100 MJ/kg. In addition, for life cycle assessment, the types and share of process fuels for each major material process are established. In China, coal, electricity and crude oil (diesel) are dominant process fuels in the raw material production processes. For example, diesel and electricity are mainly used in ore mining while refinery processes mainly consume coal and electricity. This is significantly different from developed countries (e.g., the U.S.) where natural gas is now more popular than coal in the industrial processes.

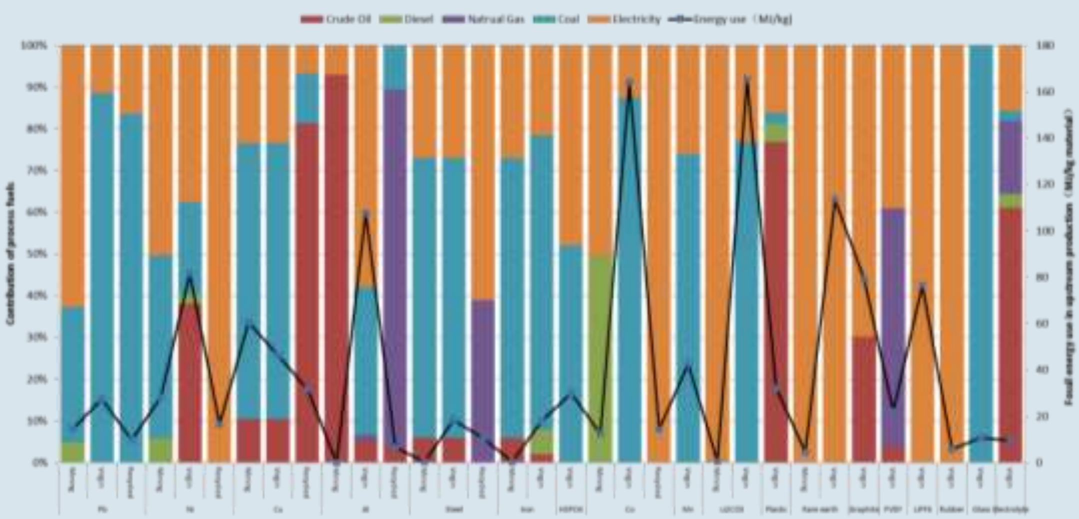

Figure 26: Share of process fuels and upstream fossil energy use for producing typical vehicle materials

1.3.4. The energy consumption in batteries module and the production of other key parts

Energy use and emissions in each process of battery module assembly were carefully collected from Chinese battery enterprises and vehicle OEMs. In this research, battery module production is further divided into two sub parts: cell production (cathode, membrane and cell synthesis) and module production (module assembly, transportation and battery assembly in a vehicle). In the first part, key metal oxides and lithium compound are combined to form cathode material, as well as membrane, anode, conductor and cover production to fabricate the battery cell. Then, in the second part, cells and cell groups are combined to form the battery module, with a steel cover. Finally, battery modules are installed on the BEVs. The contributions of energy use for all module production processes are shown in Figure 27, of which cell synthesis uses the largest amount.

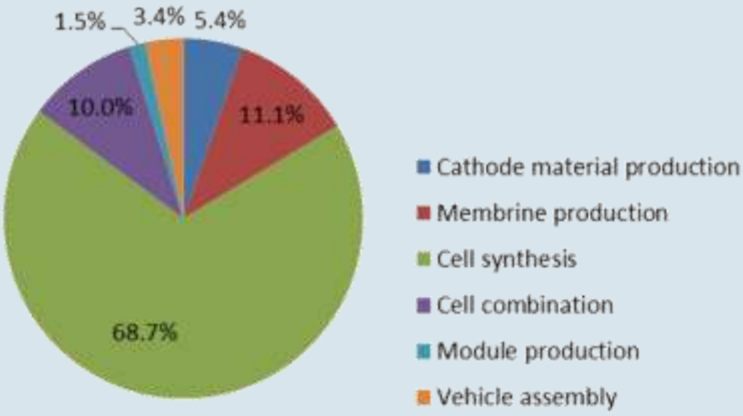

Figure 27: Contribution rate of each process in battery module production

Battery material flow data are available for other key parts of a vehicle using similar evaluation methods. The results are shown in Chapter 3.

1.3.5. Database on Key Time-series Parameters

This study covers the time-span from 2010 to 2030. Many materials and vehicle components (e.g., battery) are experiencing fast improvement in production process in China. As a result, the improvement in energy intensity over time needs to be carefully examined. This database collects key time-series parameters (e.g., energy intensity data) for major material and key processes. These data are primarily derived from the national plans and standards which classify backward, current, and future existed and advanced production technologies. Due to data constrains, the technical progress of ore mining and material refinery processes are examined every five years. For example, the share of advanced technologies would rise from about 20% in 2010 to 90% in 2030 for Li-ion battery production. These new industries could benefit more from technical revolution and faster production capacity expansion.

The upstream material production divides into three processes: ore mining, ore dressing and material refinery. Figure 28 illustrated these three processes plus battery production. We examine the technical progress every five years with relative energy saving ratio, from the baseline year 2010.

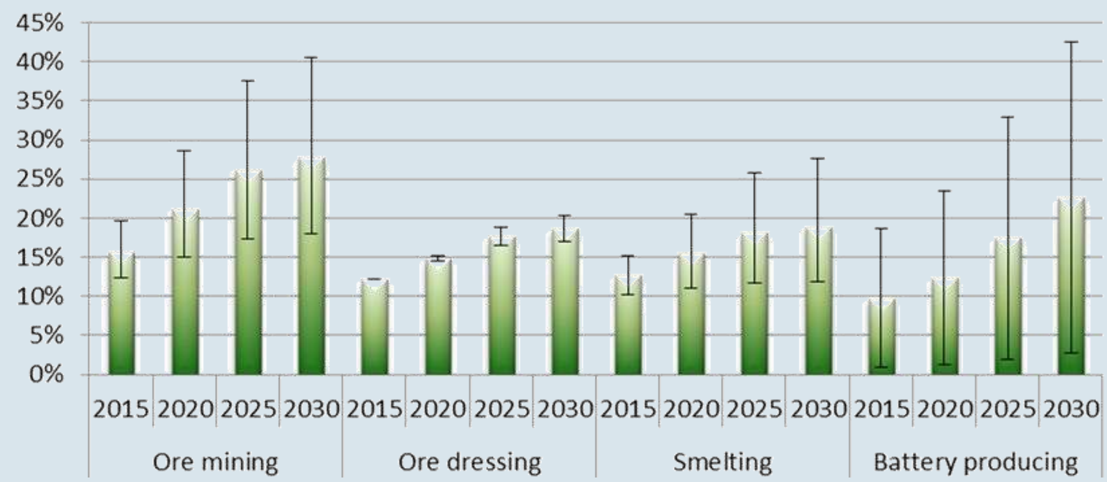

Figure 28: Technical Progress of Production Processes of Key Materials

1.4. Projection of light-duty vehicles population and fleet composition

1.4.1. Vehicle growth forecast model

Many researchers have performed forecast studies on the China's vehicle stock using different methods. Among these studies, the approach using the Gompertz curve is considered as the preferred solution to project the mid- and long-term trends in China's vehicle stock. Huo and Wang reviewed the historical Chinese vehicle stock data with three functions: the Gompertz function, the logistic function, and the Richards function. The Gompertz function fit the original data better than the other two. The Gompertz curve is an S-shaped curve, representing three periods of vehicle growth. In the beginning, the vehicle stock grows slowly when the income levels are relatively low. In the second period (also called the boom period), the vehicle stock growth is swift along with the rapid development of the economy. In the third period, the vehicle growth slows and approaches a saturation level. We applied the Gompertz function to relate per-capita LDPV ownership to per-capita GDP, as Equation 1-4 illustrates.

$$VSper_i = VSper_S \times e^{\alpha e^{\beta EconomyFactor_i}} \quad (1\text{-}4)$$

Where: $VSper_i$ is the long-term equilibrium level of on-road vehicles, vehicles per 1,000 people;

$VSper_s$ is the saturation level of vehicle, vehicles per 1,000 people;

$EconomyFactor_i$ is GDP per capita in this study, 10 thousand RMB per 1,000 people;

α and β are two parameters fitted by historical statistical data.

The historical data of population and GDP for each municipality and province are obtained from the China Statistical Yearbook 2012. Currently, JJJ and PRD are similar in both population and economic level, while YRD has a relatively higher population and economy. As of 2010, the population of JJJ, PRD and YRD reached 100.96 and 148 million, respectively; and their GDP per capita values were 29,000, 33,000 and 39,000 RMB (the year 2000 is set as the price base year). To project the future regional population and GDP growth, we relied on other relevant literature. We forecast that by 2030 the population of JJJ, PRD and YRD will reach 110, 116 and 162 million, respectively, a roughly 10-20% increase over 2010 data. However, the economic level continues to increase significantly for all of these three regions during the next two decades. The GDP per capita of these three regions will climb up to 131,000, 175,000 and 215,000 RMB (21,000-34,000 USD per capita equivalent), which are 3.5 to 4.5 times higher than the 2010 values. These values are also close to current levels in developed countries. Such an economic level is significantly higher than the national average which suggests that the growth in LDPV stock in these regions might lead to different patterns compared to the previous national-based forecasts.

The historic data of LDPV stock by each province within three regions were derived from the series of China Automotive Industry Yearbooks (2000-2012). As of 2009, the total LDPV population of JJJ, PRD and YRD reached 7.2, 4.9 and 8.1 million, respectively. The LDPV ownership numbers are 72, 51 and 55 per 1,000 people, also significantly higher than the national average (34 per 1,000 people).

Regional variation of LDPV ownership was clearly observed from this study despite a similarity in economic level between these provinces. Figure 29 presents the trends in LDPV growth from 2002 to 2010 for the seven municipalities and provinces. Beijing is the leading city in LDPV ownership with as high as 190 LDPV per 1,000 people; in contrast, Shanghai has a significantly lower LDPV ownership at only 60 per 1,000 people. It should be noted that Shanghai has the leading GDP per capita data among these cities and provinces. Therefore, other factors might also play a major role in LDPV ownership. For Shanghai's case, the ownership value is primarily attributed to the strict purchase restrictions on the number of new cars in that city since 1994 supplemented by factors such as urban structure and priority in public transportation. By reviewing many metropolitan areas in Japan and European countries (e.g., Britain) with their LDPV growth patterns, Lin discovered that when the population density was greater than 1,000 people per km^2, LDPV ownership declined with the increase in population density. Among the seven districts, only Shanghai has high population density at 3,000 people per km^2; other municipalities and provinces are characterized by a density of about 1,000 people per km^2 or lower. For provinces and municipalities other than Shanghai and Beijing, the relationship between LDPV ownership and GDP per capita is somewhat close. For example, at an income level of 30,000 RMB per capita, the LDPV ownership data for Jiangsu, Guangdong, Zhejiang and Tianjin range from 35 to 50 per 1,000 people.

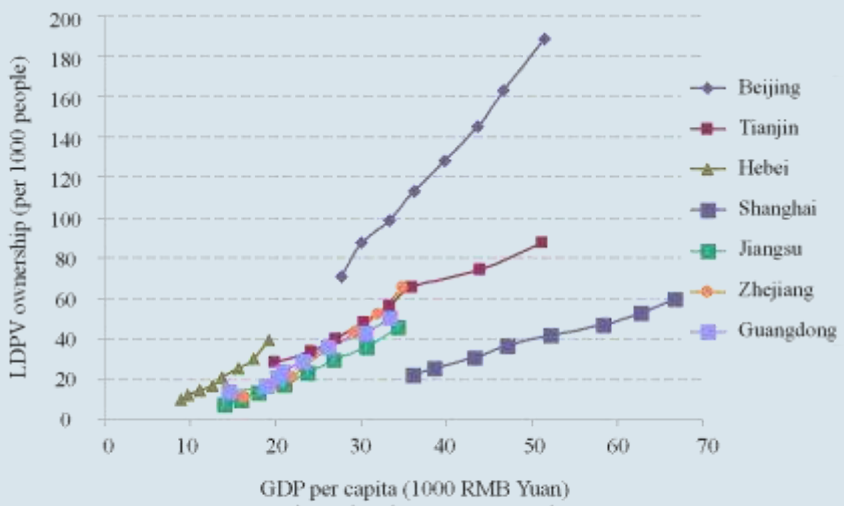

Figure 29: Regional variations in historical LDPV growth, 2002-2009

1.4.2. Projection of light-duty vehicles

1.4.2.1. Projection of light-duty vehicles in China

For each provincial region, we design two scenarios for comparison (i.e., the high and low scenarios). We estimate $VSper_s$ at 450 LDPVs per 1,000 people under the high scenario, which is comparable to that for the metropolitan areas in UK and Japan with a population density at 3,000 people per km^2. By contrast, we estimate $VSper_s$ at 350 LDPVs per 1,000 people under the low scenario by taking into account recent policies such as restrictions on new car purchase adopted in Beijing (i.e., license control policy). Figure 30 presents the estimated trends in total LDPV population in China, from 2010 to 2030. The population will increase by an annual growth rate of 12.9% under high scenario and 10.8% under low scenario on average during that period. By 2030, the total population of LDPVs in China would rise to 560 and 453 million under the high and the low scenarios, respectively.

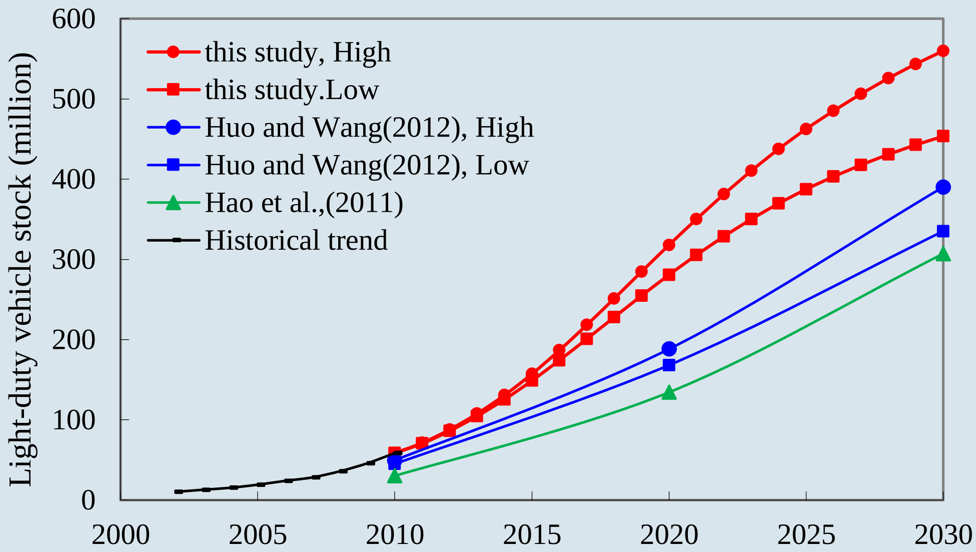

Figure 30: Projected LDPV stock in China, 2010-2030

1.4.2.2. Projection of light-duty vehicles in three developed regions

Figure 31 presents the LDPV stock projection for the three regions during the period of 2013-2030. The historical data for 2000-2012 are also illustrated in the same chart. Not surprisingly, these three regions will continue to increase in LDPV stock before 2030, especially during the first half of the next two decades (2010 and 2020). By 2030, the total LDPV population of JJJ, YRD and PRD will reach 50, 74 and 51 million, respectively, with the high scenario. Even with the low scenario, the total LDPV population of the three regions will also reach 40, 58 and 41 million, respectively.

Before 2020, the growth patterns of these three regions are close to each other although the absolute LDPV numbers could be different. This indicates that the vehicle boom period (i.e. high growth rate) in the S-shaped curve will not end sooner than 2020 for all of the regions of China. During the period of 2010-2020, the average annual LDPV growth rates of JJJ, YRD and PRD will be still as high as 12-14%, 16-18% and 17-18%, respectively, depending on the saturation level.

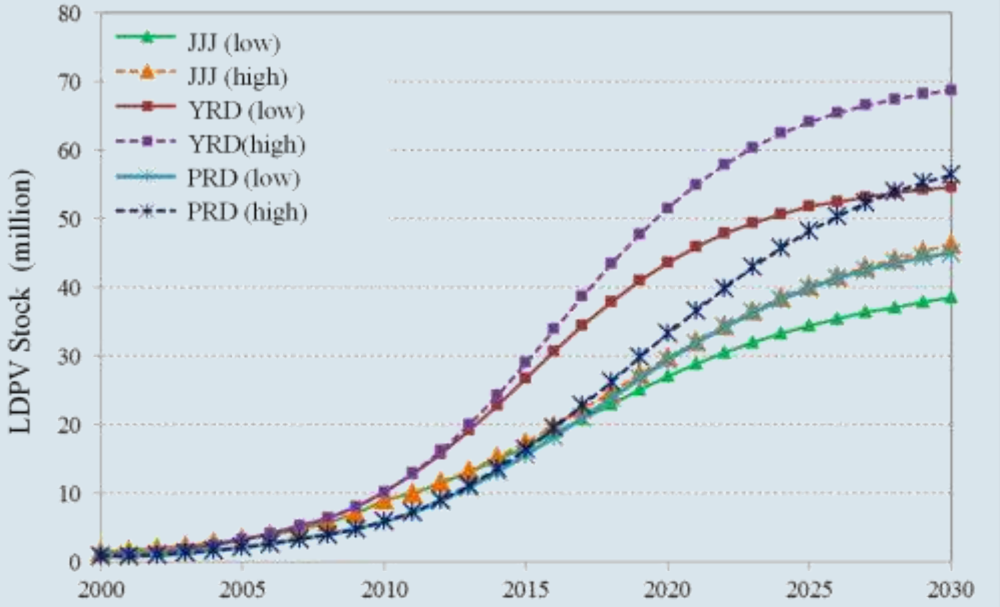

Figure 31: Projected LDPV stock of the three developed regions, 2010-2030

As shown in Figure 31, the high rate in national growth will continue after 2020, and will not end before 2030. However, after 2020, the growth trend of these three regions will slow, which is quite different from the national projection. During the period of 2020-2030, the average annual LDPV growth rates of JJJ, YRD and PRD will decrease significantly to 3-4%, 2-3% and 4-5%, respectively. By 2030, the LDPV ownership in these three regions will achieve 346 (low)/431 (high), 335/427 and 338/424 per 1,000 people, respectively. These numbers are all very close to the saturation level of LDPV ownership. Therefore, these three developed regions will be the leading regions in China to move into the vehicle saturation period (i.e., the third period of the S-shaped Gompertz curve representing low growth in vehicle stock) between 2020 and 2030.

1.4.3. The analysis of fleet composition and travelled distance

1.4.3.1. The survival curve of light-duty vehicles

Bandivadekar and Davis used Logistic regression to make mathematical statistics regression analysis for survival rate of vehicles. They found a strong correlation between Logistic curve and survival rate of a vehicle. The study is based on the Logistic model to simulate the survival rate of LDPVs in different regions. The data used for the regression simulation are mainly collected from the Yearbook and the registration and in-use vehicle inspection data from the traffic department in Beijing, Guangzhou, Macau, etc. The regression function is shown as Equation 1-5,

$$SR_i = \frac{1}{1 + b_0 \times b_1^{-i}} \qquad (1\text{-}5)$$

Where: i is the vehicle age;

SR_i is the survival rate of vehicle at the age of i;

b_0 and b_1 are the fitting parameters of the regression analysis.

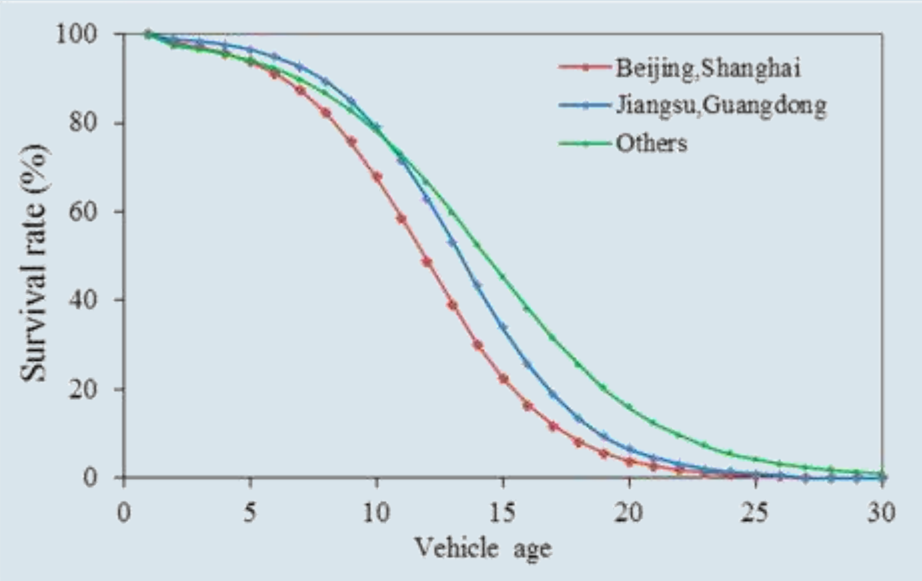

Figure 32: Survival rates for LDPVs with vehicle age in China

Given the estimated survival rates (see Figure 32) based on vehicle registration data and market shares of vehicles by year, we can calculate the LDPV population distribution by vehicle age with Equation 1-6.

$$VP_i = \sum_j \sum_{k=0}^{\sigma} \left(Sales_{i-k} \times Market_{i-k,j} \times SR_{k,j} \right) \tag{1-6}$$

Where: σ is the assumed longest life time of LDPVs, 30 years in this study;

k is the vehicle age for LDPVs;

VP_i is LDPV population in year i;

$Sales_{i-k}$ represents the sales of LDPVs in model year i–k;

$Market_{i-k,\,j}$ is market share of vehicle type j in model year i–k, %;

$SR_{k,\,j}$ is the survival rate of vehicle type j at age k, in percent.

1.4.3.2. The trends for average annual mileage of light-duty vehicles

Annual VKT data for LDPVs were estimated based on the survey data from Vehicle Emission Control Center (VECC) in 2007 and survey data by Tsinghua University in several cities (e.g., Beijing, Guangzhou). In the past decade, the annual VKT in most cities shows a decreasing trend because of rising proportion of the private cars. Moreover, traffic control measures in Beijing and several other cities will further reduce the vehicle use intensity.

Therefore, we also design two scenarios of trends in fleet-average VKT of LDPVs, as shown in Figure 33. For example, the annual VKT per vehicle in 2010 for Beijing is 17,000 km while in 2030 the annual VKT will be about 10,000 km under high scenario and 8,000 km under low scenario.

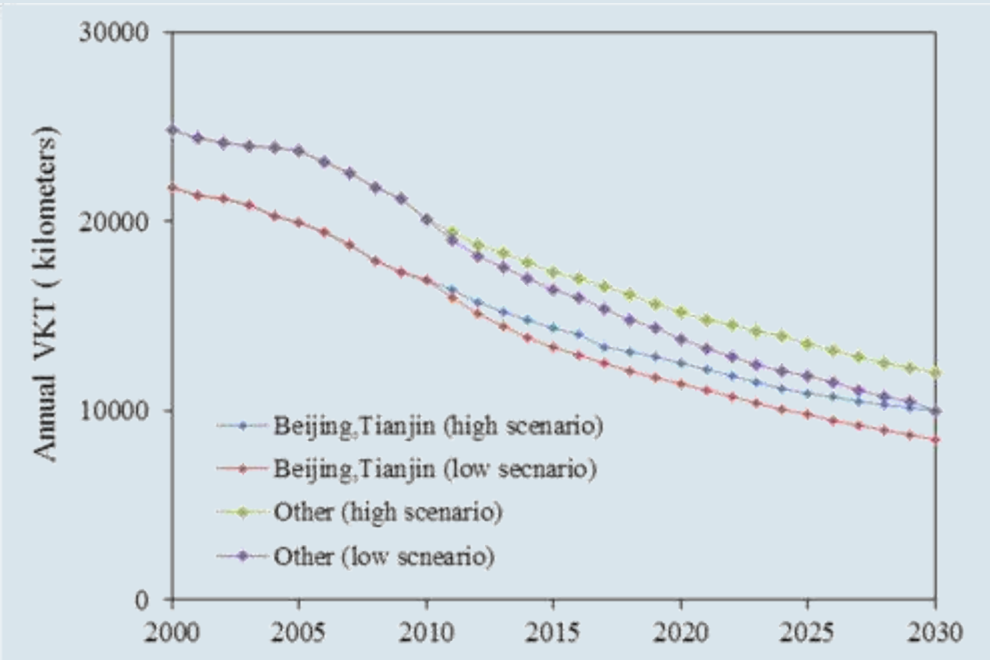

Figure 33: The trends for average annual VKT of light-duty vehicles

2. The Well to Wheels Energy Consumption and Pollutant Emissions of LDPV

2.1. Well-to-wheel result in China

This chapter focuses on the WTW energy use and pollutant emissions of LDPV in China and six regions: Jing-Jin-Ji, Yangtze River Delta, Pearl River Delta, Northeast China, Central China and Northwest China. The generation mix of these regions is significantly different from the national level. Further, the light duty vehicle emission standards of three developed regions (JJJ, YRD and PRD) are usually 2-3 years ahead of the other three regions and national standard. We set two scenarios for WTW results depending on the two scenarios of generation mix, fuel economy and emission factors.

The comparison of energy consumption and emission factors is based on vehicle stock. Taking CY (Calendar Year) 2015 as an example, because, on fleet average, the midpoint of lifetime use for Chinese light-duty vehicles is about five years, we adopted the fuel economy and emission factor values of vehicles' MY (Model Year) 2010, which is five years ahead of the target year for simulation. In other words, we assume the average fuel economy and emission factors of LDPV's fleet in 2015 equal to those of vehicles released in 2010, with a five-year deterioration of fuel consumption and pollutant emissions in 2015.

2.1.1. Energy use and CO_2 emissions

2.1.1.1. Petroleum consumption

Figure 34 presents the per-kilometre WTW petroleum use for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart in China from 2015 to 2030. Petroleum consumption is concentrated in the vehicle driving stage. Because the proportion of oil-based electricity in the generation mix is slight, and the petroleum consumption is relatively low in the upstream stage with high efficiency in the oil refining industry. For example, the petroleum consumption in TTW amounts to 90% of the total for ICEV. With the improvement of fuel economy, the WTW petroleum consumption for ICEV will decline 28% by 2030, compared to the petroleum consumption in 2015.

HEV, PHEV15 and PHEV50 can reduce WTW petroleum energy use by ~30%, ~40%, and ~50%, respectively, relative to ICEV. BEV almost eliminates petroleum use, because BEV totally relies on electricity during TTW stage with a small proportion of oil power. For PHEV, the reduction of petroleum consumption is related to the battery capacity. Elgowainy et al.'s study pointed out that the petroleum consumption of PHEV will decrease with the increase of AER (increase of battery capacity), which is similar to the results in this study. Considering that the imported crude oil for China will still increase in the future, promoting the PHEV and BEVs have significant value for the energy security of China.

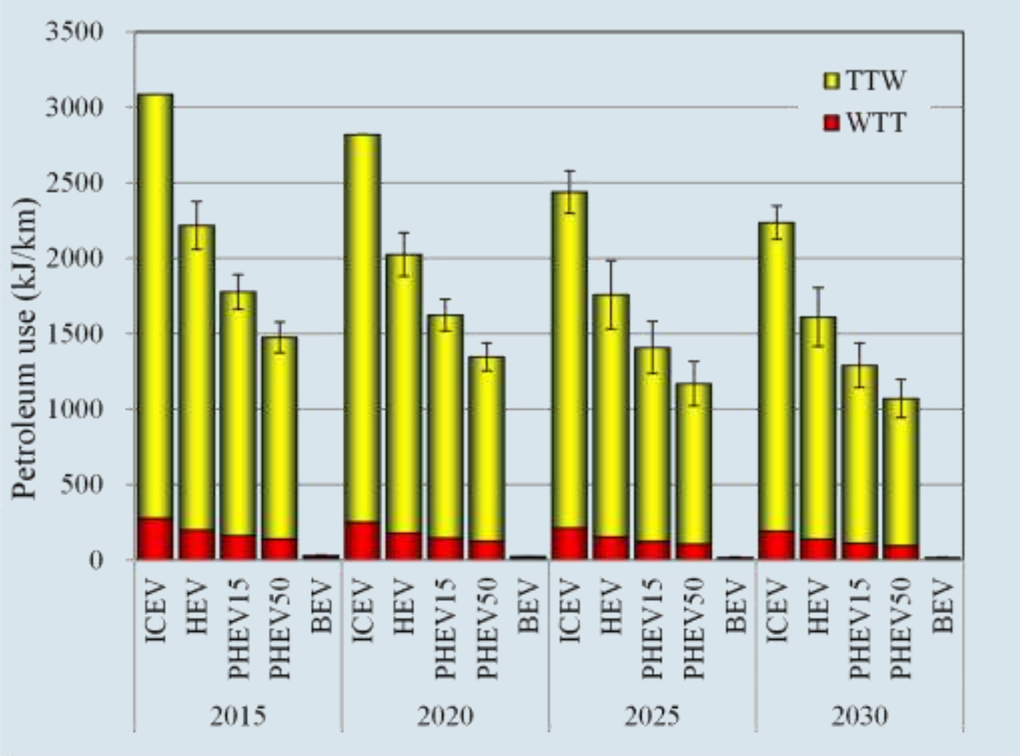

Figure 34: WTW petroleum consumption of LDPV in China, 2015-2030

2.1.1.2. Fossil energy consumption

Figure 35 presents the per-kilometre WTW fossil fuel consumption for HEV, PHEV15, PHEV50, and BEV relative to their ICEV counterpart in China from 2015 to 2030. Traditional fossil fuels include petroleum, natural gas, and coal. The WTW fossil fuel consumption for ICEV, HEV, PHEV and BEVs will decrease in the future with the improvement of fuel economy, generating efficiency and clean energy ratio.

EVs can reduce WTW fossil fuel consumption effectively, compared to ICEV. In 2015, WTW fossil fuel consumption of HEV, PHEV15, PHEV 50 and BEV decreases 28%, 33%, 22% and 37%, respectively, compared to ICEV. By 2030, WTW fossil fuel consumption of HEV, PHEV15, PHEV 50 and BEV will decrease 28%, 34%, 27% and 50%, respectively, compared to the ICEV. The increase of clean energy proportion in the up-stream, leads to the obvious reduction of fossil fuel consumption for PHEV and BEV.   The ICEV comparison: I assume that is to ICEVs of 2015 and 2030 respectively.

The fossil fuel consumption advantage is not as good as petroleum consumption advantage for PHEV and BEV, because they consume more fossil fuel than ICEV during the WTT stage. The PHEV50 and BEV increase fossil fuel consumption by 52% and 119% in the WTT stage, respectively, relative to ICEV. By 2030, the PHEV50 and BEV increase 36% and 73% fossil fuel consumption in the WTT stage, respectively, relative to ICEV, with the increase of the nuclear power and wind power. So the generation mix and coal-fired efficiency are the key factors to effect the fossil fuel consumption for PHEV and BEV. Therefore, we will discuss the WTW fossil fuel consumption for EVs in different regions in the following 2.2.1.

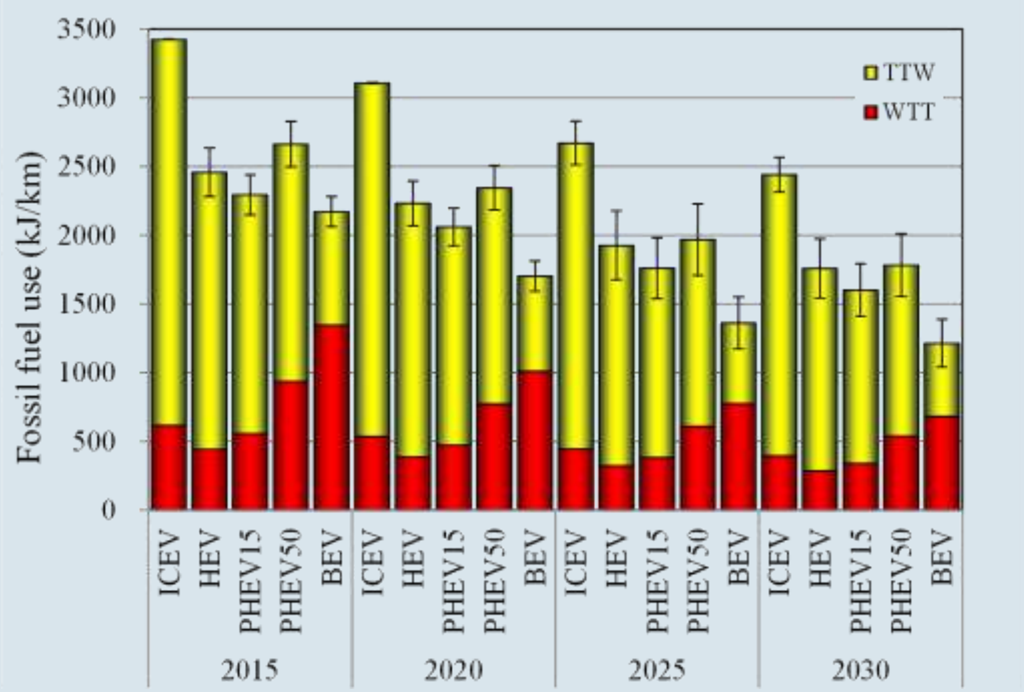

Figure 35: WTW fossil fuel consumption of LDPV in China, 2015-2030

2.1.1.3. CO_2

Figure 36 presents the per-kilometre WTW CO_2 emissions for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart in China from 2015 to 2030. CO_2 emissions are directly related to the consumption of fossil fuels, which will continuously decrease over the next two decades due to the improvement of the upstream generation efficiency and the downstream vehicle fuel economy, and the increase of clean energy generation share.

In 2015, the CO_2 emissions of HEV, PHEV15, PHEV50 and BEV reduce by 28%, 31%, 16% and 23%, respectively, relative to ICEV. The advantage of CO_2 emission reduction is the same as the fossil fuel reduction for HEV. But CO_2 emission reduction for PHEV and BEV are not as good as fossil fuel reduction, due to the high carbon content of coal and the large amount of CO_2 that would be emitted during the upstream coal-fired generation stage. With the improvement of generating efficiency and clean energy share, the CO_2 emissions of PHEV15, PHEV50 and BEV reduce by 33%, 22% and 41%, respectively, relative to ICEV. So promoting the PHEV and BEV should be combined with the clean power generation technology and carbon capture and storage (CCS) to reduce the WTW CO_2 emissions.

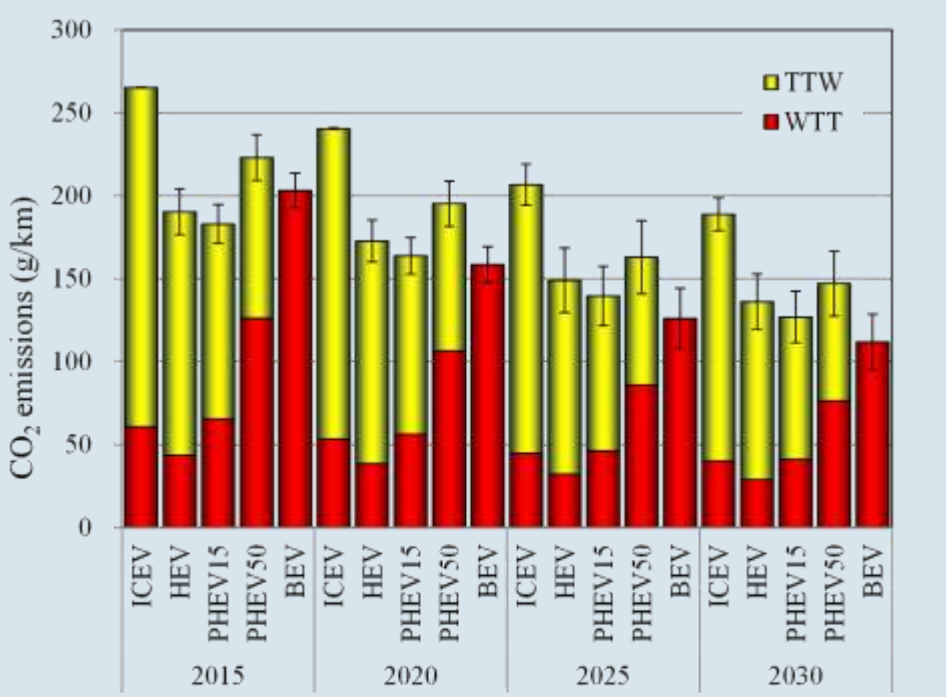

Figure 36: WTW CO2 emissions of LDPV in China, 2015-2030

2.1.2. Air pollutant emissions

2.1.2.1. VOC

Figure 37 presents the per-kilometre WTW VOC emissions for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart in China from 2015 to 2030. WTW VOC emissions are mainly attributed to the vehicle driving stage, which will decrease with the improvement of vehicle emission standards. This study assumed the vehicle emission level in 2015 is equal to the 5-year degraded vehicle emission level under China III emission standard, and that no more stringent standards for LDGV would be implemented after the China VI standard. Therefore, TTW VOC emissions will remain unchanged, while the WTW VOC emission reduction will slow with time.

EVs can reduce WTW VOC emissions effectively, compared to ICEV. VOC emissions in the WTT stage take about 20% of the total for HEV and PHEV, so the reduction in the TTW stage is quite beneficial for the WTW VOC emissions. While for BEV, the VOC emissions are all come from the upstream electricity generation. So in 2015, WTW VOC emissions of HEV, PHEV15, PHEV 50 and BEV decreases by 39%, 53%, 65% and 96%, respectively, compared to ICEV. For CO_2 emissions, the reduction benefit for EVs is reduced in the future. By 2030, WTW VOC emissions of HEV, PHEV15, PHEV50 and BEV will decrease 30%, 35%, 56% and 93%, respectively, compared to ICEV. This is due to the reduction rate of VOC emissions from the vehicle operation of ICEV occurs more quickly than the reduction rate of VOC emissions from the power plants for BEV. BEV has huge advantage for the VOC emissions, compared to other EVs.

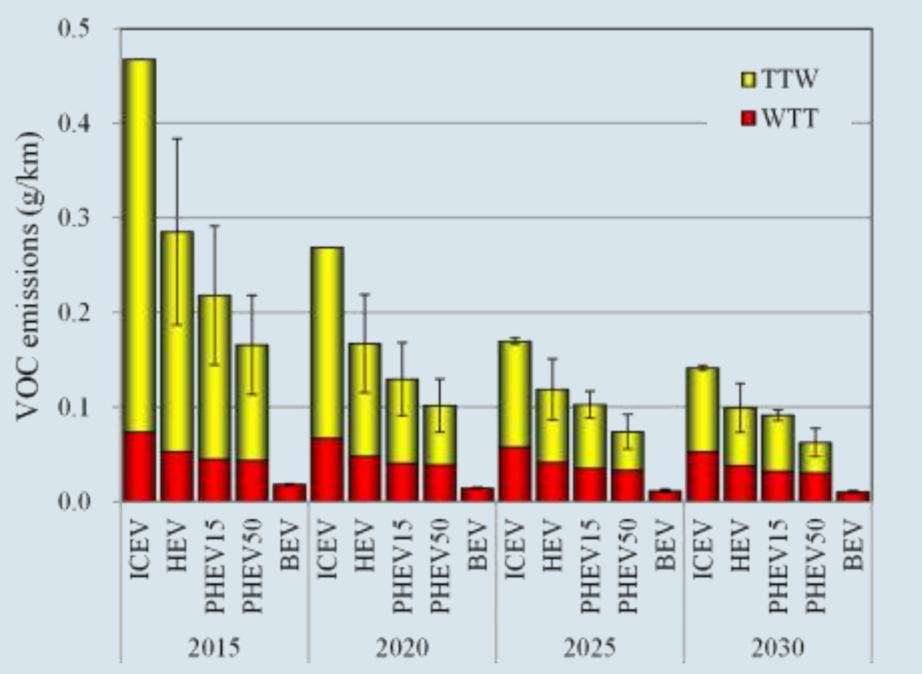

Figure 37: WTW VOC emissions of LDPV in China, 2015-2030

2.1.2.2. CO

Figure 38 presents the per-kilometre WTW CO emissions for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart in China from 2015 to 2030. Similar to VOC emissions, CO emissions mainly come from the vehicle driving stage. With the improvement of vehicle emission standards, CO emissions will decline in the future.

EVs can reduce WTW CO emissions effectively, compared to ICEV. CO emissions in the WTT stage are about 5% of the total for HEV and PHEV, so the reduction in the TTW stage is particularly beneficial for the WTW CO emissions. For BEV, the CO emissions are all come from the upstream electricity generation. So in 2015, WTW CO emissions of HEV, PHEV15, PHEV 50 and BEV decreases by 51%, 63%, 73% and 99%, respectively, compared to ICEV. Similar to the VOC emissions, the reduction benefit for EVs is reduced in the future. By 2030, WTW CO emissions of HEV, PHEV15, PHEV50 and BEV will decrease 36%, 52%, 65% and 98%, respectively, compared to ICEV. This is because the reduction rate of CO emissions from the vehicle operation of ICEV is more rapid than the reduction rate of CO emissions from the power plants for BEV. BEV has a huge advantage for the CO emissions compared to other EVs.

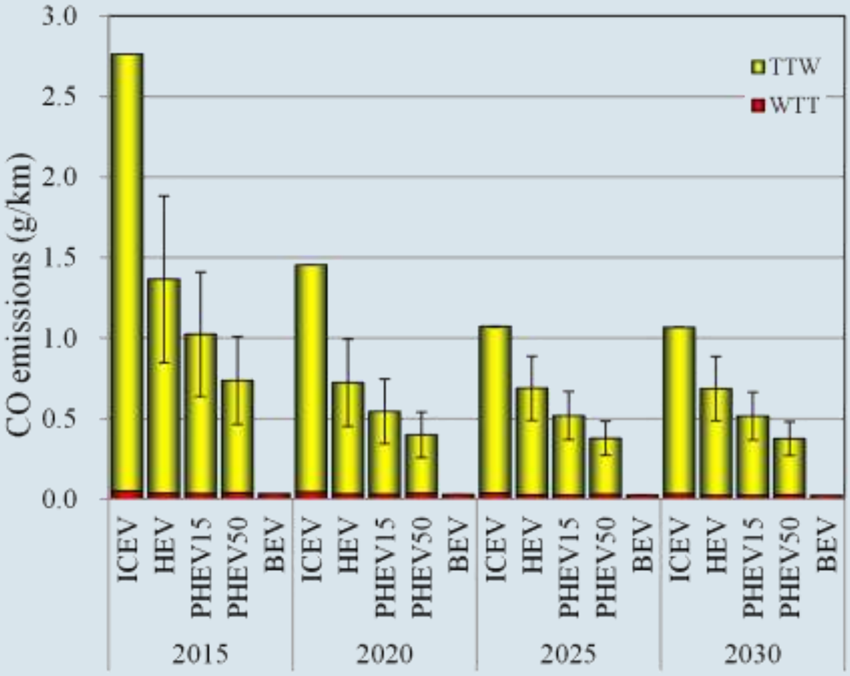

Figure 38: WTW CO emissions of LDPV in China, 2015-2030

2.1.2.3. NO_X

Figure 39 presents the per-kilometre WTW NO_X emissions for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart in China from 2015 to 2030. Unlike the VOC and CO emissions, WTW NO_X emissions mainly originate from the upstream producing stage. With the improvement of power generating efficiency and clean energy generation share, NO_X emissions decrease gradually.

WTT NO_X emissions of HEV and PHEV account for 70%~90% of total WTW emissions. For BEV, all WTW NO_X emissions come from the WTT stage, which is very high due to the domination of coal-based electricity generation in the energy generation mix. In 2015, HEV and PHEV15 could reduce WTW NO_X emissions by about 30% compared to the ICEV. PHEV50 almost has the same NO_X emissions as the ICEV. For BEV, WTW NO_X emissions increase by 30% over ICEV. With more stringent emission standards of power plant and increase of clean power generation technology, WTW NO_X emissions of BEV will decline in the future. By 2030, WTW NO_X emissions of HEV, PHEV15, PHEV50 and BEV decreases by 25%, 26%, 5% and 5%, respectively, compared to ICEV. So promoting the PHEV and BEV should be combined with the clean power generation technology and high generation efficiency.

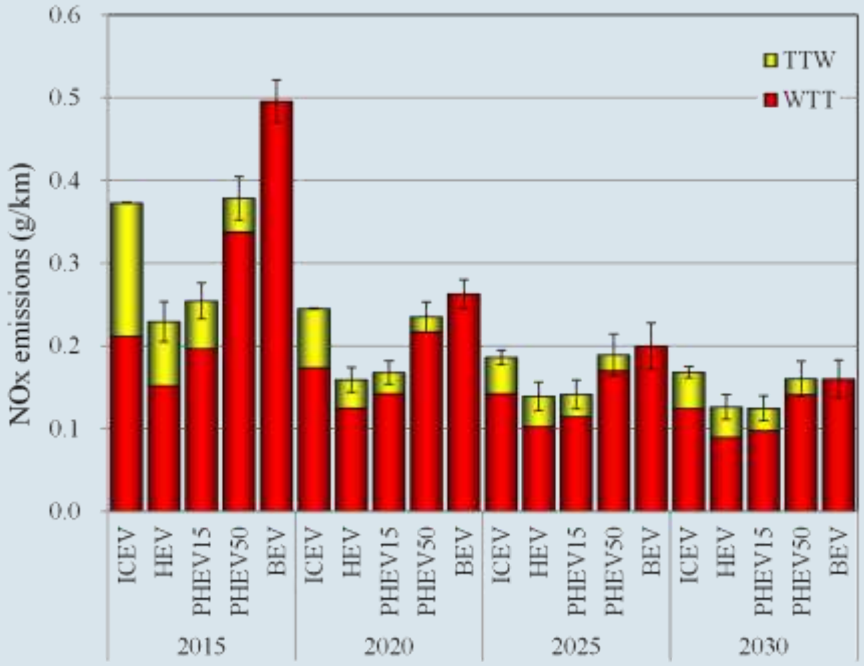

Figure 39: WTW NOX emissions of LDPV in China, 2015-2030

2.1.2.4. $PM_{2.5}$

Figure 40 presents the per-kilometre WTW $PM_{2.5}$ emissions for HEV, PHEV15, PHEV50, and BEV relative to their ICEV counterpart in China from 2015 to 2030. Similar to NO_X emissions, WTW $PM_{2.5}$ emissions mainly come from the upstream producing stage. As the power generating efficiency and clean energy generation share increase, $PM_{2.5}$ emissions decreases gradually.

WTT $PM_{2.5}$ emissions of HEV and PHEV account for 80%~95% of total WTW emissions. For BEV, all WTW $PM_{2.5}$ emissions come from the WTT stage, which is very high due to the domination of coal-based electricity generation in the generation mix. In 2015, HEV and PHEV15 could reduce WTW $PM_{2.5}$ emissions by 34% and 12%, respectively, compared to ICEV. PHEV50 and BEV increase WTW $PM_{2.5}$ emissions by 56% and 130%, respectively, compared to ICEV. With more stringent emission standards of power plant and increase of clean power generation technology, WTW $PM_{2.5}$ emissions of PHEV50 and BEV will decline in the future. By 2030, WTW $PM_{2.5}$ emissions of HEV and PHEV15 will decrease by 30% and 14%, respectively, compared to ICEV. PHEV 50 and BEV will still increase by 42% and 91%, respectively, compared to ICEV. So promoting the PHEV and BEV should be combined with the clean power generation technology and $PM_{2.5}$ emissions removal technology.

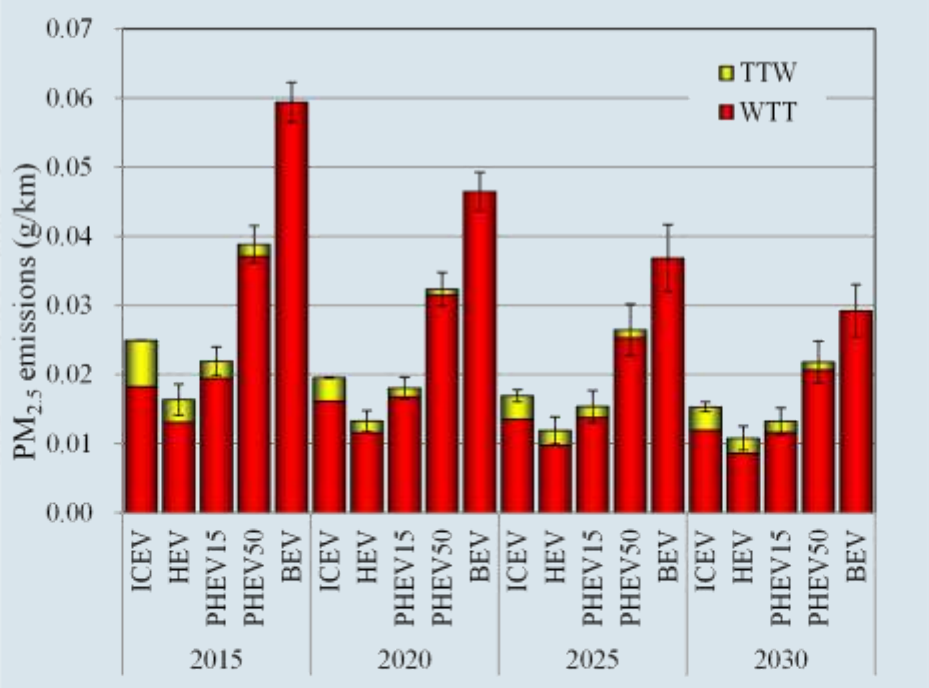

Figure 40: WTW PM2.5 emissions of LDPV in China, 2015-2030

2.1.2.5. SO_2

Figure 41 presents the per-kilometre WTW SO_2 emissions for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart in China from 2015 to 2030. Similar to NOX and $PM_{2.5}$ emissions, WTW SO_2 emissions mainly come from the upstream producing stage. As the power generating efficiency and the clean energy generation share increase, SO2 emissions decrease gradually.

WTT SO_2 emissions of HEV and PHEV account for 90%~95% of total WTW emissions. For BEV, all WTW SO_2 emissions come from the WTT stage, which is very high due to the domination of coal-based electricity generation in the generation mix. In 2015, HEV could reduce WTW SO_2 emissions by 28% compared to ICEV. WTW SO_2 emissions of PHEV15 are comparative with ICEV. For PHEV50 and BEV, WTW SO_2 emissions increase by 90% and 200%. With more stringent emission standards for power plant emissions and an increase of clean power generation technology, WTW SO_2 emissions of PHEV50 and BEV will decline in the future. By 2030, WTW SO_2 emissions of HEV and PHEV15 will decrease by 28% and 13%, respectively, compared to ICEV. PHEV50 and BEV will still increase by 41% and 82%, respectively, compared to ICEV. As mentioned for other pollutants, clean power generation technology and SO_2 emissions control technology is very necessary when considering PHEV and BEV emissions from the perspective of fuel-cycle.

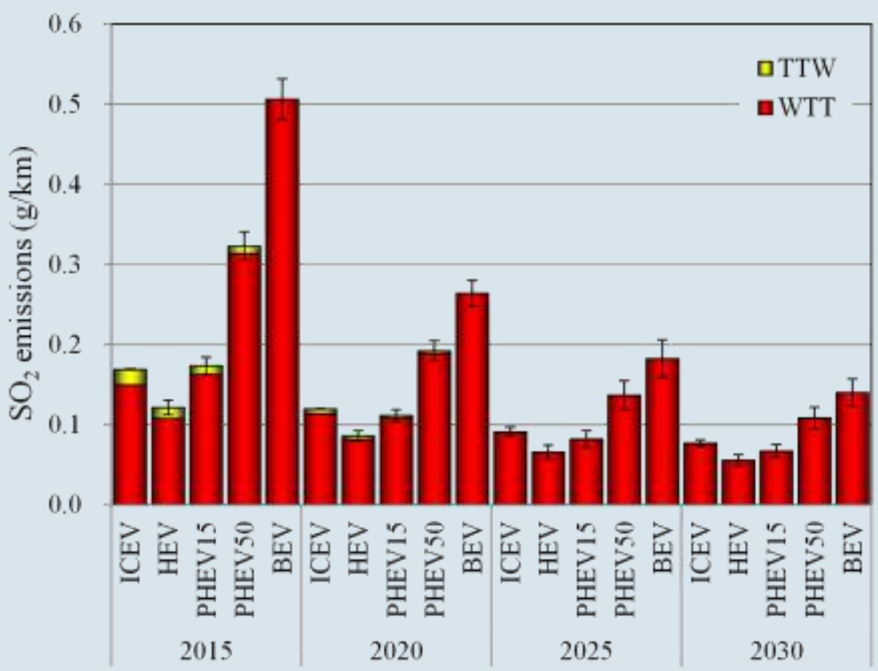

Figure 41: WTW SO2 emissions of LDPV in China, 2015-2030

2.1.3. Discrepancy between the two scenarios

To address the uncertainty of the projection for future generation mix and the improvement rates of new energy vehicles compared to conventional counterparts on national average level, we built up two different scenarios for the simulation, namely, the conservative scenario and ambitious scenario. In this sector, we chose the WTW emission results of CO and NO_X to quantitatively measure the discrepancy between the two scenarios.

As shown in Figure 38, HEV could reduce 32% and 69% WTW CO emissions compared to ICEV in 2015, respectively in the conservative and ambitious scenario; PHEV15 could reduce 49% and 77% and PHEV50 could reduce 63% and 83% by the same year. While BEV has almost the same WTW CO emissions in the two scenarios. With the cover rate of electric driving increases, the discrepancy of WTW CO emissions between the two scenarios narrows. Therefore, the discrepancy of WTW CO emissions of these new energy vehicle technologies between the two scenarios is primarily due to the differences of fuel economy among these vehicle technologies, because CO emissions mostly come from the TTW stage, namely the vehicle driving stage.

However, the story of WTW NO_X emissions is quite different from that of CO. As shown in Figure 39, HEV could reduce 32% and 45% WTW CO emissions compared to ICEV in 2015, respectively in the conservative and ambitious scenario; PHEV15 could reduce 26% and 38% in the two scenarios. However, PHEV50 has 9% higher WTW NO_X emissions than ICEV in the conservative scenario and 6% lower in the ambitious scenario by the same year. BEV even has 40% and 26% higher than ICEV in the conservative and ambitious scenario, respectively. For all of these vehicle types, the discrepancies of WTW NO_X emissions between the two scenarios are comparable, which is quite different from that of WTW NO_X emissions. That is because NO_X emissions mainly come from the WTT stage. Therefore, the discrepancy of WTW NO_X emissions of these new energy vehicle technologies between the two scenarios is primarily caused by the difference of generation mixes, which is identical to all of these vehicle technologies.

2.2. Well-to-wheel result in six regions

2.2.1. Energy use and CO_2 emissions

2.2.1.1. Petroleum consumption

Reduction of petroleum use in the transportation sector is a very useful and effective method to ensure the energy security of China, as the vehicle population and imported crude oil of China still increases. Figure 42 presents the per-kilometre WTW petroleum use for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart by each of the six regions from 2015 to 2030. The main difference for parameter setting for the same model year among the six regions is the generation mix. The parameters impacting the petroleum consumption in the driving stage like fuel economy are all considered equal to the national level, causing the regional results of petroleum consumption to be quite close to each other.

Petroleum consumption is concentrated in the vehicle driving stage because the proportion of oil-based electricity in generation mix is slight in all the six regions, and the petroleum consumption is relatively low in the upstream stage with high processing efficiency in oil refining industry. Furthermore, WTW petroleum consumption will decrease over time. Not surprisingly, HEV, PHEV15 and PHEV50 can reduce WTW petroleum energy use by ~30%, ~40% and ~50%, respectively, compared to ICEV; and BEV almost eliminates petroleum use.

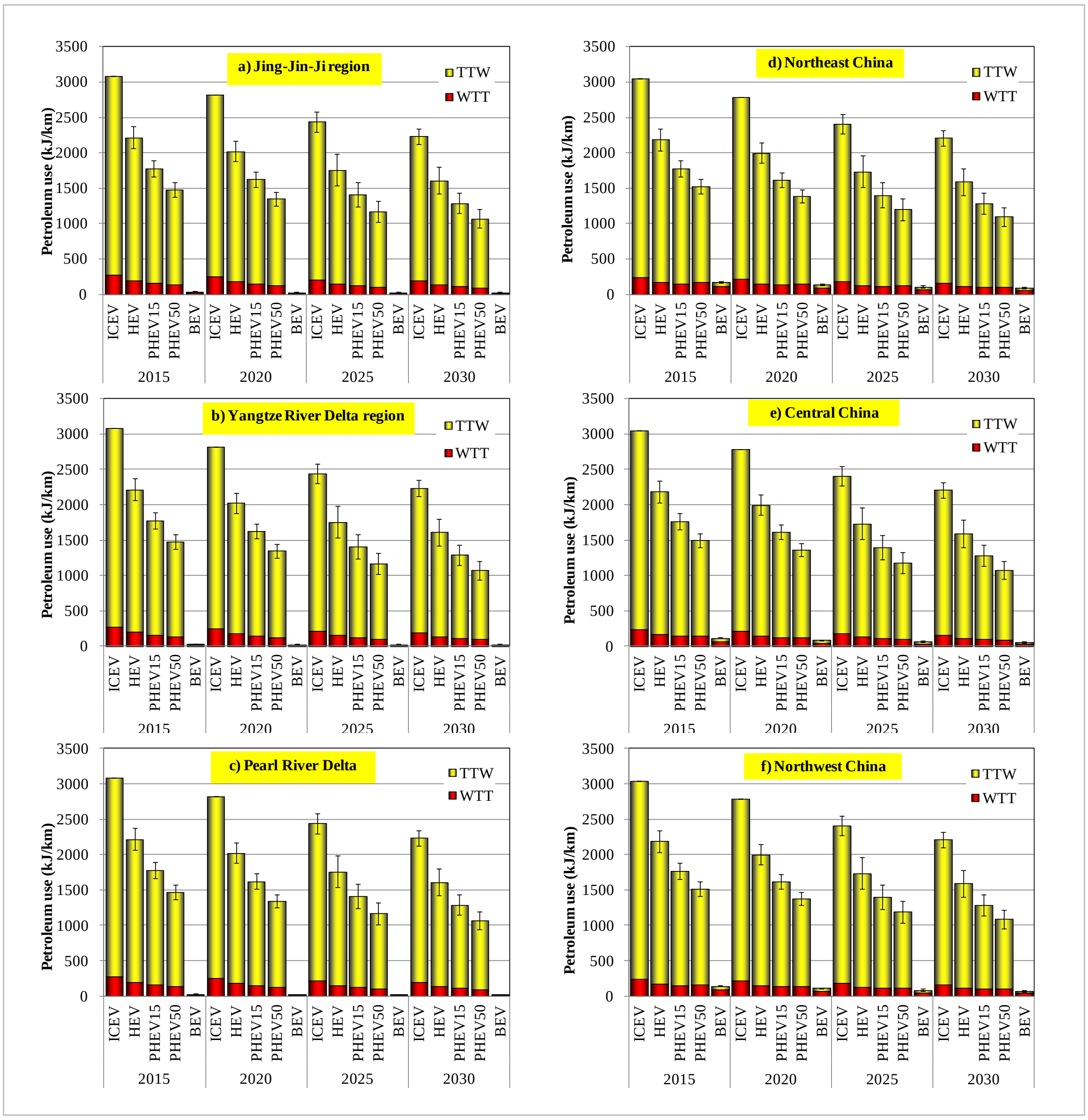

Figure 42: WTW petroleum consumption of LDPV in six regions of China, 2015-2030

2.2.1.2. Fossil energy consumption

Traditional fossil fuels include petroleum, natural gas and coal. Figure 43 presents the per-kilometre WTW fossil fuel consumption for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart by each of the six regions from 2015 to 2030. Similar to the petroleum consumption results, HEV could realize remarkable reduction by 28% in WTW fossil fuel consumption compared to ICEV.

In 2015, WTW fossil fuel consumption of PHEV15, PHEV 50 and BEV decrease by 31%, 16% and 24% compared to ICEV in JJJ, respectively, which is significantly lower than the reduction rate of petroleum consumption. This is mainly attributed to an overwhelming share of coal-based electricity in this region (more than 95%). A reduction in TTW stage is partly offset by a significant increase in fossil fuel consumption in WTT stage. The promotion of PHEV and BEV in PRD and Central China region could achieve a higher reduction in fossil fuel use due to lower coal-based electricity share in these regions. Taking the PRD as an example, it has only 63% of total electricity generation from coal, while a considerable 28% from hydropower and 5% from nuclear energy. Consequently, driving PHEV15, PHEV50, or BEV in this region could reduce WTW fossil energy use by 35%, 28%, and 48%, respectively, compared to ICEV.

WTW fossil energy consumption will decrease continuously. If we only consider the generating efficiency improvement of coal-fired power plants, and assume a remaining high share of coal-based electricity generation by 2030 (the conservative scenario in the generation mix projection), BEV could reduce WTW fossil energy consumption by 35%, 41%, 56%, 36%, 58%, and 47% compared to ICEV in JJJ, YRD, PRD, Northeast, Central, and Northwest, respectively. If clean energy generating were further pursued in these three regions to meet the targets in the generation mix projection of the ambitious scenario, BEV could achieve more reduction in the WTW fossil energy consumption. In the ambitious scenario by 2030, the reduction rate could be 44%, 50%, 64%, 47%, 68% and 58%, respectively, for BEV compared to ICEV in the six regions with corresponding projected coal-based electricity generation share of 77%, 64%, 44%, 70%, 42% and 55%, respectively.

It should be noted that PHEV and BEV do not show considerable advantages on WTW fossil energy consumption compared to HEV. For those regions with a high coal-based generation share, such as JJJ, BEV has comparable WTW fossil energy consumption in most cases with HEV in the next two decades. Only in those regions with a relatively high clean energy generation share, such as PRD and Central China, could the BEV achieve better reduction effects in WTW fossil energy consumption compared to HEV.

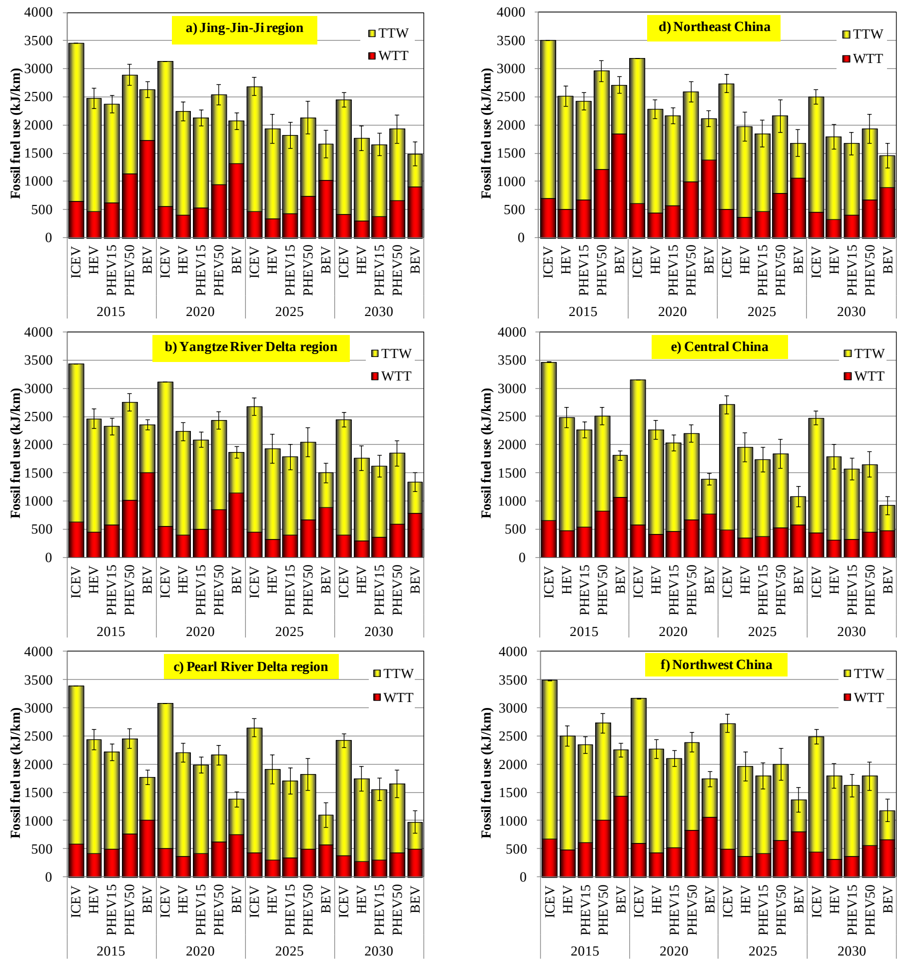

Figure 43: WTW fossil fuel consumption of LDPV in six regions of China, 2015-2030

2.2.1.3. CO_2

Figure 44 presents the per-kilometre WTW CO_2 emissions for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart by each of the six regions from 2015 to 2030. CO_2 emissions are directly related to the consumption of fossil fuels, which will continuously decrease over the next two decades due to the improvement of the upstream generation efficiency and the downstream vehicle fuel economy, and the increase of clean energy generation share.

Similar to the petroleum and fossil energy consumption results, HEV could achieve a remarkable reduction in WTW CO_2 emissions by 28% compared to ICEV. However, if we look to PHEV and BEV, the results are different. Due to high carbon content of coal, large amounts of CO_2 would be emitted during the upstream coal-fired generation stage. In 2015, the WTW CO_2 emissions of PHEV15, PHEV50 and BEV in JJJ are lower by 29%, 9% and 8%, respectively, than ICEV, which are much lower reduction rates than that in those regions with relatively high clean energy generation share, such as PRD (33%, 23% and 37%, respectively).

WTW CO_2 emissions will decrease continuously over time. If we only consider the generating efficiency improvement of coal-fired power plants, and assume a remaining high share of coal-based electricity generation by 2030 (conservative scenario in the generation mix projection), BEV could reduce WTW CO_2 emissions by 21%, 30%, 48%, 20%, 48% and 34% compared to ICEV in JJJ, YRD, PRD, Northeast, Central and Northwest China, respectively. If clean energy generation in these three regions is further promoted to meet the targets in the generation mix projection of the ambitious scenario, PHEV and BEV could achieve more potential in the WTW CO2 emissions reduction. In the ambitious scenario by 2030, the reduction rate could be 34%, 42%, 58%, 35%, 60% and 49%, respectively, for BEV compared to ICEV in the six regions. In addition, as both the generating efficiency of upstream coal-fired power plants and the clean energy generation share increase, the WTW CO_2 emissions reduction rate of PHEV and BEV will be higher than that of ICEV.

In summary, HEV is better than PHEV and BEV to mitigate WTW CO_2 emissions on transportation sector in those regions with high coal-based electricity generation share, especially in JJJ. Experience indicates that it is difficult to make a significant change in the generation mix in different regions of China in the mid-term, which raises a substantial challenge to significantly lower the coal-based generation share in these regions. Therefore, only in those regions with a considerable proportion of clean energy generation, such as PRD and Central China, will an improved WTW CO_2 emissions reduction be achieved through the promotion of PHEV and BEV on a large scale in the next two decades.

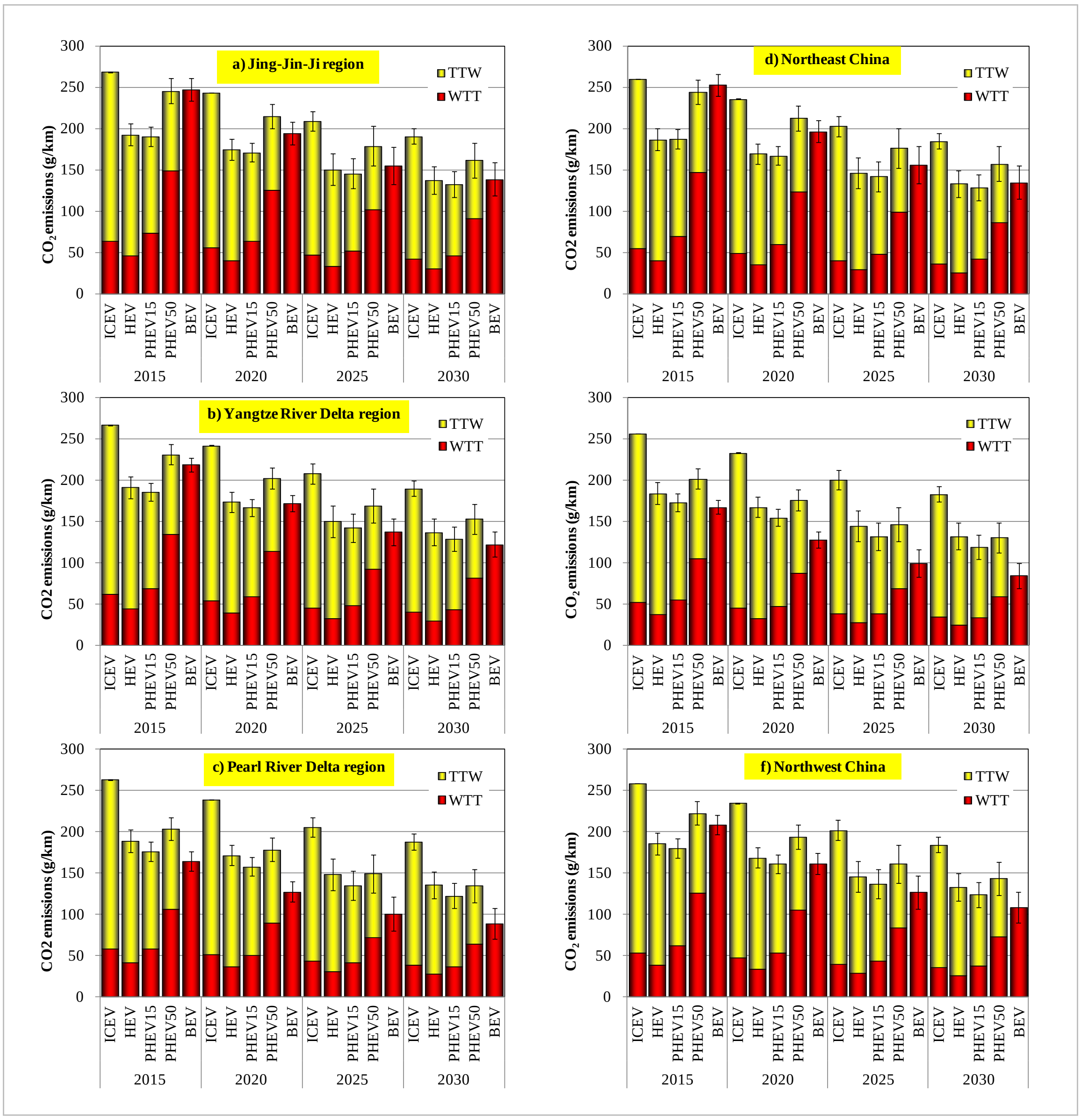

Figure 44: WTW CO2 emissions of LDPV in six regions of China, 2015-2030

2.2.2. Air pollutants emissions

2.2.2.1. Volatile organic compound (VOC)

Figure 45 presents the per-kilometre WTW VOC emissions for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart by each of the six regions in five year periods from 2015 to 2030. WTW VOC emissions are mainly attributed to the vehicle driving stage, which will decrease with the improvement of vehicle emission standards. This study assumed the vehicle emission level in 2015 is equal to the 5-year degraded vehicle emission level under China IV emission standard in JJJ, YRD and PRD, while for other three regions the vehicles adopted China III standards, and no more stringent standards for LDGV would be implemented after the China VI standards. Therefore, TTW VOC emissions will remain unchanged, while the WTW VOC emission reduction will slow with time.

There are no significant differences among the WTT VOC emissions of the six regions. But for WTW VOC emissions, the discrepancy is remarkable because VOC emissions from the vehicle driving stage are quite different among the six regions.

For ICEV, TTW VOC emissions in Northeast, Central, and Northwest China are significantly higher than that of the other three regions, especially in 2015. This is because the vehicle emission standards in Northeast, Central, and Northwest China are lagging, the other three developed regions. However, the vehicle emission standards of different regions will tend to be consistent with time, which makes the WTW VOC emissions in all the six regions almost identical in 2030. HEV could reduce 30%~40% WTW VOC emissions compared to ICEV. In addition, PHEV15 and PHEV50 could achieve about 50% and 60% reduction of WTW VOC emissions in 2015, respectively, compared to ICEV. But the reduction rate slightly decreases after 2020, due to the decrease of VOC emission factors of ICEV caused by the continuously improving vehicle emission standards. Consequently, BEV has the highest 90%~95% reduction rate for VOC emissions as a zero-emission vehicle in the driving stage.

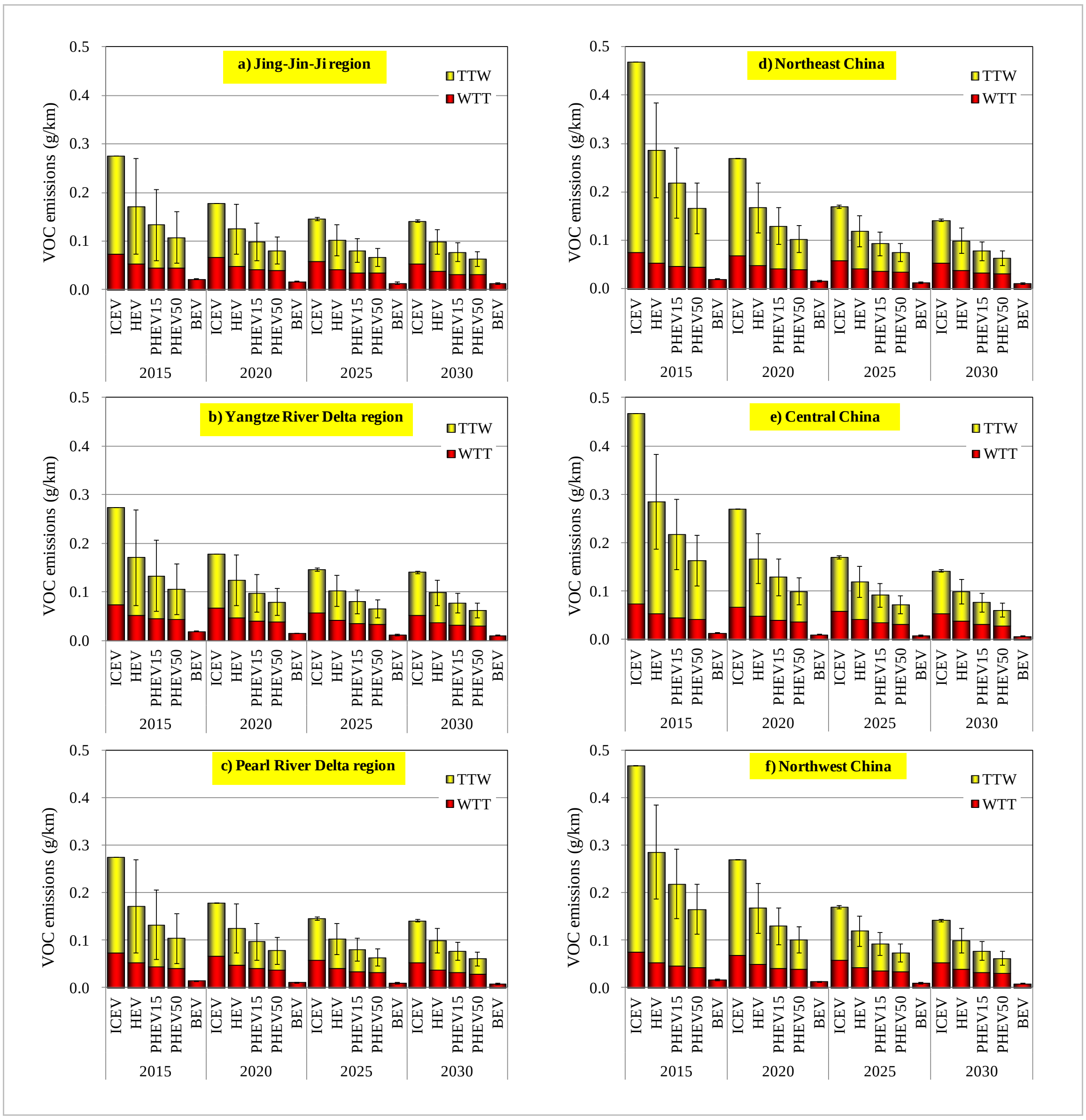

Figure 45: WTW VOC emissions of LDPV in six regions of China, 2015-2030

2.2.2.2. Carbon Oxide

Figure 46 presents the per-kilometre WTW CO emissions for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart for each of the six regions from 2015 to 2030.

WTW CO emissions are mainly attributed to the emissions in vehicle driving stage that will decrease with the improvement of vehicle emission standards. Quite similar to VOC, there are no significant differences among the WTT CO emissions of the six regions. But for WTW CO emissions, the discrepancy is significant because CO emissions from the vehicle driving stage are quite different among the six regions. But the proportion of WTT CO emissions in the entire WTW emissions is much smaller than that of VOC. For the three regions of Northeast, Central, and Northwest China, WTW CO emissions of the ICEV will decrease sharply from 2015 to 2025, then slowly to a consistent level of the other three regions.

HEV could reduce CO emissions by 50% and 36% WTW in 2015 and 2020, respectively, compared to ICEV. In addition, PHEV15 and PHEV50 could achieve 60% and 70% reduction of WTW CO emissions in 2015, respectively. While the reduction rate will decrease to 50% and 60% in 2030, respectively, due to the decrease of CO emission factors of ICEV caused by the continuously tightening vehicle emission standards. Consequently, BEV has the highest 98% reduction rate for CO emissions as a zero-emission vehicle in the driving stage. In conclusion, BEV has more the greatest advantages for WTW CO emission reduction.

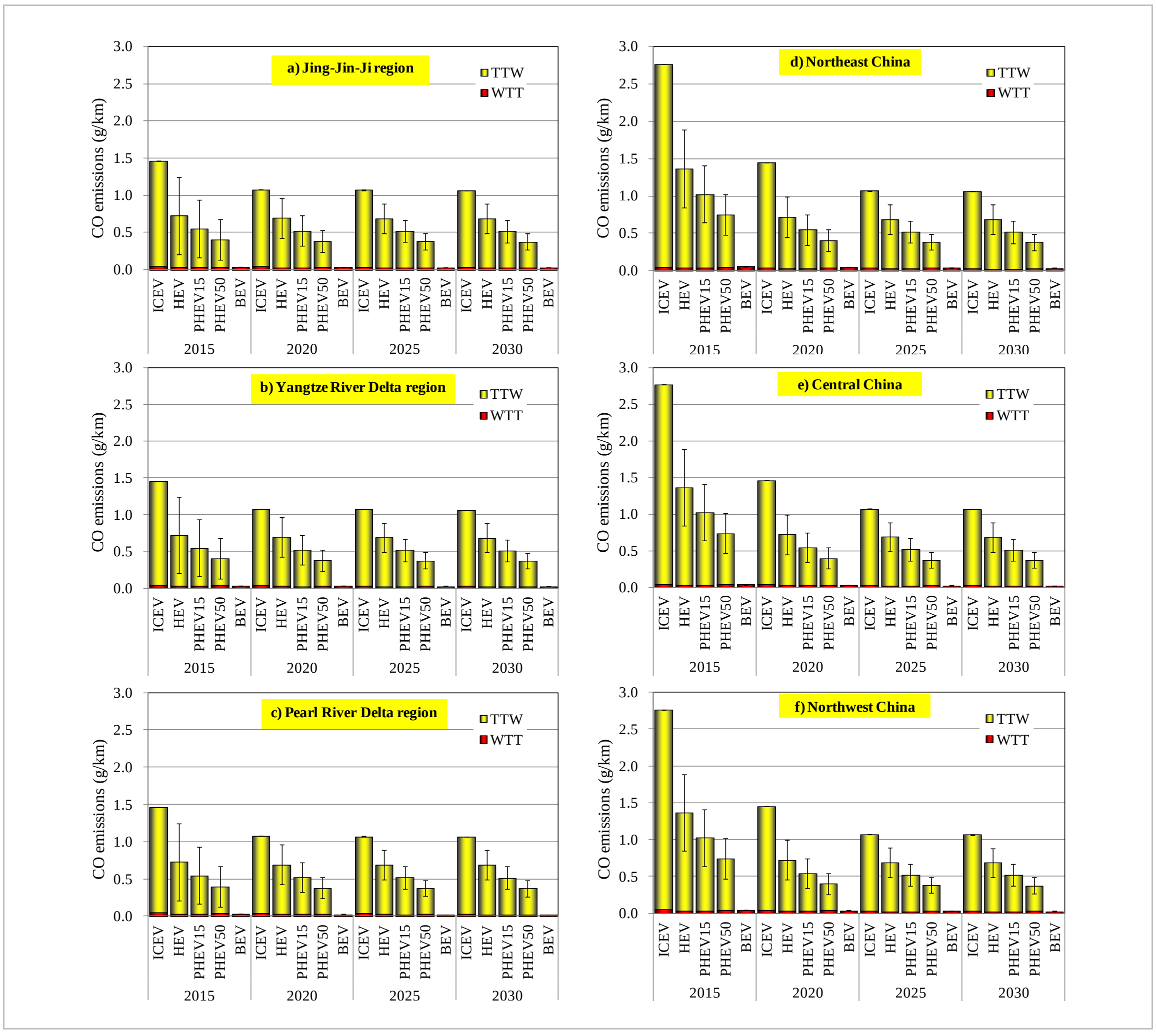

Figure 46: WTW CO emissions of LDPV in six regions of China, 2015-2030

2.2.2.3. NO_X

Figure 47 presents the per-kilometre WTW NO_X emissions for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart by each of the six regions from 2015 to 2030. WTW NO_X emissions mainly come from the upstream production stage, which is quite different from the VOC and CO emissions described before. As the power generating efficiency and clean energy generation share increase, NO_X emissions decrease gradually. Because of the different generation mixes, the six regions have significant differences for WTW NO_X emissions: those regions with high coal-based generation share like JJJ and Northeast China have higher NO_X emissions than the regions with relatively high clean energy generation share like PRD and Central China.

WTT NO_X emissions of HEV and PHEV account for 70% to 90% of total WTW emissions. For BEV, all WTW NO_X emissions come from the WTT stage, which is very high due to the domination of coal-based electricity generation in the generation mix. In 2015, HEV and PHEV15 in the six regions could reduce WTW NO_X emissions by about 30% and 20%, respectively, compared to ICEV. But PHEV50 has higher NO_X emissions than ICEV in all regions except the Central China, such as 42% higher in JJJ and 24% higher in Northeast China. For BEV, NO_X emissions are even worse: 108% higher in JJJ, 80% higher in Northeast China. Under the projected generation mix in 2015, driving a BEV in all the six regions has no advantages for NO_X emission reduction. Regional variations will gradually expand over time. In 2030, HEV in the six regions could reduce WTW NO_X emissions by about 25%, while PHEV15 could reduce 20%-30%, compared to ICEV. WTW NO_X emissions of PHEV50 and BEV in Northeast China are higher by 13% and 30%, respectively, compared to ICEV, but in PRD NO_X emissions in 2030 are lower by 16% and 26%, respectively. This shows that PHEV50 and BEV in those regions with high clean energy generation share could achieve the goal of WTW NO_X emission reduction.

In conclusion, PHEV and BEV may be primarily promoted in those regions with high clean energy generation share to achieve a meaningful and effective NO_X emission reduction.

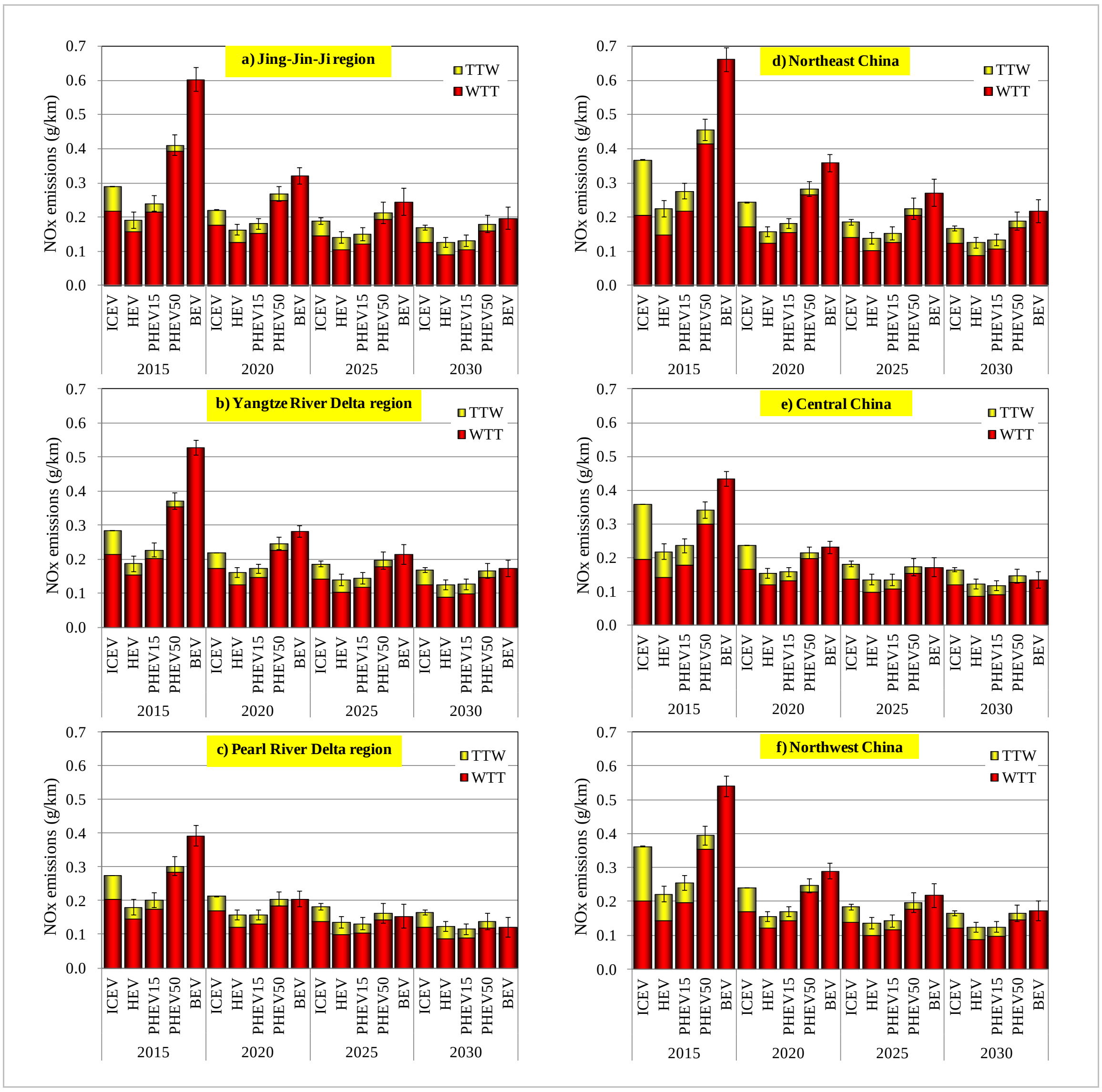

Figure 47: WTW NOX emissions of LDPV in six regions of China, 2015-2030

2.2.2.4. $PM_{2.5}$

Figure 48 presents the per-kilometre WTW $PM_{2.5}$ emissions for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart by each of the six regions from 2015 to 2030. Similar to NO_X emissions, WTW $PM_{2.5}$ emissions mainly come from the upstream producing stage. As the power generating efficiency and clean energy generation share increase, $PM_{2.5}$ emissions decrease gradually. Because of different generation mix, the six regions have significant differences for WTW $PM_{2.5}$ emissions: those regions with high coal-based generation share like JJJ and Northeast China have higher $PM_{2.5}$ emissions than the regions with a relatively high clean energy generation share like PRD and Central China.

WTT $PM_{2.5}$ emissions of HEV and PHEV account for 75% to 95% of total WTW emissions. For BEV, all WTW $PM_{2.5}$ emissions come from the WTT stage, which is very high due to the domination of coal-based electricity generation in the generation mix. In 2015, HEV in all six regions could reduce WTW $PM_{2.5}$ emissions by about 35% compared to ICEV. WTW $PM_{2.5}$ emissions of PHEV15 are similar to ICEV in JJJ and Northwest China, while lower by 12% and 9% than ICEV in

PRD and Central China. For PHEV50 and BEV, WTW $PM_{2.5}$ emissions in all six regions are much higher than ICEV.

Under the projected generation mix in 2015, driving a BEV in all the six regions has no advantages for $PM_{2.5}$ emission reduction. Regional variations will gradually expand over time. In 2030, HEV in all the six regions could reduce WTW $PM_{2.5}$ emissions by about 30%, compared to ICEV. For PHEV15, only in Northeast China with a high coal-based electricity share and relatively high emission factors of combustion boilers are WTW PM2.5 emissions higher than for ICEV. WTW $PM_{2.5}$ emissions of PHEV50 and BEV in Northeast China are higher by 108% and 221%, respectively, compared to ICEV; in PRD these emissions are higher by 19% and 45%, respectively. This shows that even in those regions with high clean energy generation share, PHEV50 and BEV could not reduce WTW $PM_{2.5}$ emissions.

In conclusion, to significantly reduce WTW $PM_{2.5}$ emissions, we need more clean energy generation and higher penetration rate of $PM_{2.5}$ control technology in power plants when selectively promoting PHEV and BEV in China.

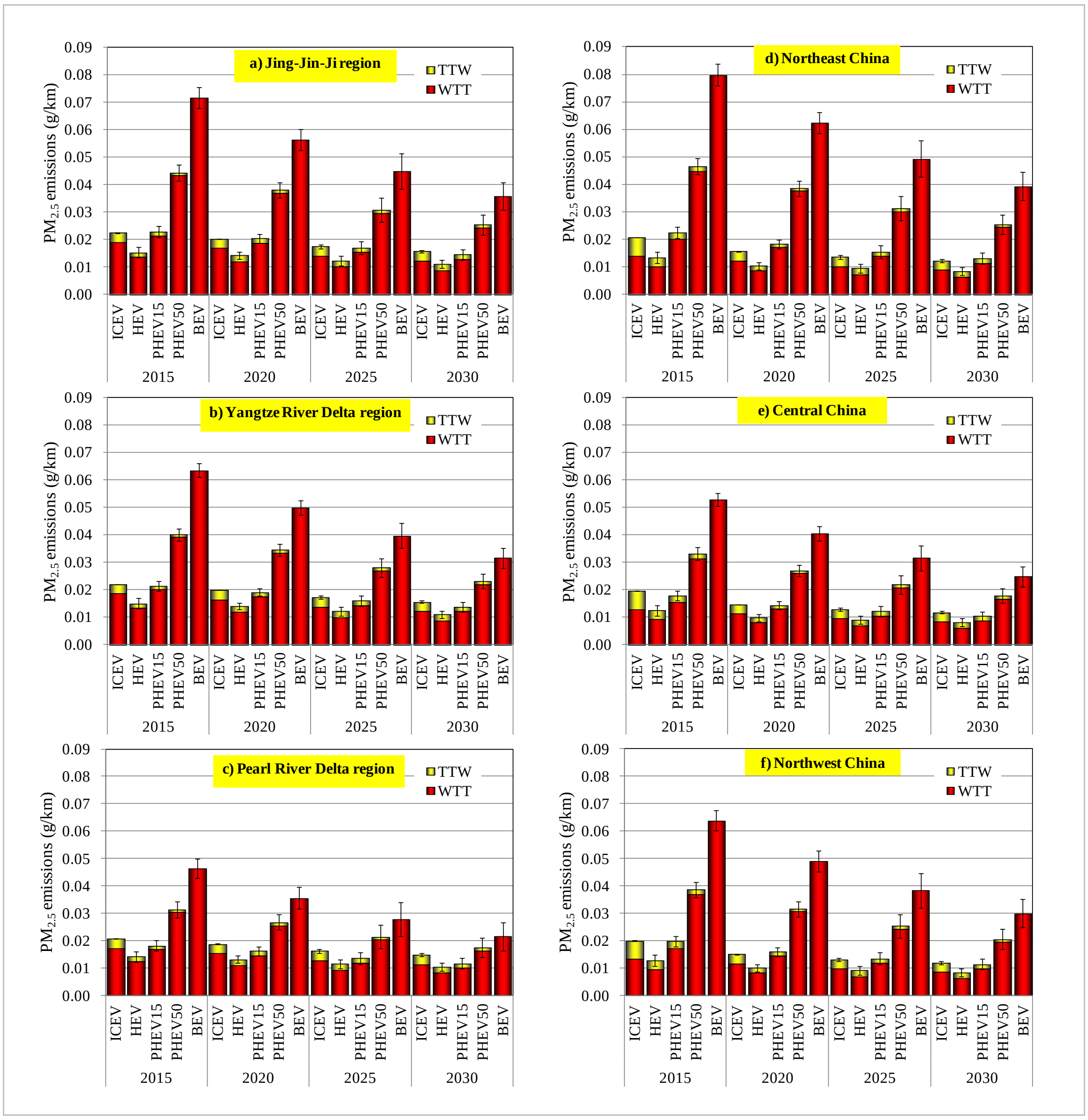

Figure 48: WTW PM2.5 emissions of LDPV in six regions of China, 2015-2030

2.2.2.5. SO_2

Figure 49 presents the per-kilometre WTW SO_2 emissions for HEV, PHEV15, PHEV50 and BEV relative to their ICEV counterpart by each of the six regions from 2015 to 2030. Similar to NO_X and $PM_{2.5}$ emissions, WTW SO_2 emissions mainly come from the upstream electricity producing stage. As the power generating efficiency and clean energy generation share improve, SO_2 emissions decrease gradually. Because of the different generation mixes, the six regions have significant differences on WTW SO_2 emissions: those regions with a high coal-based generation share like JJJ and Northeast China have higher SO_2 emissions than the regions with a relatively high clean energy generation share like PRD and Central China.

WTT SO_2 emissions of HEV and PHEV account for 85% to 95% of total WTW emissions. For BEV, all WTW SO_2 emissions come from the WTT stage, which is very high due to the substantial use of coal-based electricity generation in the generation mix. In 2015, HEV in all six regions could reduce WTW SO_2 emissions by 28% compared to ICEV. WTW SO_2 emissions of PHEV15 are comparable with ICEV only in PRD, while SO_2 emissions are considerably higher than ICEV in the other five regions. For PHEV50 and BEV, WTW SO_2 emissions in JJJ are higher by 133% and 287% than ICEV emissions, respectively; in PRD, they are 80% and about 177% higher than ICEV, respectively.

Under the projected generation mix in 2015, driving a PHEV15, PHEV50 or BEV in all the six regions has no advantages for SO_2 emission reduction. Regional variations will gradually increase over time. In 2030, HEV in the six regions could reduce WTW SO_2 emissions by 28% compared to ICEV. PHEV15 could achieve 11% and 20% WTW SO_2 emission reduction compared to ICEV in Central China and PRD, respectively, but still have slightly higher emissions than an ICEV in Northeast China with high coal-based electricity share and high emission factors of combustion boilers. WTW SO_2 emissions of PHEV50 and BEV in Northeast China are higher by 102% and 200%, respectively, compared to ICEV, and in PRD they are higher by 18% and 37%, respectively, as the best case. It shows that even in those regions with high clean energy generation share, PHEV50 and BEV could not reduce WTW SO_2 emissions compared to an ICEV.

In conclusion, to significantly reduce WTW SO_2 emissions, we need more clean energy generation and penetration of desulfurization technology in power plants when promoting PHEV and BEV in China.

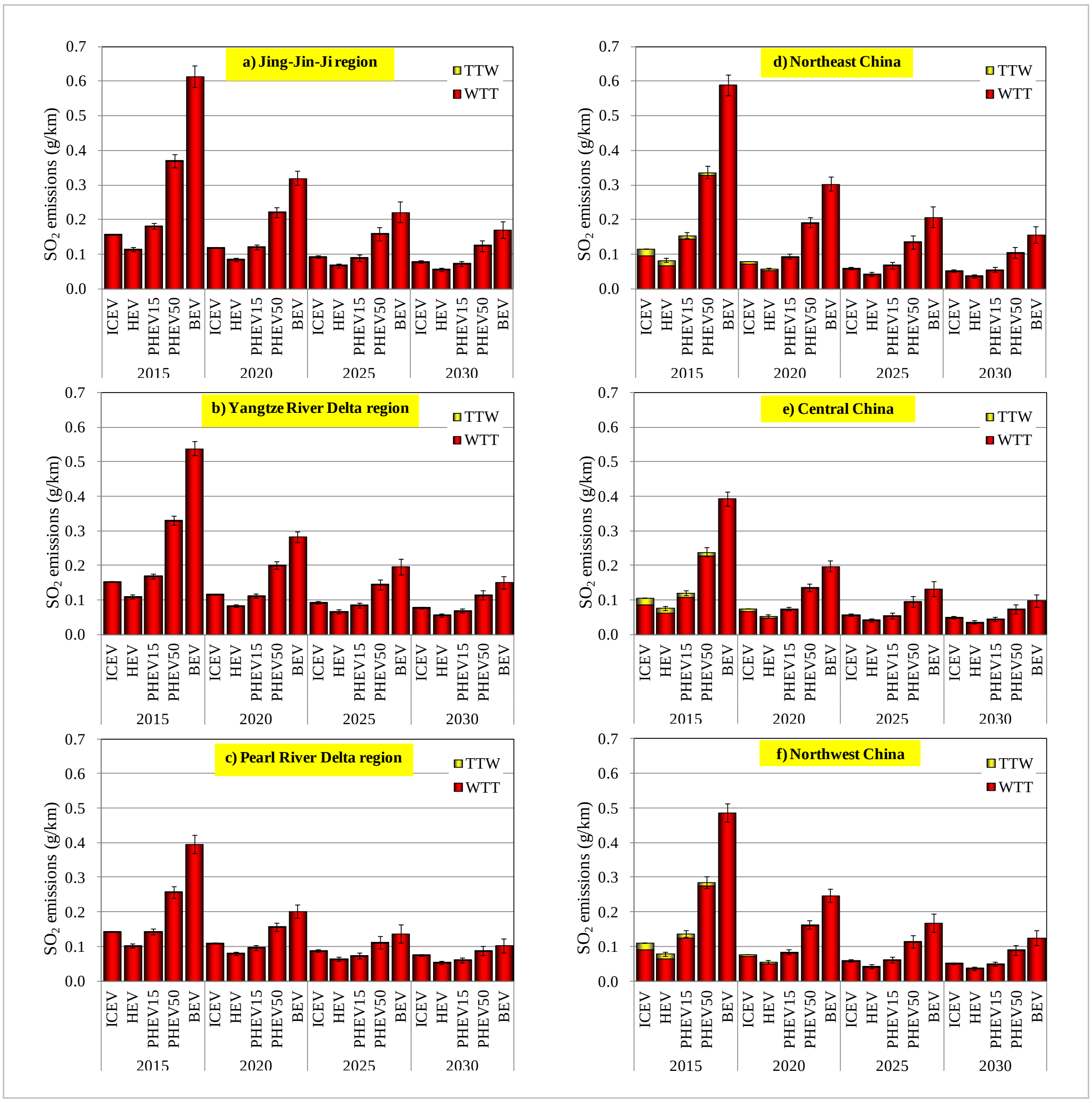

Figure 49: WTW SO2 emissions of LDPV in six regions of China, 2015-2030

As discussed before, for the pollutants dominated by WTT stage emissions, such as NO_X, $PM_{2.5}$, and SO_2, the coal-based electricity share in the generation mix and the emission factors of power plant combustion boilers are the two determinant factors for the entire WTW emissions.

2.3. Specific case study for Beijing

Besides analysing the national average level and the regional level based on power grid, we also evaluated the situation of Beijing as an individual local case. Beijing, the capital of China, has the most stringent standards for not only vehicle emissions, but also vehicle fuels. Moreover, for the purpose of controlling particle concentration, especially fine particles, to respond to the recent persistent haze pollution events, Beijing has to accelerate the release and implementation of all of its environment protection policies and measures. Therefore, Beijing is a special case of interest.

We set up two cases for Beijing: one is an analysis of the WTW energy use and emissions of a special natural gas development scenario from 2015 to 2030; and the other one is an evaluation of the possibility of using redundant wind generated electricity from other regions to serve the charging requirement of electric vehicles in Beijing under different penetration scenarios in 2015.

2.3.1. The natural gas case for Beijing

Although Beijing is a part of the Jing-Jin-Ji region, it has its uniqueness that a majority of electricity supply comes from outside the area. To lower the consumption of coal, all of the coal-fired power plants in Beijing will be changed to natural gas power plants, specifically utilizing the advanced natural gas combined cycle technology by the end of 2015. Compared to the fuel cycle parameters of JJJ, Beijing has a different generation mix and lower vehicle emission factors as a result of more stringent emission standards for vehicles.

Therefore, a projection of generation mix of Beijing from 2015 to 2030 was performed through a comprehensive study of the research literature and government plans, which is shown in Figure 50. A considerable share of coal-fired electricity used by Beijing will be replaced by natural gas electricity, but not all. Because a majority of electricity supply comes from the Jing-Jin-Ji power grid, which is dominated by coal-fired power plants.

Furthermore, a comparison of vehicle emission factors generated by vehicle emission factor model between hypothetical new standards (Tier 3, standard limits for vehicle emission factors in the U.S., shown in Table 9) in Beijing city and the JJJ was also conducted, as Figure 51 shows. The Beijing light-duty vehicle fleet will achieve Tier 3 (Bin70-Bin160, see Appendix A) by 2020, and will achieve Tier3 (Bin30) by 2030.The reduction with time of THC emission factors is the most remarkable, compared to the other three pollutants. The $PM_{2.5}$ emissions show a stable decrease from 2020 to 2030.

Finally, we made comparisons of WTW energy use and emissions of LDPV in Beijing and Jing-Jin-Ji region from 2015 to 2030.

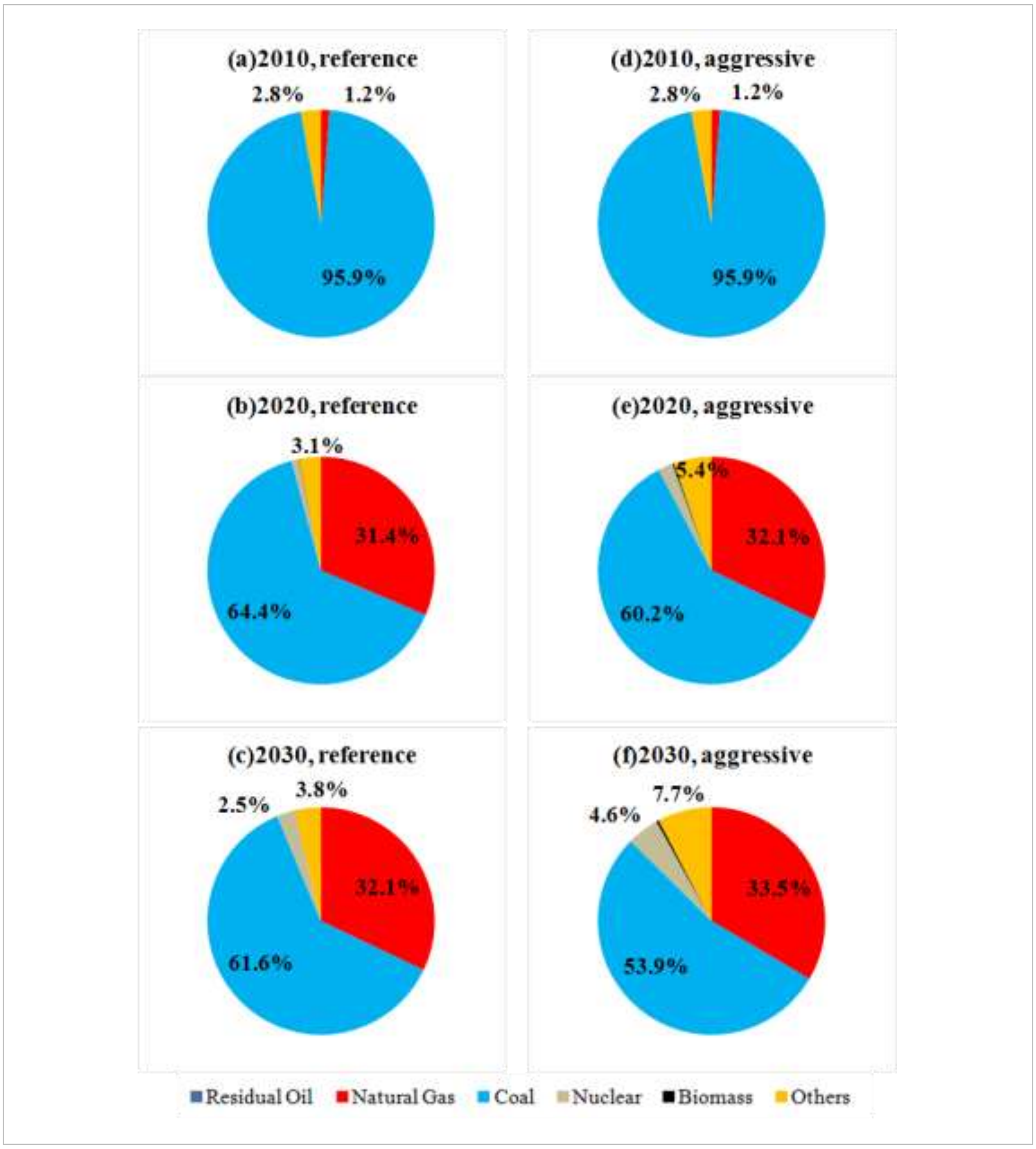

Figure 50: Generation mix in Beijing, 2010–2030

Table 9: Tier 3 standard limits for vehicle emission factors under FTP test

Bin	NMOG+NO_X (mg/mi)	PM (mg/mi)	CO (g/mi)	HCHO (mg/mi)
Bin 160	160	3	4.2	4
Bin 125	125	3	2.1	4
Bin 70	70	3	1.7	4
Bin 50	50	3	1.7	4
Bin 30	30	3	1.0	4
Bin 20	20	3	1.0	4
Bin 0	0	0	0	0

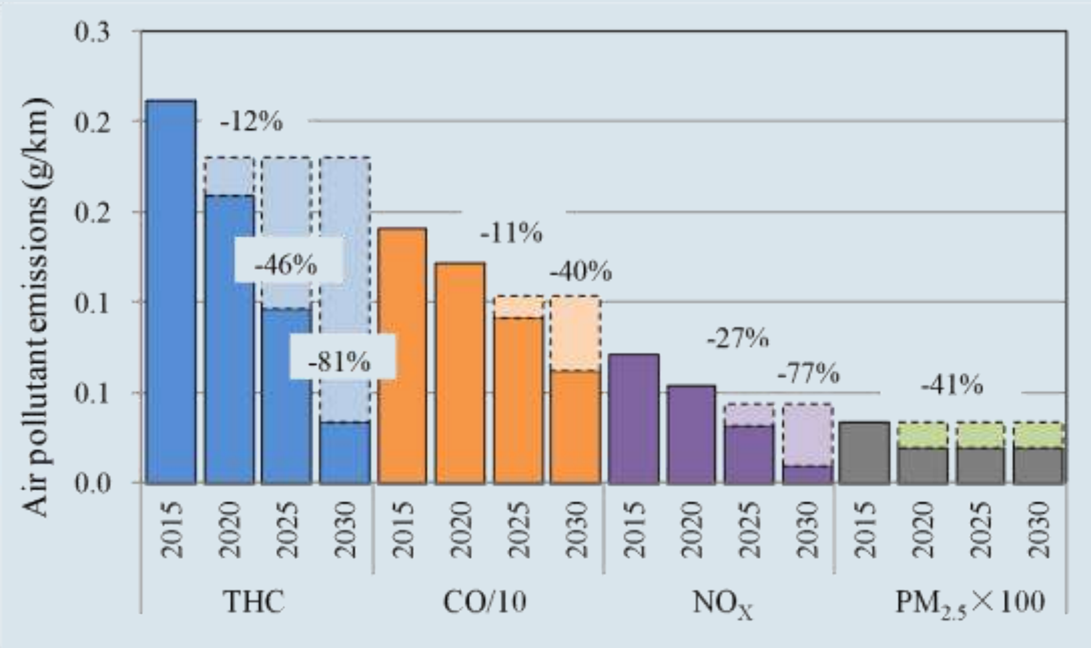

Figure 51: Comparison of vehicle emission factors under old and new standards in Beijing, 2015-2030

2.3.1.1. Petroleum consumption

Figure 52 shows the WTW petroleum consumption of LDPV in Beijing and JJJ. Compared to the counterparts in the Jing-Jin-Ji region, all vehicle types except BEV are almost identical for the WTW petroleum use in Beijing. Driving a BEV in Beijing can save about 20% of petroleum energy, compared to JJJ from the perspective of WTW analysis, although the absolute values of WTW petroleum use are negligible compared to the other kinds of vehicles.

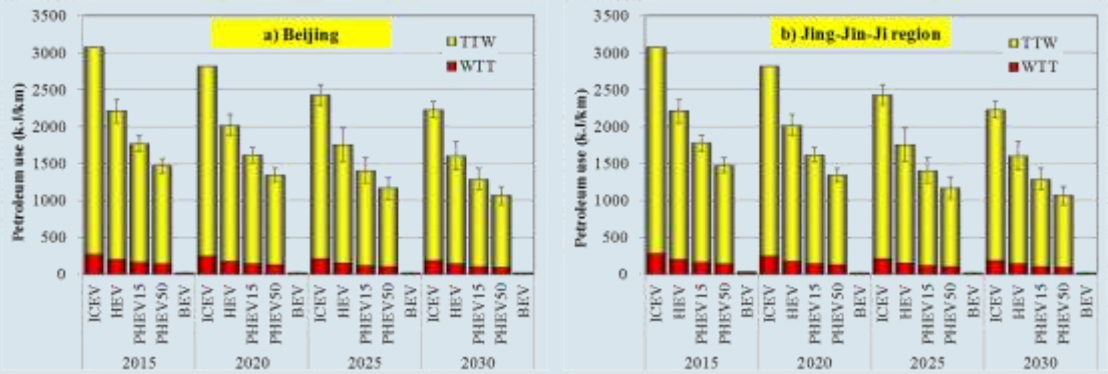

Figure 52: WTW petroleum consumption of LDPV in Beijing and JJJ, 2015-2030

2.3.1.2. Fossil energy consumption

Figure 53 shows the WTW fossil fuel consumption of LDPV in Beijing and JJJ. As for WTW fossil energy consumption, PHEVs in Beijing have a slight reduction compared to that of JJJ in 2015. While the result is opposite in 2030, PHEVs in Beijing has a slight increase compared to that of the Jing-Jin-Ji region.

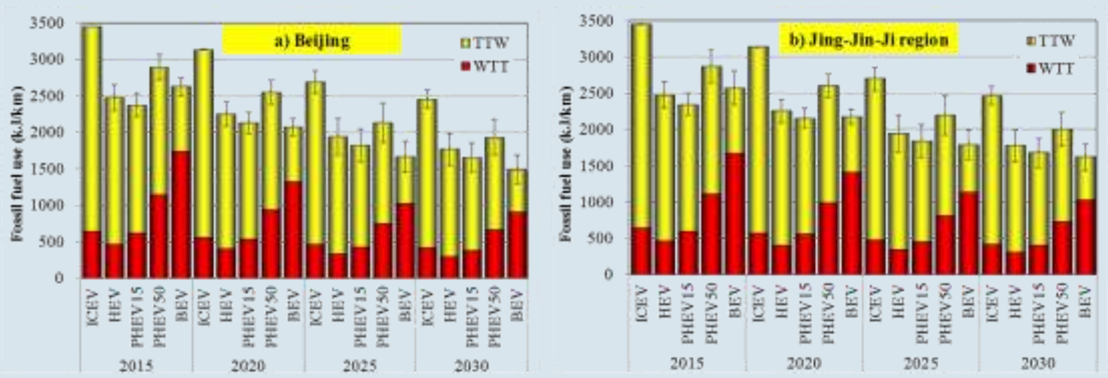

Figure 53: WTW fossil fuel consumption of LDPV in Beijing and JJJ, 2015-2030

2.3.1.3. CO_2

Figure 54 shows the CO_2 emissions of LDPV in Beijing and JJJ. In 2015, PHEVs in Beijing can reduce WTW CO_2 emissions by 4%-7% compared to that of the Jing-Jin-Ji region, and BEV can reduce 14% of CO_2 emissions under the same conditions. In 2030, PHEVs in Beijing can reduce by 1%-3% WTW CO_2 emissions compared to that of the Jing-Jin-Ji region, while BEV can reduce 7% CO_2. For both PHEVs and BEV, the reduction rate will decrease with time.

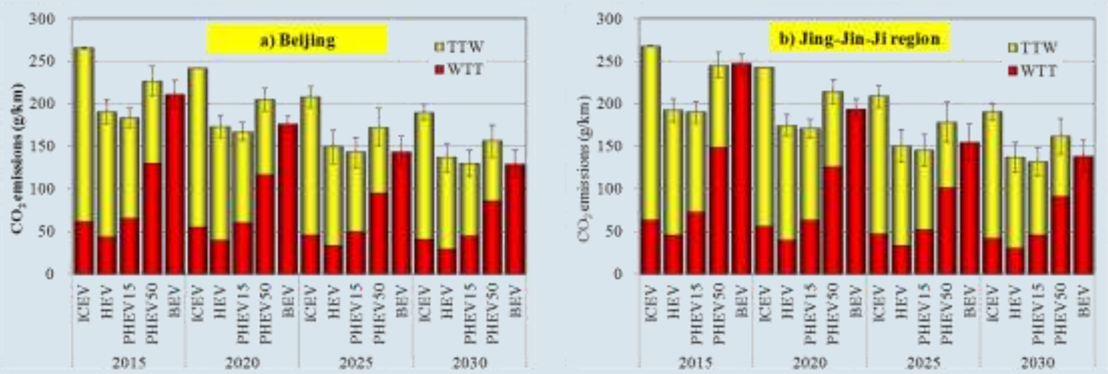

Figure 54: WTW CO2 emissions of LDPV in Beijing and JJJ, 2015-2030

2.3.1.4. VOC

Figure 55 shows the VOC emissions of LDPV in Beijing and JJJ. In 2015, PHEVs in Beijing provide a slight reduction of WTW VOC emissions, compared to that of JJJ; BEV can reduce VOC emissions by 13% under the same conditions. In 2030, ICEV, HEV and PHEVs in Beijing can lower WTW VOC emissions by 55-60%, compared to that of Jing-Jin-Ji region, while BEV can only reduce VOC by 2%. The significant VOC emission reduction of ICEV and HEV in the medium and long term is attributed to the more stringent vehicle emission standards in Beijing.

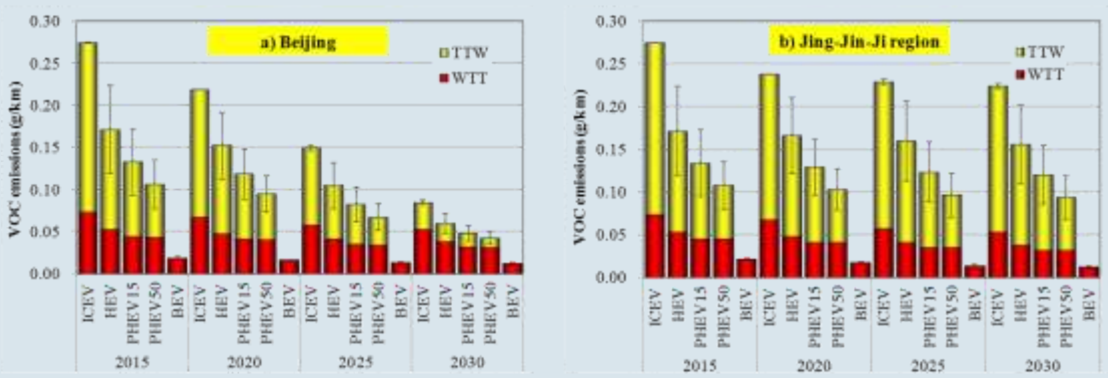

Figure 55: WTW VOC emissions of LDPV in Beijing and JJJ, 2015-2030

2.3.1.5. CO

Figure 56 shows the CO emissions of LDPV in Beijing and JJJ. For WTW CO emissions, BEV in Beijing have a 24% increase compared to that of JJJ in 2015, because the CO emissions from natural gas power plants are higher than from coal-fired power plants. In 2030, benefiting from more stringent vehicle emission standards, ICEV, HEV and PHEVs in Beijing can reduce WTW CO emissions by 35%-40% compared to JJJ.

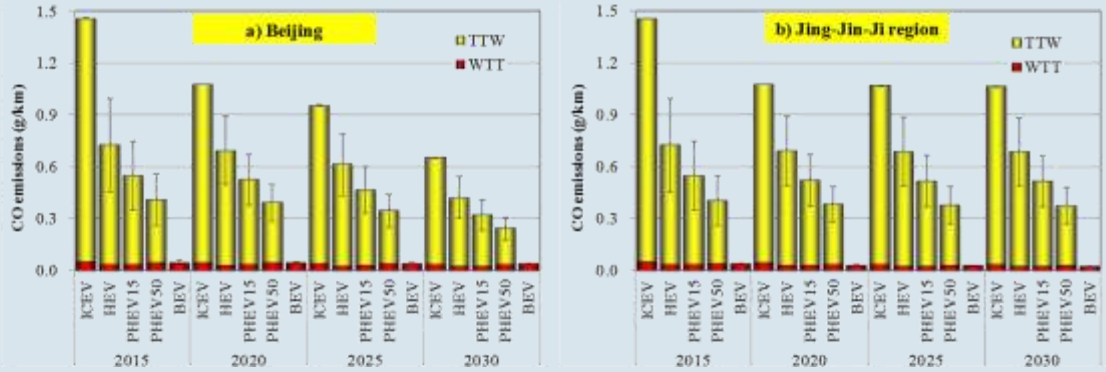

Figure 56: WTW CO emissions of LDPV in Beijing and JJJ, 2015-2030

2.3.1.6. NO_X

Figure 57 shows the NO_X emissions of LDPV in Beijing and JJJ. HEV, PHEVs and BEV in Beijing can reduce WTW NO_X by 5%, 15-20% and 30% respectively, compared to JJJ in 2015. In 2030, all of these vehicles in Beijing can reduce 20% WTW NO_X emissions compared to JJJ. If we compare the results from the perspective of time series, the data are different from each of the two regions: WTW NO_X emission reduction in Jing-Jin-Ji region is more significant than that of Beijing city.

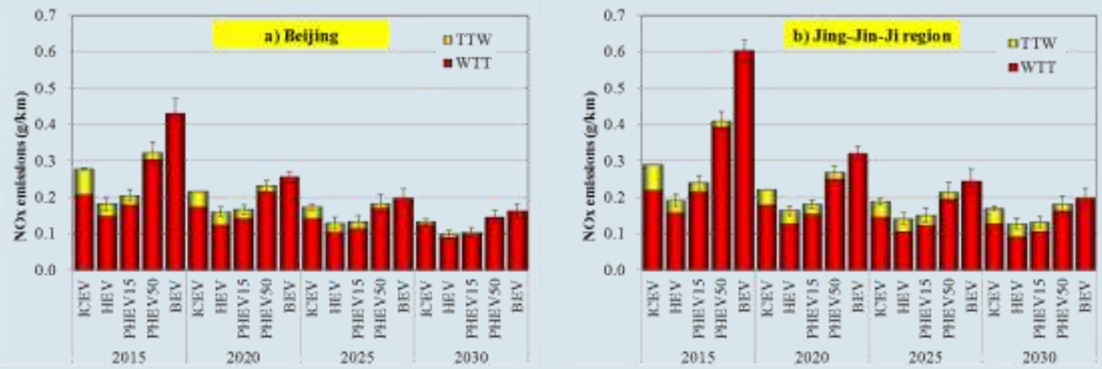

Figure 57: WTW NOX emissions of LDPV in Beijing and JJJ, 2015-2030

2.3.1.7. $PM_{2.5}$

Figure 58 shows the $PM_{2.5}$ emissions of LDPV in Beijing and JJJ. For WTW $PM_{2.5}$ emissions, ICEV, HEV, PHEVs and BEV in Beijing can reduce 7%, 7%, 18-26% and 31% respectively, compared to those of JJJ in 2015. In 2030, both ICEV and HEV in Beijing have a 13% reduction of WTW $PM_{2.5}$ emissions compared to JJJ, which are more remarkable than that in 2015. PHEVs and BEV in Beijing can reduce $PM_{2.5}$ by 18-22% and 22% compared to Jing-Jin-Ji region in 2030, which are comparable to that in 2015.

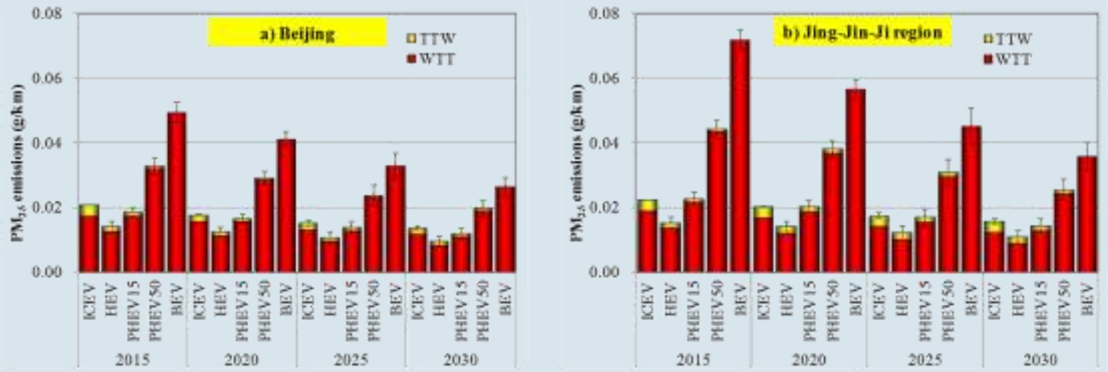

Figure 58: WTW PM2.5 emissions of LDPV in Beijing and JJJ, 2015-2030

2.3.1.8. SO_2

For WTW, ICEV, HEV, PHEVs and BEV in Beijing can reduce SO_2 emissions by 9%, 9%, 20-27% and 32%, respectively, compared to those of JJJ in 2015. In 2030, all of these vehicles have a slightly lower reduction ratio than those in 2015: ICEV, HEV, PHEVs and BEV in Beijing can reduce SO_2 by 4%, 4%, 13-20% and 27%, respectively, compared to the counterparts of JJJ.

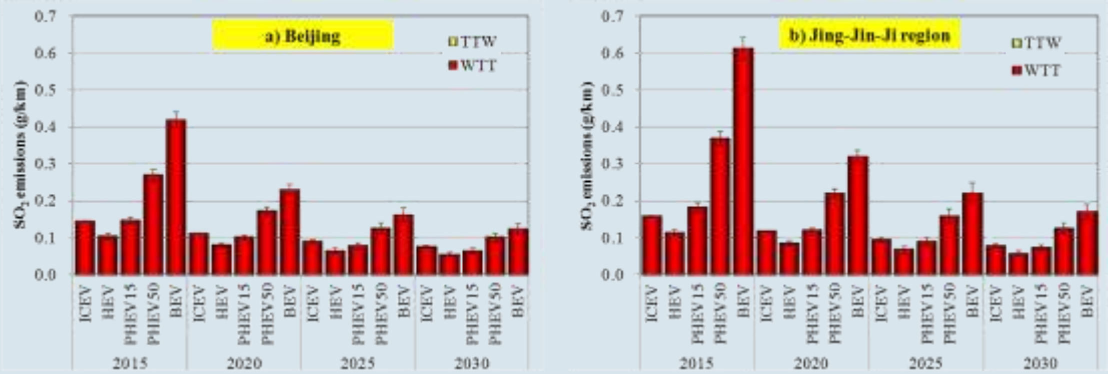

Figure 59: WTW SO2 emissions of LDPV in Beijing and JJJ region, 2015-2030

In conclusion, for the pollutants dominated by TTW stage emissions, such as VOC and CO, the emission reductions over time in Beijing are more significant than that of JJJ, because a more stringent vehicle emission standard will be implemented in Beijing. For the WTT-dominated pollutants, such as NO_X, $PM_{2.5}$ and SO_2, the emission reductions in JJJ are more significant than that in Beijing, because Beijing is projected to have a relatively stable natural gas electricity share in the medium and long term, while Jing-Jin-Ji region will have a continuous decrease of coal-based electricity share.

2.3.2. The wind power case for Beijing

In this case, we first selected the potential wind power exporters to Beijing, then calculated the approximate available redundant wind generated electricity of the selected regions and the charging requirement of electric vehicles under different penetration scenarios in Beijing, and finally compared the results of requirement and supply capacity.

2.3.2.1. Wind power generating development in North China

Among the known renewable and clean energy sources, wind power is one of those has been widely developed and utilized globally. At present, we can see wind turbines in every corner of the world, both onshore and offshore. It was reported by the Global Wind Energy Council that wind power market increased by more than 11% in 2012, compared to 2011. At the end of 2012, the global wind power generation capacity was 282.5 GW in total. From the perspective of global distribution, Europe has the greatest wind generation capacity currently. Some developed countries, such as Denmark and Spain, have considerable wind power share in their total electricity generation. Asia had the most rapid rate of growth of wind power generation capacity in the past several years, which is attributed to its rapid growth in China and India.

In China, large-scale wind power development started at the beginning of the 21st century, as Figure 60 shows. After a decade of rapid growth, wind power generation accounts for only a very slight proportion, about 2% of total electricity generation in 2012, as shown in Figure 61.

According to the China's 12th five-year plan for energy development, target of wind power electricity capacity in 2015 is 100 GW, which is larger by 40GW than the grid-connected wind generating capacity in 2012. It means that China needs to build more than 10 GW of new wind generating capacity per year from 2013 to 2015.

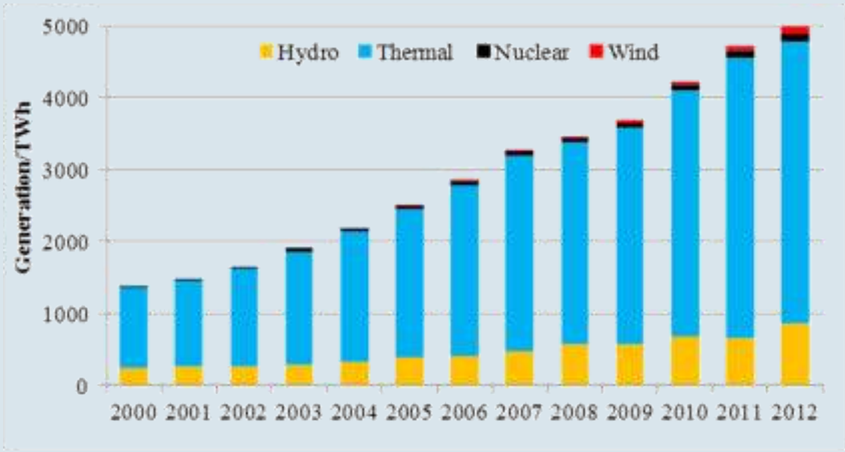

Figure 60: Power generation development in China, 2000-2012

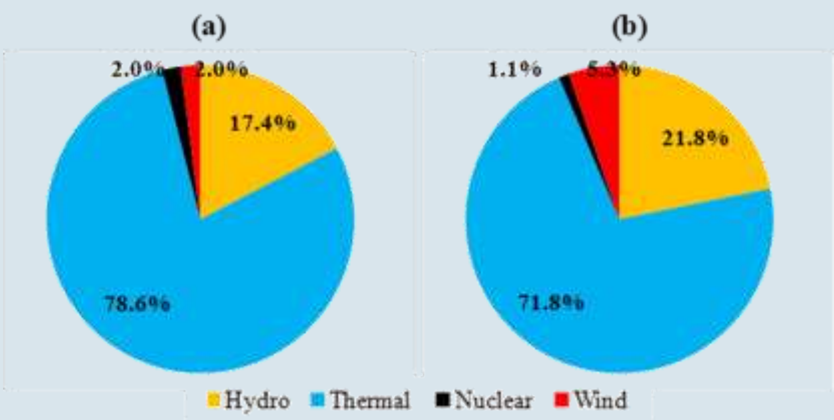

Figure 61: a) Generation share in 2012 and b) Generation capacity share in 2012

China has substantial wind power resources, which are mainly located in the three northern areas, namely, the North China Plain, Northwest China, and Northeast China, as shown in Figure 2. Specifically, there are six major wind power generating basins located in these areas. Coincidentally, Beijing is at the geographical centre of the wind-rich areas. Therefore, we wanted to evaluate whether the imported wind-based electricity could serve charging requirements under various penetration scenarios of electric vehicles in Beijing. As shown in Figure 63, we selected four provinces as potential wind power exporters with relatively abundant wind power sources and geographically close to Beijing: Inner Mongolia, Liaoning, Hebei, and Shandong.

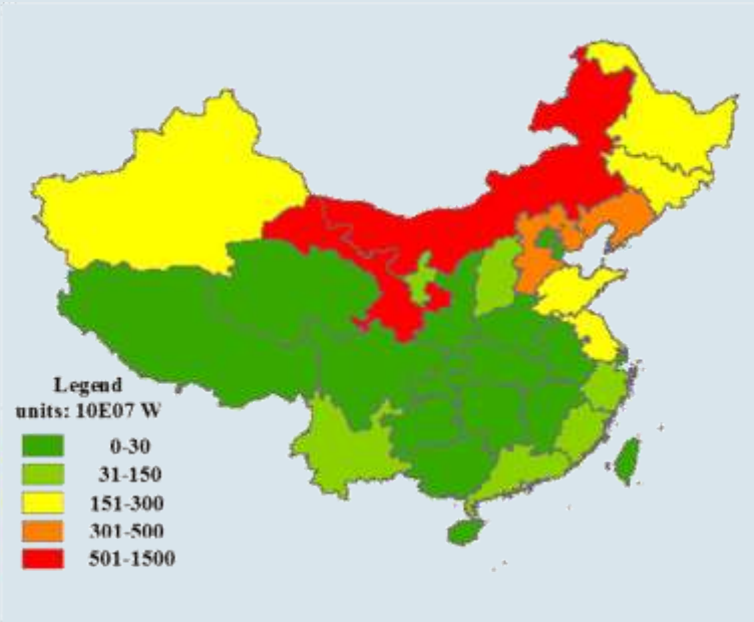

Figure 62: Wind generation capacity distribution of China in 2011

Figure 63: The potential wind power electricity exporters for Beijing

From the perspective of electricity exchange among provinces in China, net electricity exporters are more likely to be the wind power supporter suppliers to Beijing. As shown in Figure 64, Inner Mongolia exported more than 100 trillion Wh (TWh) of electricity in 2010, which makes it the best option to transfer its wind power electricity to Beijing, compared to the other three selected options--Liaoning, Hebei, and Shandong, which are all net importers of electricity.

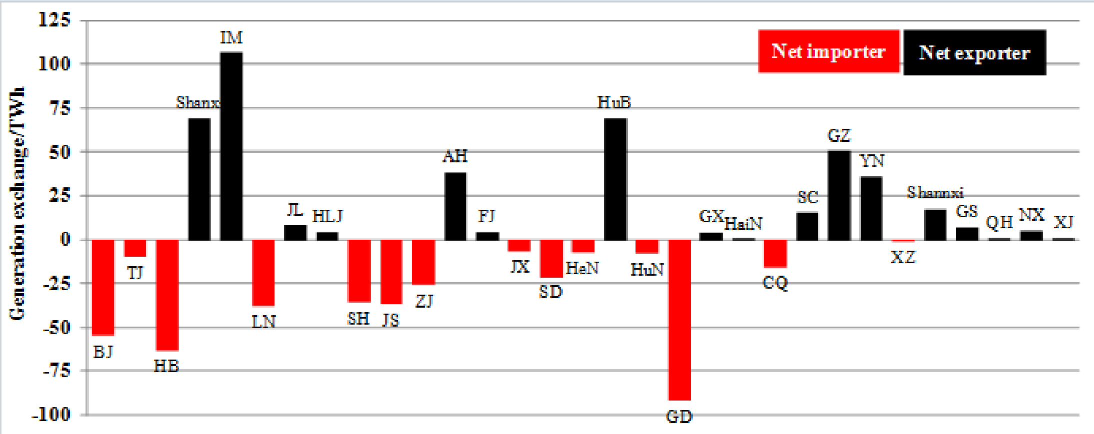

Figure 64: Electricity production and consumption of provinces of China in 2010

Although China has substantial wind power resources, and has also built substantial wind generating farms in the past decade, the wind-based electricity still cannot be used conveniently and effectively. One reason is that the spatial distribution of wind energy source is not well matched to the energy consumption distribution, and the other reason is the development of a transmission system of wind-based electricity lagged behind the construction of wind generating farms. Therefore, in the seasons with relatively abundant wind power, some wind farms have to abandoned part of their generated electricity to ensure the stability of the entire power grid.

The 2012 Annual Review and Outlook on China Wind Power counted the wind curtailment ratio of main areas for wind power and the number of wind farms in China, which is shown in Table 10. As we can see, wind curtailment is a common phenomenon for wind generated farms in China. Inner Mongolia has the greatest abandoned wind electricity of at least 5 TWh over all the counted areas in 2011.

Therefore, for this example, it is assumed that Inner Mongolia can export 5 TWh wind generated electricity to Beijing per year, for a simple calculation and preliminary comparison.

Table 10: Top 10 regional wind curtailment areas in China, 2011

Region	Wind curtailment ratio	Wind generation/ TWh	Abandoned wind electricity/ TWh
East of Inner Mongolia	22.99%	8.75	2.61
Jilin	20.49%	3.99	1.03
West of Inner Mongolia	17.51%	13.23	2.81
Gansu	16.99%	7.12	1.46
Heilongjiang	14.49%	4.39	0.74
Liaoning	10.34%	6.6	0.76
Xinjiang	3.21%	2.8	0.09
Hebei	3.09%	8.9	0.28
Shandong	1.46%	4.2	0.06
Ningxia	0.64%	1.3	0.01
Total		**61.28**	**9.86**

2.3.2.2. Penetration scenarios and charging patterns of electric vehicles in Beijing

In China, Beijing is one of the cities to implement more stringent pollution control measures for vehicles ahead of the national average level, such as alternate day driving rules based on vehicle license and the license control policy. These measures mitigate the vehicle tail pipe emissions but do not eliminate them. Under the continuously growing pressure of air pollution in Beijing, not only the Beijing local government, but also the Chinese central government has urged control of anthropogenic source emissions from various sectors such as industry, transportation, and area sources. With this as a background, promoting electric vehicles seems a win-win strategy for lowering oil consumption and reducing air pollutant emissions in the transportation sector.

As shown in Table 11, most of in-use electric vehicles in Beijing are taxis, commercial vehicles and special purpose vehicles in 2012 because they are easily regulated and controlled for the change from traditional liquid fuel vehicles to new energy vehicles. But they are apparently not the source of large-scale penetration of electric vehicles into the market in the future.as compared to personal passenger vehicles. Therefore, personal passenger vehicles will dominate both the reference scenario and the ambitious scenario in 2015. Through a comprehensive study of government policy, corporate plans, literature research, and news reports, two penetration scenarios of electric vehicles in Beijing in 2015 were projected.

It is noted that the potential of commercial vehicles and special purpose vehicles to use electricity may be affected by possible conversion to natural gas for sanitation vehicles in 2015.

Table 11: Penetration scenarios for electric vehicles in Beijing

Vehicle category	2012	2015 reference	2015 ambitious
Total	4837	30000	50000
Taxis	1200	6000	10000
Buses	360	4800	8000
Personal passenger vehicles	120	18000	30000
Commercial vehicles and special purpose vehicles	3087	1200	2000

Various types of electric vehicles have a variety of charging patterns, as Table 12 lists. Through a comprehensive study of manufacturers' data, research literature, and news reports, we separately calculated the charging requirement for those four kinds of electric vehicles.

Table 12: Charging patterns for electric vehicles

Vehicle category	Charging pattern
Taxis	capacity: 19kWh AER: 100km Driving 350km per day Three times per day
Buses	95kWh/100km 60000km/year
Personal passenger vehicles	capacity: 19kWh AER: 100km Driving 50km /day Once in two days
Commercial vehicles and special purpose vehicles	3300kWh in half a year

2.3.2.3. Results of different scenarios

For the two scenarios in year 2015, variation of vehicle penetration is the only factor of concern. It is assumed that battery technology, vehicle fuel economy, and charging efficiency would not change until 2015, which means the same kinds of electric vehicles in 2012 and 2015 have identical charging properties. Then, we calculated the charging requirement of electric vehicles in Beijing for the baseline scenario in 2012 and the two penetration scenarios in 2015.

As shown in Table 13, even under the ambitious penetration scenario of electric vehicles in 2015, electricity requirements of charging will be just 16.7%, compared to the abandoned wind generated electricity in Inner Mongolia in 2011 of about 5000 GWh, as roughly estimated earlier. This implies that if a feasible transmission, distribution, and storage system for wind generated electricity in Inner Mongolia exists, all electric vehicles in Beijing under a reasonable penetration projection can be fuelled with the clean and renewable wind power electricity.

Table 13: Results of charging requirement for different scenarios

Scenario	Charging requirement of Beijing/GWh	Ratio to wind curtailment of Inner Mongolia in 2011
2012_baseline	70	1.4%
2015_reference	500	10%
2015_ambitious	835	16.7%

3. The Energy Consumption and CO2 Emissions of LDPV in Full Life Cycle

3.1. Vehicle-cycle result of LDPV

3.1.1. Energy use and CO_2 emissions for battery

Figure 65 presents vehicle-cycle energy use and CO_2 emissions of major battery materials in China from 2010 to 2030. The energy results are quite different between virgin and recycled metals. 143 MJ energy are need to produce 1 kg of virgin nickel and 47 MJ needed for recycling that amount of nickel, while lead and copper have 75% and 42% energy reduction from recycling. In China the recycling rate of aluminium, copper and lead has only reached 19.8%, 35.7% and 24.5% , respectively, in 2010, and this needs to be enhanced. Merged into the recycled weight share, the combined energy use for each major metal is shown in Figure 65 a). In general, rare earths, graphite, cobalt, $LiPF_6$ and lithium oxide consume over 200 MJ/kg which is obviously higher than other materials. On the other hand, common metals like steel and plastics use no more than 50 MJ to produce one kg of material. Lithium oxide, the key carrier of ions, uses 80% of its vehicle cycle energy in roasting to remove magnesium (pyrometallurgical process). If evaporation of materials was completed by equipment heating but not naturally evaporation, the vehicle cycle result of lithium oxide will be higher. Aluminium electrolyzing needs large amounts of electricity while cobalt and graphite need high temperature roasting with coal and heavy oil. Among cathode metals, cobalt is leading in both fossil energy use and CO_2 emissions (212 MJ/kg and 21.4 kg/kg in 2010), followed by nickel (166 MJ/kg and 16.0 kg/kg in 2010) and manganese (63 MJ/kg and 6.4 kg/kg in 2010). Such a gap in energy consumption leads to significant differences in energy use for power batteries, which contain a large percentage of these metals.

Within the next two decades, the fossil energy use as well as CO_2 emissions of most materials will continue to decrease steadily because of technology improvement and increased recycling rate. As of 2020, vehicle materials production will save 10-35% (21% on average) life cycle energy and 13-65% (31% on average) by 2030. For example, aluminium will consume 74 MJ/kg of total fossil energy in 2030, only 41% of its 2010 value (181 MJ/kg). The main reasons are 1) recycling rate rises from 19.8% to 61%; 2) refinery technology improves by 5% (aluminium as a mature industry in China). Currently, if promoting aluminium to substitute for ferrous in vehicles for light weighting, the energy saving benefit is rarely seen. The parts may be 50% lighter but are 6 times more energy intensive, which may offset the energy saving by light weighting in the on-road stage. But the energy gap is less in the future decades and the promotion of light-weight vehicles in China and other countries (export) begin to gain benefit.

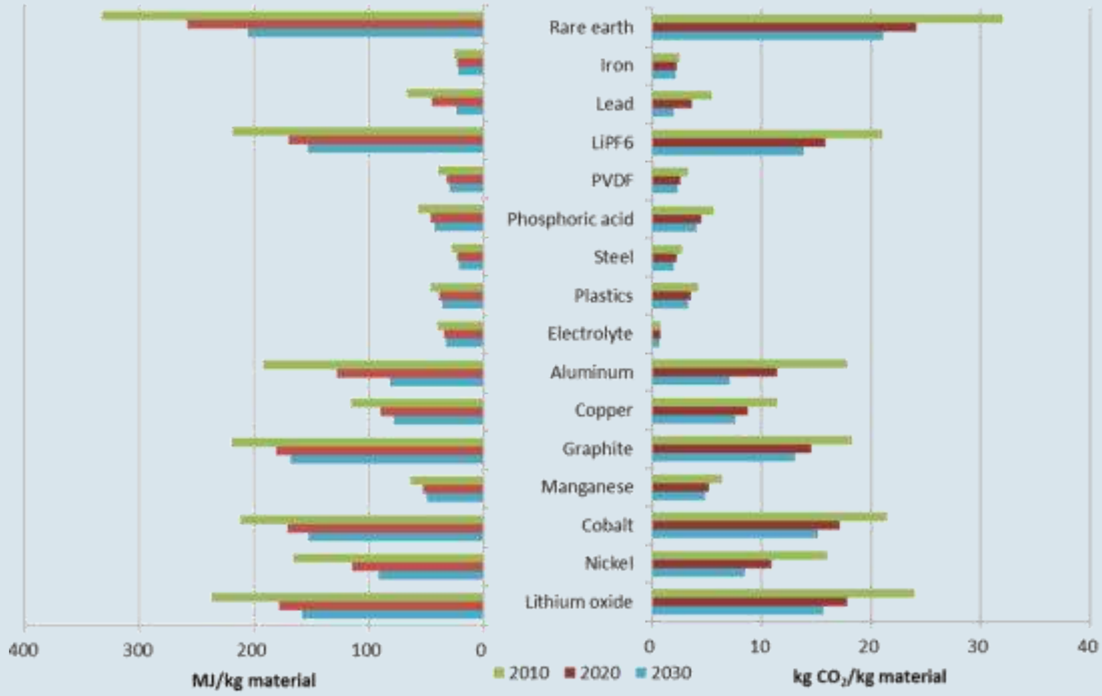

Figure 65: a) Vehicle-cycle fossil energy use and b) CO2 emissions of major materials, 2010-2030

Power batteries, which consist of many high energy intensive materials, are the key factors affecting vehicle-cycle energy use as well as CO_2 emissions among electrified vehicle models. This part will discuss energy use both in material production and battery module production. A module contains cells or several cell groups. Figure 66 illustrates vehicle-cycle energy use and CO_2 emissions of cell materials production batteries, excluding all module production energy use. In 2010, Ni-MH and Pb-Ac battery cells consume 92 MJ and 71 MJ to produce 1 kg of cells respectively, while the four Li-ion battery cells consume much higher energy ranging from 130 to 167 MJ/kg. Among the Li-ion groups, other sub parts except the cathode, such as graphite, aluminium, and copper contribute similar ratios. $LiCoO_2$ is the highest energy intensive cathode type, nearly 30% higher than that of $LiFePO_4$. This is primarily due to the materials energy intensity differences. For example, cobalt is the major material for the cathode of $LiCoO_2$ and NCM, which has relatively high energy intensity in material production; on the other hand, steel and manganese are major materials for $LiMn_2O_4$ and $LiFePO_4$, which consume less energy in weight specific production.

Those materials, such as aluminium, copper, cobalt and nickel, which heavily rely on electricity as the process fuel will emit large amount of CO_2. Chinese power grid is dominated by coal, which will emit the highest CO_2 at the same heat value compared to natural gas and petroleum.

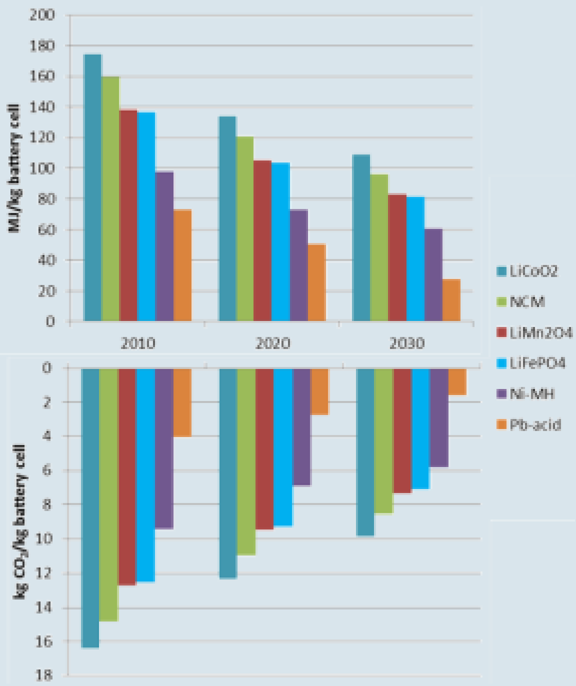

Figure 66: Vehicle-cycle energy use and CO2 emissions of cell materials, 2010-2030

When discussing the combined module results of different battery types between China and US, the trends are more complicated. The NCM battery (representative for Li-ion) consumes as high as 230MJ/kg of fossil energy use, which is 2.1 and 5.8 times as that of Ni-MH and Pb-Ac, respectively (Figure 67). Inside each module, material production contributes 46%, 57% and 78% of total energy in NCM, Ni-MH and Pb-Ac battery, respectively. The module assembly processes do change the relative ratio of total energy consumption for material production, especially in Li-ion batteries. NCM needs 124MJ to produce a 1kg module, compared to 49MJ for Ni-MH and 9MJ for Pb-Ac. Ni-MH and Pb-ac industries are mature and strictly regulated by local standards, although Ni-MH assembly is highly electricity intensive. On the other hand, Li-ion battery market is quite new and no scale effect benefit is achieved in China. So the energy consumption of each unit of battery cell or module is high. If compared to the US ANL results released in GREET 2012, energy use of the Pb-Ac battery (42MJ/kg) is only 2/3 of US level, in which the proportion of energy use for module assembly contributes more than in the US (22% to 11%) (Figure 67 a)). For Ni-MH energy use is the same in both countries at about 100MJ for 1 kg products, but also use more energy in module production, as reported by ANL. Li-ion battery use 3.3 times the vehicle cycle energy as the US level, especially in the module production part, nearly 12 times. The differences are due to a series of reasons:

1. ANL may only consider module assembly energy as module production energy (module assembly in this study is 8% of module production, seen in Figure 8), while in this study cell production is included.
2. A weight ratio of 33% (based on a company interview in this study) of the module cover (steel, low energy intensive) was used, which could decrease per kg energy use of the module.
3. Li-ion module production in China uses a large amount of electricity due to low scale effects and automatic facilities, in which some key processes like cell synthesis are quite energy intensive.

4. Database of China are from an enterprise interview (Li-ion) and local standards (Ni-MH and Pb-ac) of integrated energy use, which may overestimate the results by adding supplemental energy use like ventilation and lighting.
5. High energy intensive materials are largely applied in Li-ion batteries, which in China consume more energy because of low mining and refinery efficiency from many small mills.
6. Finally, the most important reason is associated with the local power grid structure, of which the high coal fired ratio (79% in 2010) and low generation efficiency, no more than 40%, enlarge the gaps of the present high process fuel (e.g. electricity) use levels between countries.

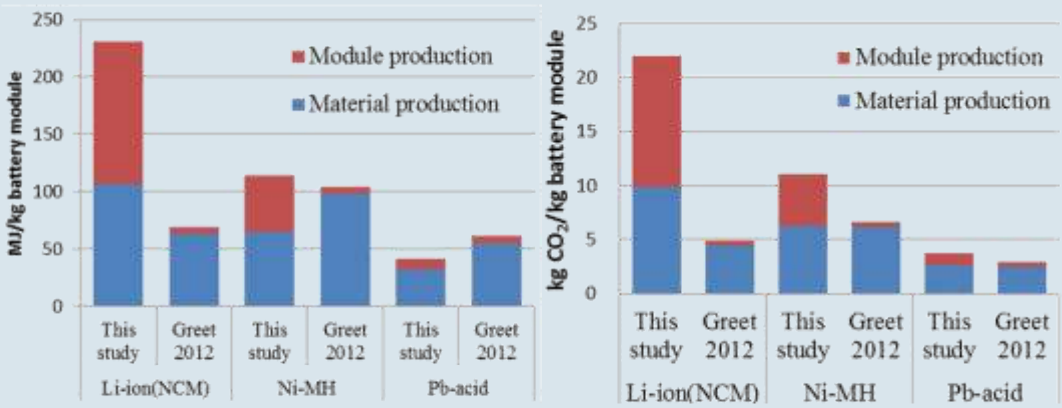

Figure 67: Comparison of vehicle cycle results of a) energy use and b) CO2 emissions of three typical batteries, 2010

The impact on the power grid structure is shown for CO_2 emissions in Figure 68 b). Full electricity dependence raises the module production CO_2 emission contribution rate to 55%, 35% and 20% for NCM, Ni-MH and Pb-Ac, respectively. The emission gaps between countries are enlarged (compared to the gap in energy use) because of the different power grid structures. More CO_2 is emitted from coal based power generation in China -- only one fourth of the CO_2 emissions in US are required to produce the same NCM batteries.

Although having higher per weight energy use, Li-ion batteries still benefit from the module level because of higher potential power densities (72-107 Wh/kg_module), compared to Ni-MH (40 Wh/kg_module) and Pb-Ac (20Wh/kg_module). On a vehicle, Li-ion modules need much less weight to achieve a target power goal. For example, a 35kWh NCM battery weights 328 kg but the same battery composed of Ni-MH weights 875kg, which is almost 50% of the weight of a passenger car. Figure 68 a) further presents vehicle-cycle energy use of various battery modules equipped on a 35kWh BEV. In 2010, the NCM battery has the lowest fossil energy use (71 GJ); while $LiMnO_4$, $LiCoO_2$ and $LiFePO_4$ consume 39%, 22% and 10% more fossil energy than NCM does. The ranking is different in Figure 66 where $LiCoO_2$ and NCM module energy burden decrease quickly because of high energy reserve capability, but $LiCoO_2$ is not so affected due to high energy intensity in material production. $LiMnO_4$ on the other hand is at the top of the energy use ranking because of its low energy density. Ni-MH batteries are seen in electric buses (high battery weight) and electric bicycles (low power demand) in China, which is considered to be one potential solutions for future pure BEV. Though its energy density is low for passenger car electrification, in the reference year Ni-MH battery energy usage is within the range of Li-ion group. But as time goes by, the storage ceilings gradually cap Ni-MH and technology improvement and scale effects for Li-ion batteries are more significant. By 2030, Li-ion batteries could cut 72-74% of the energy use reflected in the 2010 data (equivalent of a 50-75 GJ of fossil energy saving for a 35 kWh BEV (all electric range of 150-200 km)) while a Ni-MH battery could only achieve 50% reduction. At the same time, the energy density potentials of $LiFePO_4$ and $LiMnO_4$ may both increase to the theoretical limits and may fall behind NCM and $LiCoO_2$ if no key breakthrough happens. The trend in CO_2 emissions is similar to that of energy use for these batteries, as shown in Figure 68 b).

Figure 68: Vehicle-cycle a) energy use and b) CO2 emissions of battery EV, 2010-2030

A battery module installed in vehicles usually contains the power battery module and a low voltage battery, even on electrified vehicles. Pb-Ac batteries only act as a low voltage source for lighting, sensors and other auxiliary systems that are all quite less energy intensive. Taking NCM for example, battery demands (AER demands) determine the battery vehicle cycle impacts for total vehicle energy intensity, as shown in Figure 69. It shows a linear relationship to battery weight among vehicle fleets. Fortunately, in the next 2 decades, the foreseeable progress in battery power intensity, production efficiency, clean power propulsion and material recycling rates could lead to an obvious energy cut (50GJ and 70% reduction of NCM in the BEV, shown in Figure 69 a)) and CO_2 reduction (5 ton CO_2 or more per vehicle as seen in Figure 69 b)).The larger battery vehicles gain more benefits in the absolute energy decrease from the time series progression, which makes long AER BEV more competitive in vehicle cycle and total life cycle assessment.

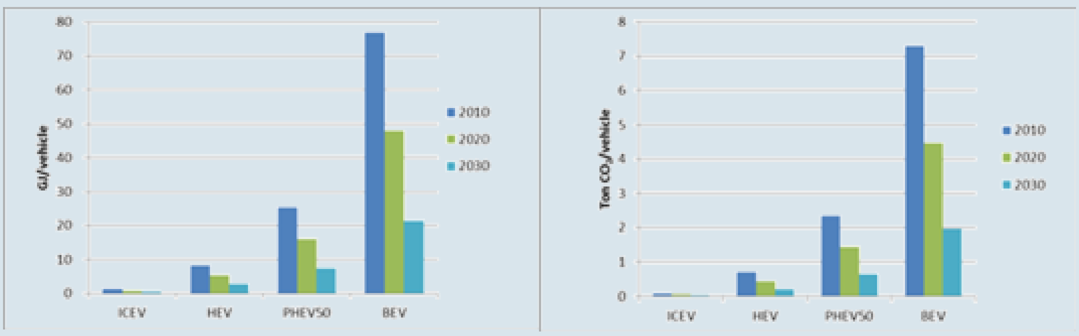

Figure 69: Vehicle-cycle a) energy use and b) CO2 emissions of NCM modules equipped on different vehicle fleet, 2010-2030

The on-board impacts of batteries need life cycle assessments of all vehicle components. This study applies ANL parameters from GREET 2012 including powertrains, body, chassis etc. and their compositions. Standing on the vehicle level and using similar life cycle methodology in battery assessment, the total vehicle cycle results of an ICEV and a 35kWh BEV are shown in the first two bars of Figure 70. The BEV in total uses 144GJ in the vehicle cycle, almost 2 times as much as ICEV`s. The battery module contributes 76GJ, which equals the total energy use of ICEV. Further the vehicle level assembly, displacement and recycling (ADR) are included with a fixed result of 17.2GJ, the same as the ANL simulation. Inside the ICEV, batteries are only applied for low voltage uses, accounting for less than 1% of total energy. But in the BEV, the ratio rises to 53% with a weight share of 22%. But as simulated in Figure 70, the fast decrease of battery energy use could benefit the BEV as time goes by. Processes as battery materials production, which is especially high energy intensive, and battery cell synthesis are the key energy control aspects of the EV from a life cycle perspective. In addition, if installing other types of batteries, other than NCM, or using an aluminium module cover instead of steel for vehicle light weighting purposes, the vehicle cycle results of the BEV could even be worse than expected. Also, cathode choices of lithium batteries may indirectly affect resource availability after boost development of electric vehicles.

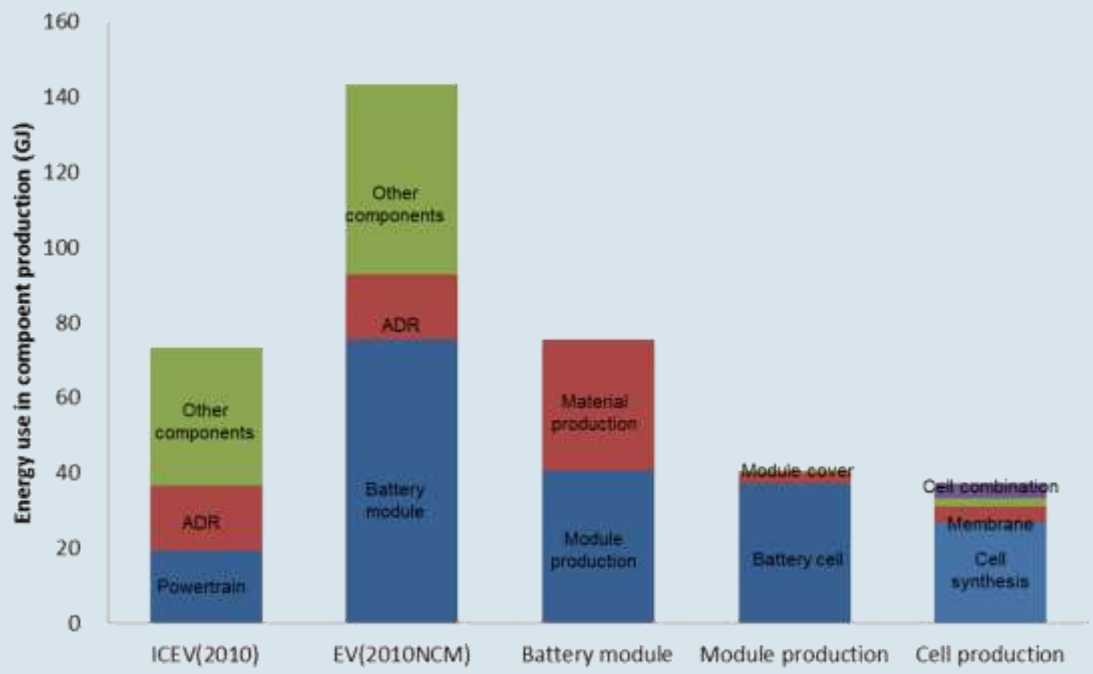

Figure 70: Contribution fraction of vehicle-cycle energy use of each sub components, 2010

3.1.2. Energy use and CO2 emissions for LDPV

Combined with the component data in Table 8, the energy use and CO_2 emission results in each part of the vehicles are shown in Figure 71. The main changes occur in powertrain parts among different vehicle fleets. Due to the large amount of steel and iron use, the non-battery parts are lower energy intensive than the batteries, though parts like body and chassis are heavier than the battery module. Along with the material production progress, the new vehicle produced in 2030 would have cleaner vehicle cycle results.

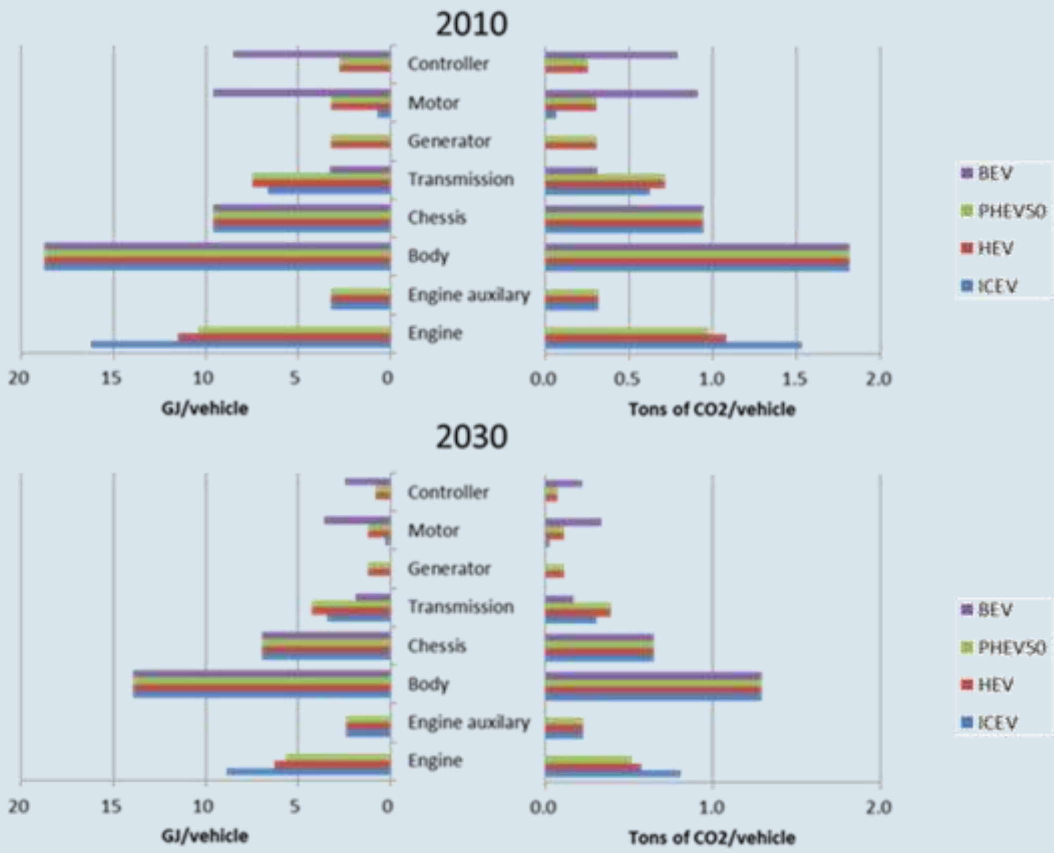

Figure 71: Contribution rate of each process in battery module production, 2010 and 2030

When all of the vehicle parts are aggregated together, we present the vehicle-cycle fossil energy use of HEV, PHEV50, BEV and ICEV in Figure 72, and CO_2 emission results in Figure 3 8. It should be noted in these two figures that only the NCM battery is used for different electric vehicle technologies. In 2010, the baseline ICEV consumes 70 GJ of fossil energy use, while HEV, PHEV50 and BEV increase their fossil energy use gradually to 80, 96 and 136 GJ, respectively, which are 14%, 37% and 94% higher. Clearly, the BEV cuts its energy use by eliminating the engine; however, adding Li-ion batteries increase by a large amount the fossil energy presently used. Over the next 20 years, the technology improvement in the Li-ion battery industry should reduce the gap in energy use between the ICEV and BEV significantly. For example, by 2030, the fossil energy use would be reduced to 61 GJ for BEV, lower by 55% relative to 2010 data. The value is still higher than that of the ICEV (48 GJ); however, the gap is much smaller at 26% (versus 94% in 2010).

Not surprisingly, the battery module is the biggest contributor in BEV in terms of energy use. In 2010, the NCM battery module contributes as high as 52% of total energy use for a BEV, surpassing the sum of all other vehicle components. Inside the battery module, upstream material production and battery module production and assembly almost share equally in fossil energy use. Furthermore within the battery module production and assembly, the production of a single battery cell itself contributes about 20% of total fossil energy use for a BEV, which is the leading energy consuming process in the BEV vehicle-cycle analysis.

In 2010, an ICEV emits 6.8 tons of CO_2, while a BEV emits 16.2 tons, which is 138% higher than an ICEV, as seen in Figure 73. It should be noted that the gap in CO_2 emissions between an ICEV and a BEV is larger than that of fossil energy use. This is primarily attributed to the fact that several processes (such as battery module production, lithium oxide and cobalt refining processes) involve large amounts of electricity consumption. Due to the fact that coal power is dominant in China, the burden of CO_2 emission is heavier for the BEV than that of energy use.

Figure 72: Vehicle-cycle energy use of ICEV, HEV, PHEV50 and BEV, 2010-2030

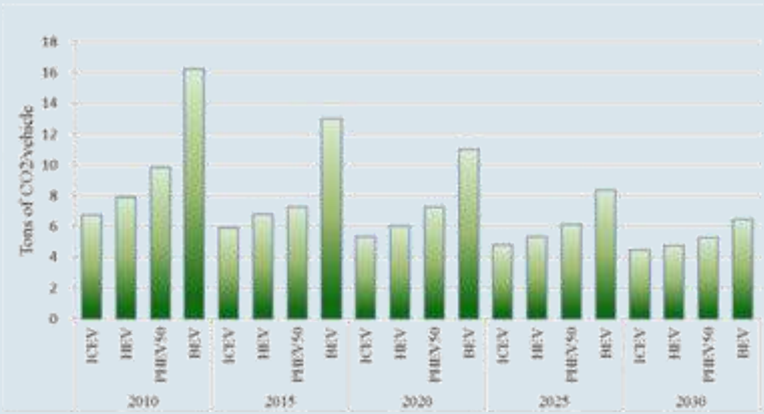

Figure 73: Vehicle-cycle CO2 emissions of ICEV, HEV, PHEV50 and BEV, 2010-2030

3.2. Full life cycle result of LDPV

3.2.1. Full life cycle result in China

3.2.1.1. Fossil energy consumption

We present full life-cycle energy use of three different electric vehicle technologies (HEV, PHEV50 and BEV) compared to the ICEV in Figure 74. Full life-cycle includes fuel-cycle (WTT and TTW) and vehicle-cycle; the total life time mileage travelled for a passenger vehicle is assumed at 200,000 km. All electric vehicle technologies achieve savings in full life-cycle energy use relative to the ICEV. In 2010, a HEV could save 20% of total energy use, followed by BEV (13%) and PHEV (11%). Due to the penalty in electricity generation and battery production stage in China, the BEV could not show significant reduction in energy use. However, as technology improves in the battery industry and more renewable power becomes available in China, BEV will show more advantage in life cycle energy use over ICEV. By 2030, BEV could achieve 40% reduction in full life-cycle energy use over ICEV, and 24% energy saving over HEV.

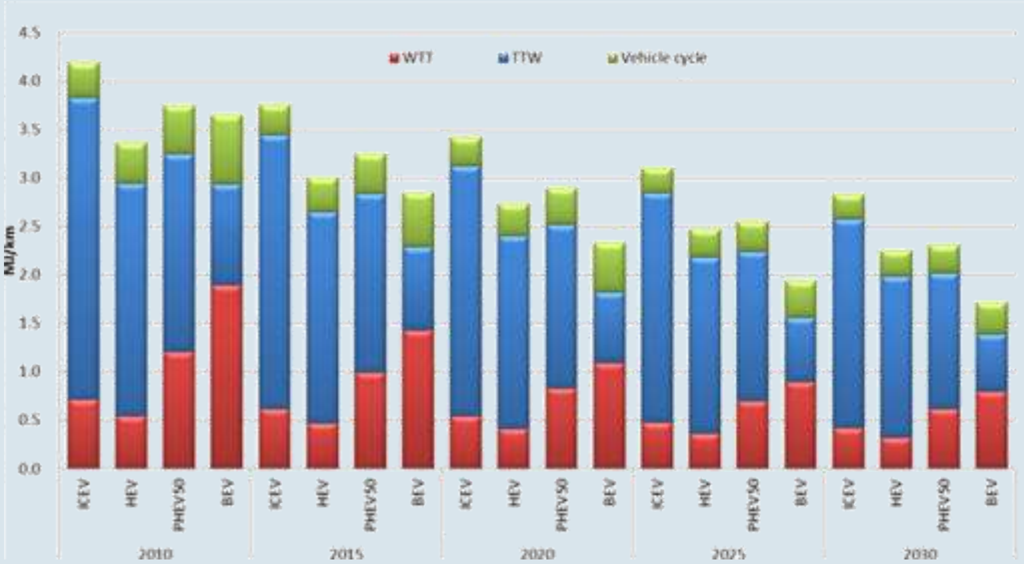

Figure 74: Full life-cycle energy use of ICEV, HEV, PHEV50 and BEV, 2010-2030

Figure 75 illustrates the contribution of three stages (WTT, TTW and vehicle-cycle) to full life-cycle energy use from 2010 to 2030. The share of energy use for each stage is relatively stable over time. For example, the vehicle-cycle stage for an ICEV remains 8-9% of total life-cycle energy use over time. However, vehicle-cycle stage for a BEV contributes a significantly higher share than that of ICEV, at 18-21% within the next two decades.

Figure 75: Contribution of each stage to total life-cycle fossil energy use of ICEV, HEV, PHEV50 and BEV, 2010-2030

3.2.1.2. CO2 emissions

The results of life-cycle CO_2 are somewhat different from fossil fuel (Figure 76). At the current stage, BEV and PHEV50 could not achieve full life-cycle benefit in CO_2 emissions over ICEV. In this study, we estimate that in 2010 BEV emits 8% more CO_2 per km than ICEV does. However, HEV could still achieve 19% reduction in CO_2 emissions relative to ICEV. This is again due to the fact of coal power emitting more CO_2 than any other fuels. As technology improves in the battery industry and more renewable power becomes available in China, the BEV will show an advantage in CO_2 emissions over ICEV. By 2030, the BEV could achieve a 27% reduction in full life-cycle CO_2 emissions over ICEV, and also be slightly lower than the HEV. The results indicate that the effort to mitigate CO_2 emissions is more difficult than reducing energy use.

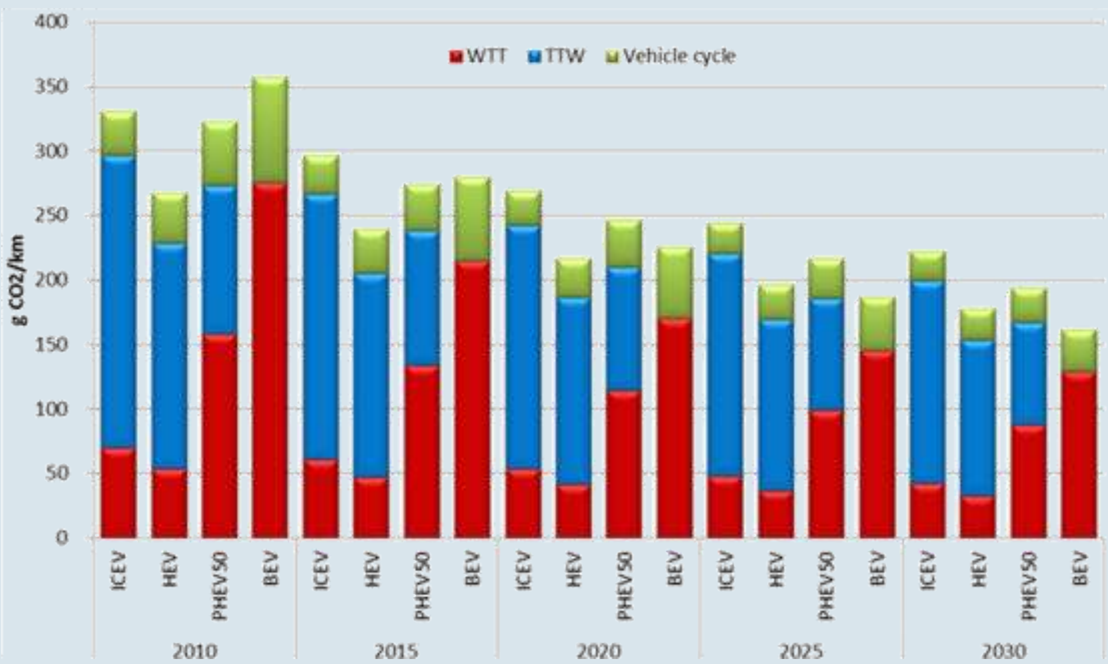

Figure 76: Full life-cycle CO2 emissions of ICEV, HEV, PHEV50 and BEV, 2010-2030

3.2.2. Full life cycle results in PRD and JJJ

3.2.2.1. Fossil energy consumption

In the lifetime of a vehicle (e.g., a model-year 2010 sedan), the vehicle-cycle energy use is fixed following its production; however, the fuel-cycle energy use only occurs when the vehicle starts to drive, and such energy consumption will accumulate over time. With the improvement in oil refinery efficiency, electric generation efficiency and promotion of more renewable power, the marginal increment rates of fuel-cycle energy use of a sedan will decline. Figure 77 illustrates accumulative full life-cycle energy use during the lifetime of a model-year sedan with different technologies, and Figure 78 presents the CO_2 results. Due to a significant difference in electricity generation mix by region in China, we further present the national average, and two cases for specific regions: one is Pearl-River Delta (PRD) region with much more renewable power (coal power contributing only 60% in 2010), and the other is Jing-Jin-Ji (JJJ) region with extremely high coal power share (about 95% in 2010).

The life cycle energy use at the start of vehicle lifetime is not zero, which is defined as the energy use of vehicle-cycle stage (with NCM battery for electric vehicles). Clearly, BEV has the highest starting energy use in 2010 due to the fact of higher energy use at vehicle-cycle stage. As HEV, PHEV50 and BEV all achieve fuel-cycle energy use reduction over ICEV; they will reach the break-even point of accumulative fossil energy use after the vehicles travel to some specific mileage. Nationwide within 1-2 years of use, HEV will reach the break-even point. While PHEV and BEV need 2-3 years and 4-5 years, respectively. When reaching 200,000 km lifetime (around 15 years), HEV, PHEV50 and BEV could save 19%, 13% and 23% of total fossil energy (on assumption that the battery module would operate for that long time). It should be noted that these results are for a current model-year sedan. With the production of improved models (e.g., model-year 2020 sedan), the break-even point for BEV would come sooner due to two major factors: first the start vehicle-cycle energy use decreases quickly, and second the power generation becomes cleaner and more efficient.

For PRD region with its higher renewable power mix, model-year 2010 BEV will reach the break-even point of total energy use much sooner, in less than 3 years while it might take a model-year 2010 BEV as long as 6-7 years to reach the break-even point in the JJJ region with its extremely high coal power share. Furthermore, a BEV will not achieve a cumulative energy saving advantage over HEV in JJJ regions with high coal power share.

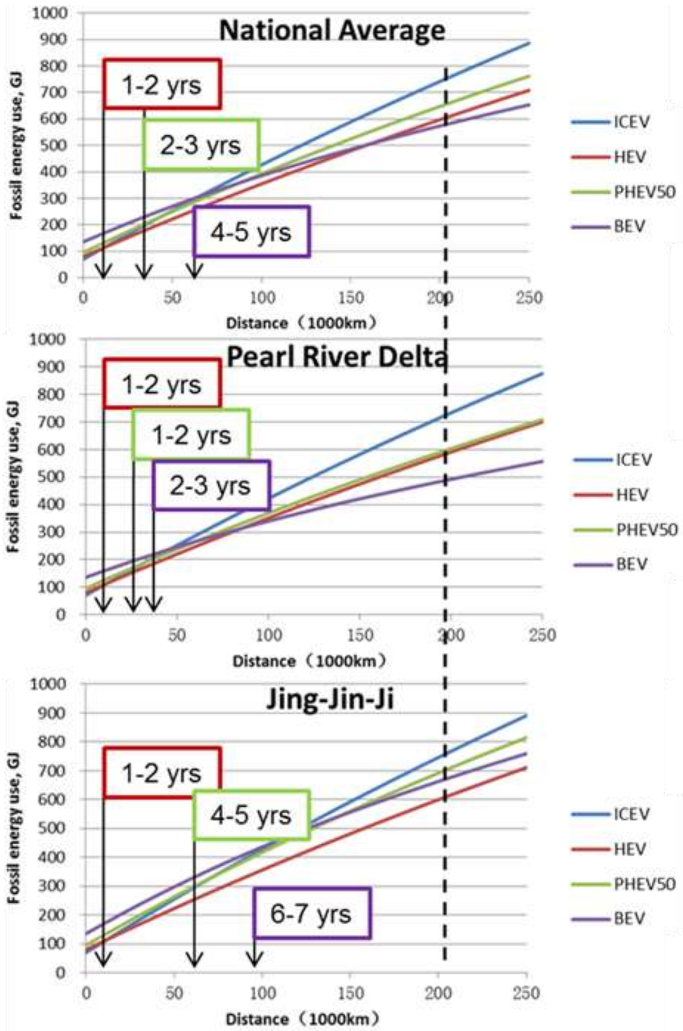

Figure 77: Accumulative full life-cycle fossil energy use during lifetime of a model-year 2010 sedan with different technologies

3.2.2.2. CO_2 emissions

To reach break-even point for accumulative CO_2 emissions is a much more difficult task for BEV. As shown in Figure 78, it takes model-year BEV as long as 8-10 years to reach the break-even point of CO_2 at the national average level, and it will never achieve break-even point in JJJ region for the current model during the lifetime of the vehicle use. PHEV50 and BEV have small benefit relative to ICEV in terms of accumulative life-cycle CO_2 emissions, within 5%. In the near future, HEV is a better option to mitigate CO_2 emissions in those regions with high coal power share. In the PRD region with current renewable power mix, BEV will reach break-even point of CO_2 emissions 2-3 years sooner than the national average. It can also achieve similar cumulative life-cycle CO_2 emissions decrease over HEV. Therefore, in those regions with cleaner power, promotion of PHEV and BEV could be a good option to mitigate the CO_2 burden.

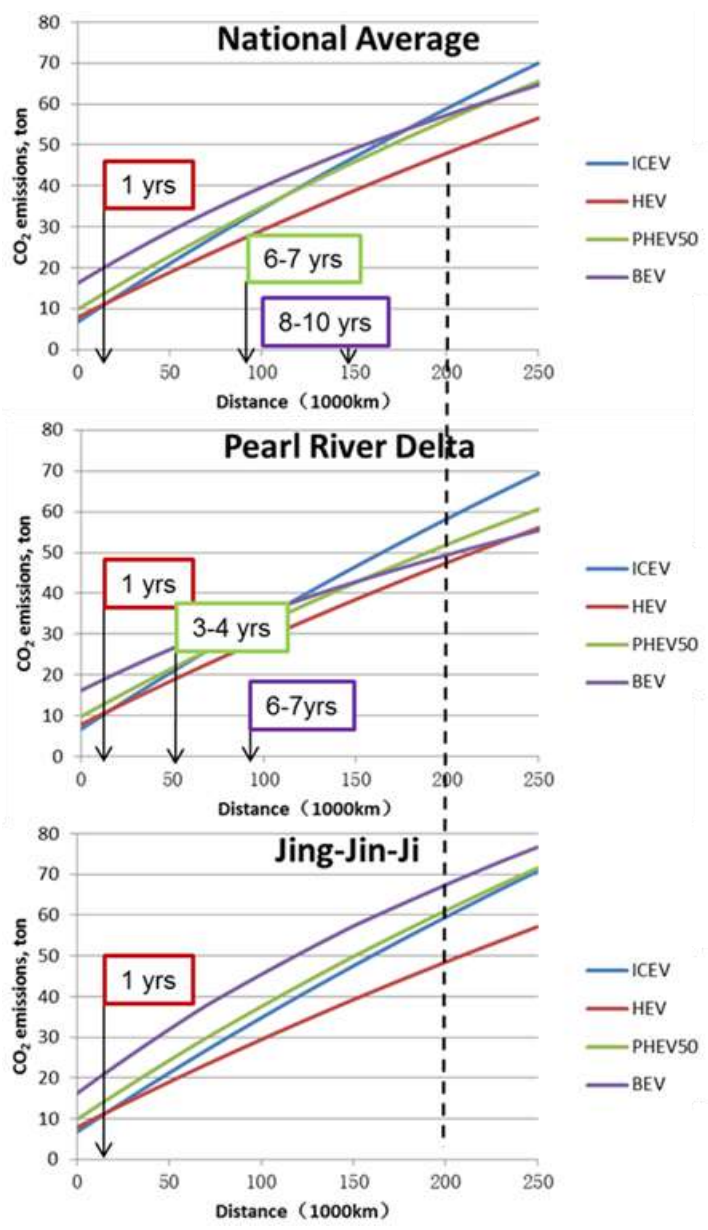

Figure 78: Accumulative full life-cycle CO2 emissions during lifetime of a model-year 2010 sedan with different technologies

4. Vehicle Fleet Energy Saving and Emissions Reductions

4.1. The current penetration status of HEV/PHEV/BEV in China

As of August 2012, the "Ten City and Thousand Units" demonstration program of China was completed. During the last 3 years, 25 cities operated 27,400 new energy vehicles, including HEV, PHEV and EV, which accounts for no more than 50% of the initial target. Among the so-called new energy vehicle catalogue, the high technical substituted model like PHEV and EV have quite limited growth rate, applied in different areas in different cities, respectively. As shown in Figure 79, PHEV/EV sale growth in the last three years in major demonstration cities is obvious but not in regular. Beijing implements pure EVs in taxi fleet but rare in private use. Shenzhen and Hangzhou penetration of PHEV and EV are mainly supported by government policies, which are not relied on the market behaviour themselves. The development of PHEV/EV has large uncertainty and stay at low fleet level both in annual sales and aggregate fleet number compared to the HEV, especially in light duty vehicle models. In this program, the central government offered 2.7 billion RMB for a purchase subsidy plus 3 billion RMB was provided by the local government.

On the other hand, the electric vehicle penetration mainly happened in heavy duty types, such as city buses. If looking at the share of light duty electric vehicle in annual sales in major cities, shown in Figure 80 (traditional HEV included), the absolute ratios are quite low, no more than 0.3%. At the same time light duty vehicles have already dominated the ownership population and new vehicle sales in China. It is clear that well developed areas have faster growth rate than the national level, but the level is only 1/10 of that in Japan.

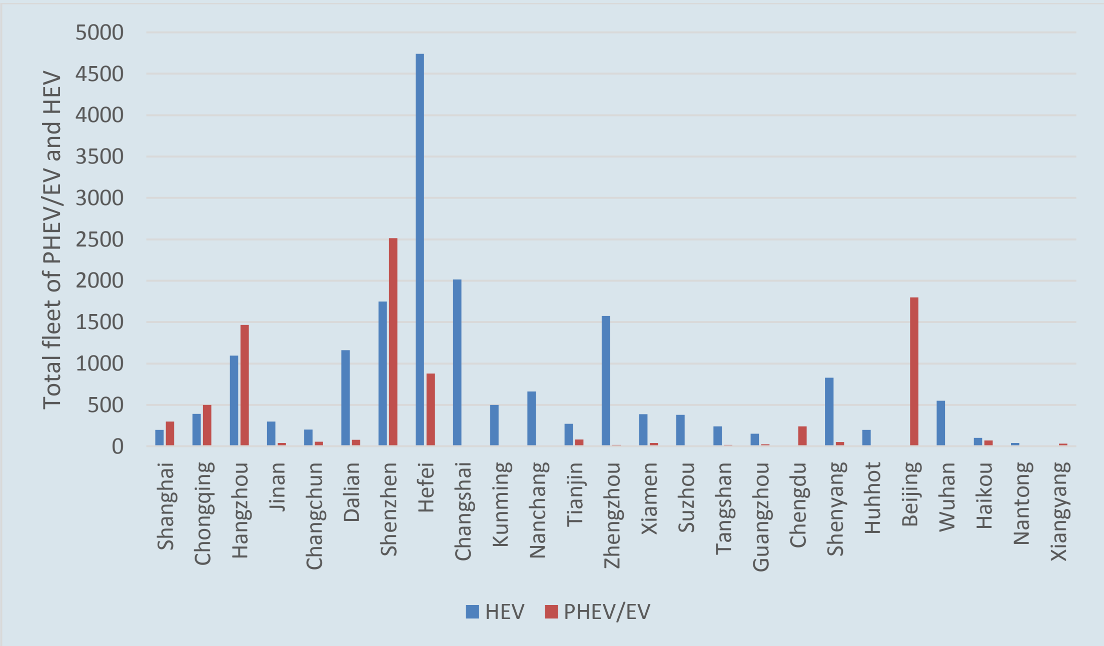

Figure 79: Total PHEV/EV and HEV sales in 25 cities, as of Dec. 2012

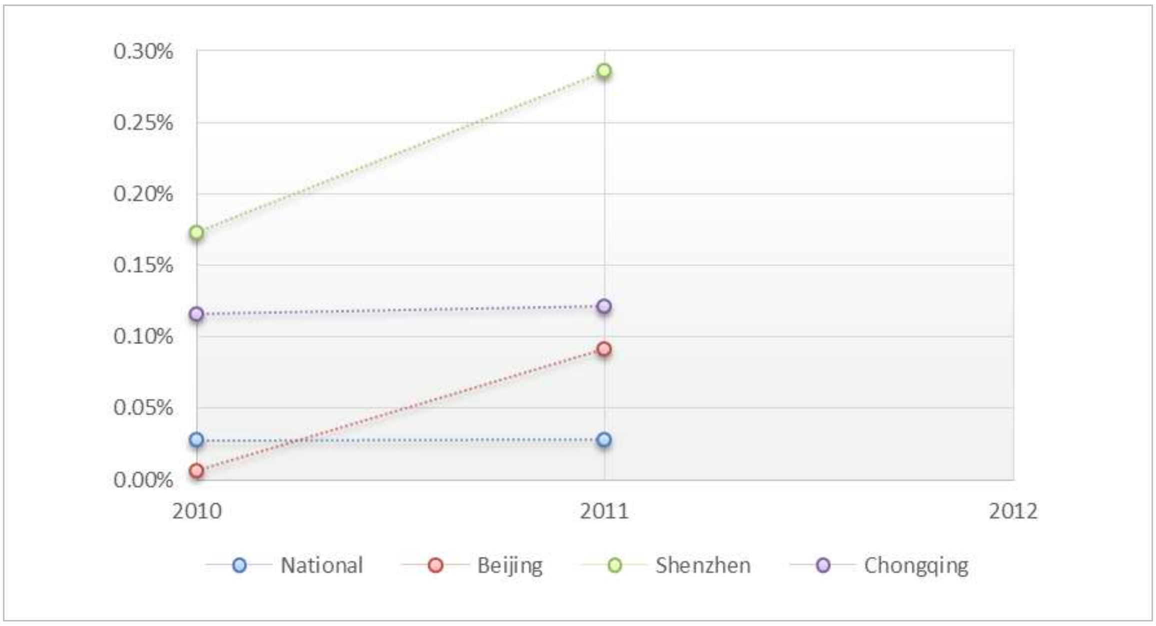

Figure 80: Annual light duty vehicle sales ratio of new energy vehicles in major cities and nationwide

The State Council announced the National Energy Saving and New Energy Vehicle Development Plan (2010-2020) in 2012, which aimed for a total sale of EV and PHEVs over 500,000 by 2015 and over 5 million, with a 2 million annual production capacity by 2020. In addition, the traditional vehicle fleet average fuel economy, excluding plug-in hybrid vehicles, should have fuel consumption controlled under 5.0 L/100 km for light duty passenger vehicles until 2020. With regard to the charging infrastructure sector, the State Council indicates it should be designed and developed simultaneously or even in advance to meet the electric vehicle increment goals.

The electric vehicle development in the 12th Five Year period has four major characteristics:

1. The electric vehicle demonstration industries should be extended, not only in bus, taxi and environmental sanitation vehicles, but expanded with regard to their logistics and personal use.
2. The demonstration fleet scale should be enlarged, which includes adding 5 or more cities to the plan and developing an intercity electric vehicle program combined with single city operation.
3. Among the total electric vehicle demonstration fleet, increase the passenger car and pure electric vehicles percentage in parallel.
4. Electric vehicle development should be connected with infrastructure construction and city planning having the combined effects to control energy resources and environment pollution with renewable energy, smart grid and smart traffic systems.

4.2. EV penetration scenarios

HEV, PHEV and BEV are widely discussed and considered as promising vehicle technologies to provide near-term and long-term energy savings and carbon and tailpipe emission reduction. Since these three vehicle technologies partly or exclusively rely on electricity (either produced from an internal combustion engine or charged from the grid), we called the commercialization of these three technologies as a process of vehicle electrification.

HEV can significantly improve fuel economy because the engine used in the HEV operates close to constant speed and is a highly efficient power source independent of road condition. Regenerative braking results in higher overall energy efficiency from the system. The battery or capacitor size in the HEV determines the power management strategy. The HEV is already a commercially available technology, best exhibited in the Toyota Prius. Another advantage for the HEV is that no additional charging infrastructure is needed; therefore, the HEV is usually considered more competitive than PHEV and BEV in the near-term.

The BEV uses only a battery and a motor to drive the vehicle, demanding a large energy storage capacity. BEV consumes electricity generated by the grid while maintaining high energy conversion efficiency during vehicle operation. The PHEV combines the characteristics of the HEV and BEV; it is capable of using only electricity when depleting its electric charge, and then operating like an HEV when the state of charge is low. Currently, battery technology is the bottle-neck for PHEV and BEV. Battery energy density, battery lifetime, safety and cost are the limiting factors. Another disadvantage is the extensive charging infrastructure network necessary. This is especially true for BEV since it exclusively relies on charging electricity.

In general, there are two views that estimate the future of these three technologies which bracket many of the growth predictions. One view is represented by the US Energy Information Agency (EIA). In reference to oil price scenarios, EIA's published Annual Energy Outlook (2009) projected HEV, PHEV and BEV together will account for 40% of total LDPV new sales in the U.S. by 2030, and could range from 38-45% depending on the fluctuation of oil price. However, such a market share is dominated by HEV. PHEV was assumed to have a small share of only 2% of total new sales, and for BEVs the share is negligible. However, a more optimistic opinion for PHEVs is held by several other institutes, such as the Electric Power Research Institute (EPRI) and Rocky Mountain Institute (RMI), etc. They assume that by 2020 PHEVs could reach 30% of total new LDPV sales in the U.S., and by 2030 such a market share could even climb to 50-70%.

The Chinese government is also actively pursuing the process of vehicle electrification. In 2009, the State Council released the Automotive Industry Restructuring and Revitalization Plan. The plan ambitiously concludes that HEV, PHEV and BEV together would account for 5% of the total passenger car sales by 2012. Further, as noted earlier, MIIT released a draft of "Energy-saving and New-energy Vehicle Development Plan (2011-2020)" in 2010. It expects that the stock for PHEV and BEV in China to reach 500 thousand by 2015, and the total stock for energy-saving and new energy vehicles to exceed 5 million by 2020.

Based on National plans mentioned above, we project the EVs' share till 2020. And based on Japan and US hybrid model developing conditions, we further forecast the HEV share till 2030. For BEV and PHEV, we mainly refer to the consumer acceptance results from hundreds of interviewees. The acceptance of BEV and PHEV varies as total usage cost change by year. We designed two scenarios for penetration of HEV, PHEV and BEV to the LDPV market in China from 2010 to 2030, as Figure 81 shows.

Scenario 1 emphasises on the development of PHEVs. We assumed HEV would reach 8% of total sales around 2020 and increase gradually its share to 18% by 2030. PHEV and BEV would contribute minimal shares, about 2.8% and 1.1% in 2020 and increase to 8.6% and 4.1% by 2030, respectively.

Scenario 2 emphasizes the development of BEVs. The BEV would reach 2.8% of total sales in 2020 and increase to 8.6% by 2030. PHEV was assumed to have a smaller share of only 1.1% of total new sales in 2020 and 4.1% by 2030.

Based on the two EV penetration scenarios, we evaluated the reduction potentials of energy use and pollutant emissions for the total LDPV fleet with introduction of HEV, PHEV and BEV in China and the three regions within the next two decades. The baseline is a scenario without development of new energy vehicles (called LDPV Growth).

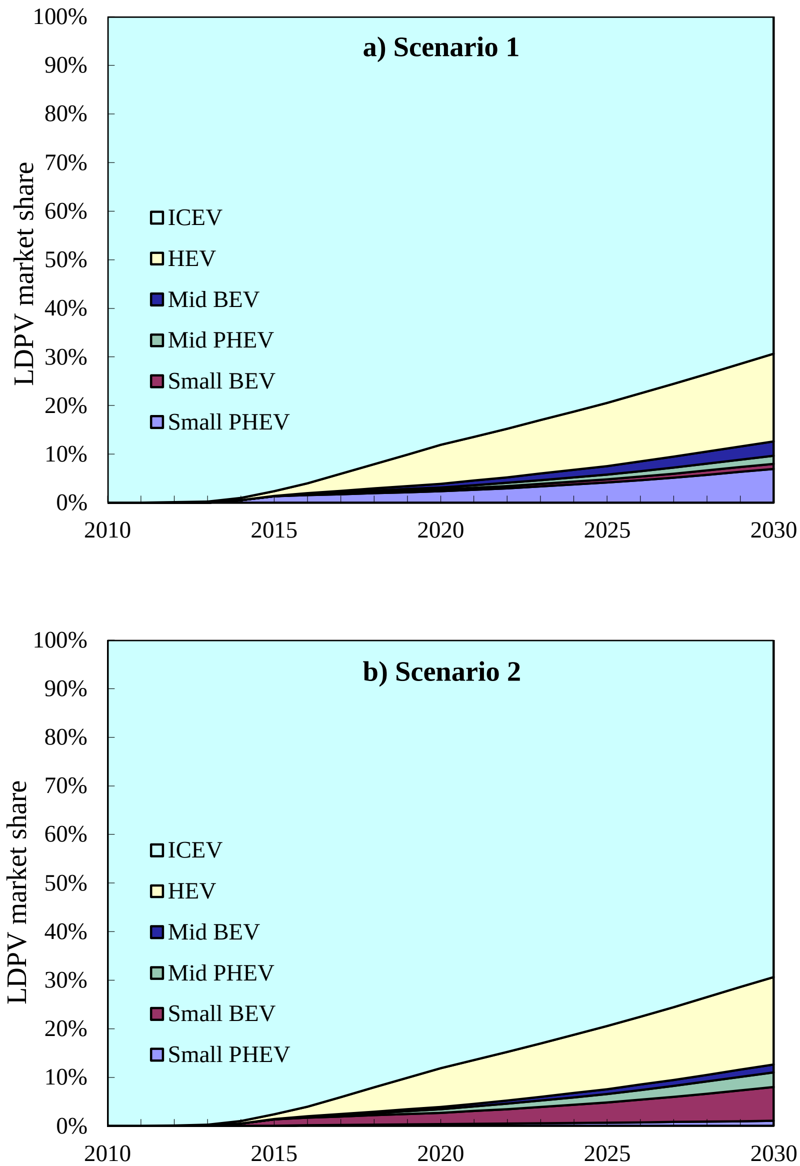

Figure 81: Share of ICEV, HEV, PHEV and BEV to total new LDPV sales, a) Scenario 1 and b) Scenario 2

4.2.1. The benefit of energy consumption and CO_2 emissions for the fleet

Figure 82 presents the trends in WTW fossil fuel use (Standard Coal Equivalent [SCE] is a measurement of energy given as the mass of coal. 1 kWh is equivalent to 0.1229 kg SCE.) and CO_2 emissions of the LDPV fleet in China. The WTW fossil fuel use for the LDPV fleet in China first will continue to increase rapidly in this decade (2010-2020), and then slow significantly and reach peak point around 2023. Promotion of HEV, PHEV and BEV could help cut WTW fossil fuel use. The reduction rate in WTW fossil energy use relative to the LDPV growth scenario is 7% (24 million tons of coal equivalent), 8% (27 million tons) in 2030, respectively, for HEV/PHEV/BEV penetration scenarios 1 and 2.

Similar to fossil energy use, the WTW CO_2 emissions for the LDPV fleet in China continue to increase rapidly this decade and then slow down significantly and reach the peak in CO_2 emissions about 2023. Such a pattern is consistent with that of gasoline demand and fossil energy use. Without penetration of alternative fuels and advanced vehicle technologies, by 2030 the WTW CO_2 emissions under the LDPV growth scenario will reach 794 million tons.

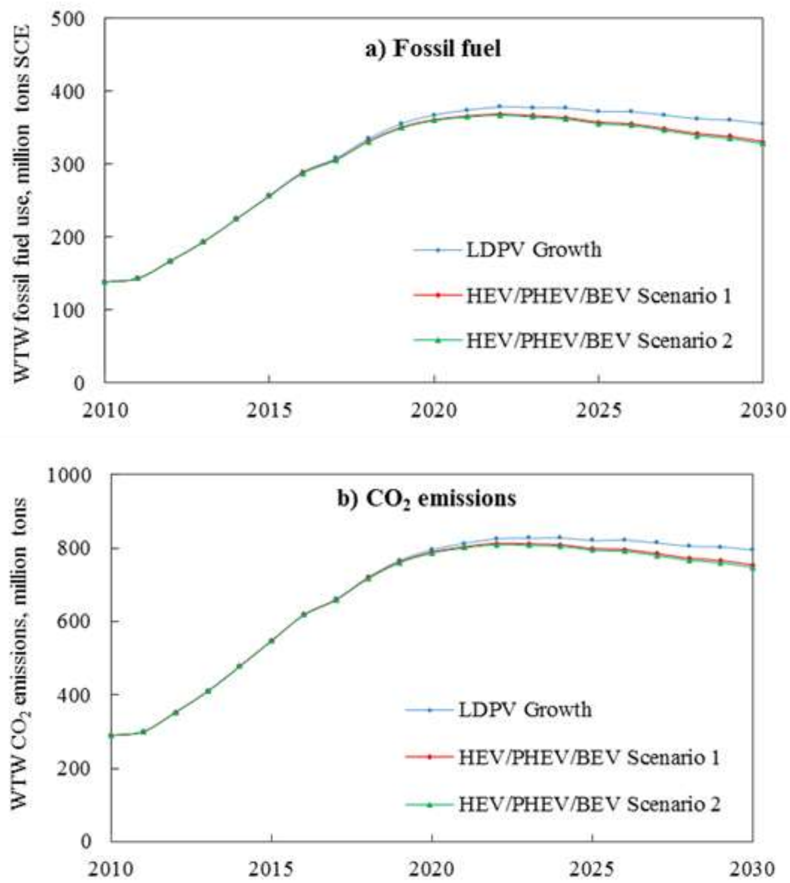

Figure 82: Trends of a) Fossil fuel and b) CO2 emissions from penetration of HEV/PHEV/BEV in China

Promotion of HEV, PHEV and BEV will also help cut WTW CO_2 emissions. However, we clearly observed a further narrower reduction benefit in WTW CO_2 emissions compared to that of WTW fossil energy use. The reduction rate in WTW CO_2 emissions relative to the LDPV growth scenario is 5% (40 million tons), 6% (47 million tons) in 2030, respectively, for HEV/PHEV/BEV penetration scenarios 1 and 2. Promotion of PHEV and BEV into the LDPV fleet in China can only achieve slight reduction in WTW CO_2 emissions. The reason is seen by the two separate WTT and TTW charts in Figure 83. Although penetration of PHEV and BEV can significantly reduce fleet TTW CO_2 emissions as much as 10% (62 million tons) by 2030, using the scenario 1 as an example, WTT CO_2 emissions for the LDPV fleet increase greatly by 14% (21 million tons) using the conservative generation mix. In this generation mix coal power still dominates, at 91% of total in 2030.

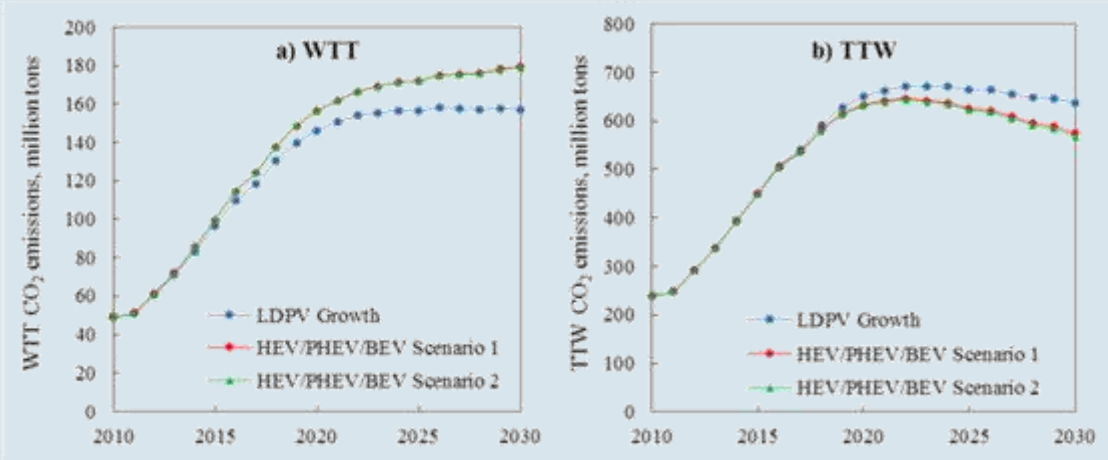

Figure 83: WTT and TTW CO2 emissions of LDPV fleet with various scenarios in China, 2010-2030

4.2.2. The benefit of pollutant emissions for fleet

The trends of pollutant emissions from LDPVs in China from 2010 to 2030 are presented in Figure 84. The trends are quite different from the trends of energy consumption of CO_2 emissions. The emissions of VOC and CO present similar trends. Under the LDPV growth scenario, for the first decade (2010-2020), emissions will decrease at an annual average rate of 1.6% and 4.6%, respectively. For the second decade (2020-2030), the decreasing trend of CO will slow significantly while the emissions of VOC will begin to increase gradually. For the emissions of NO_X, it is much harder to make the trend become a decline; the slowly decreasing trend will start in the second decade (2020-2030). For the $PM_{2.5}$, emissions first will continue to increase rapidly in this decade (2010-2020), and then slow significantly and reach peak point around 2023.

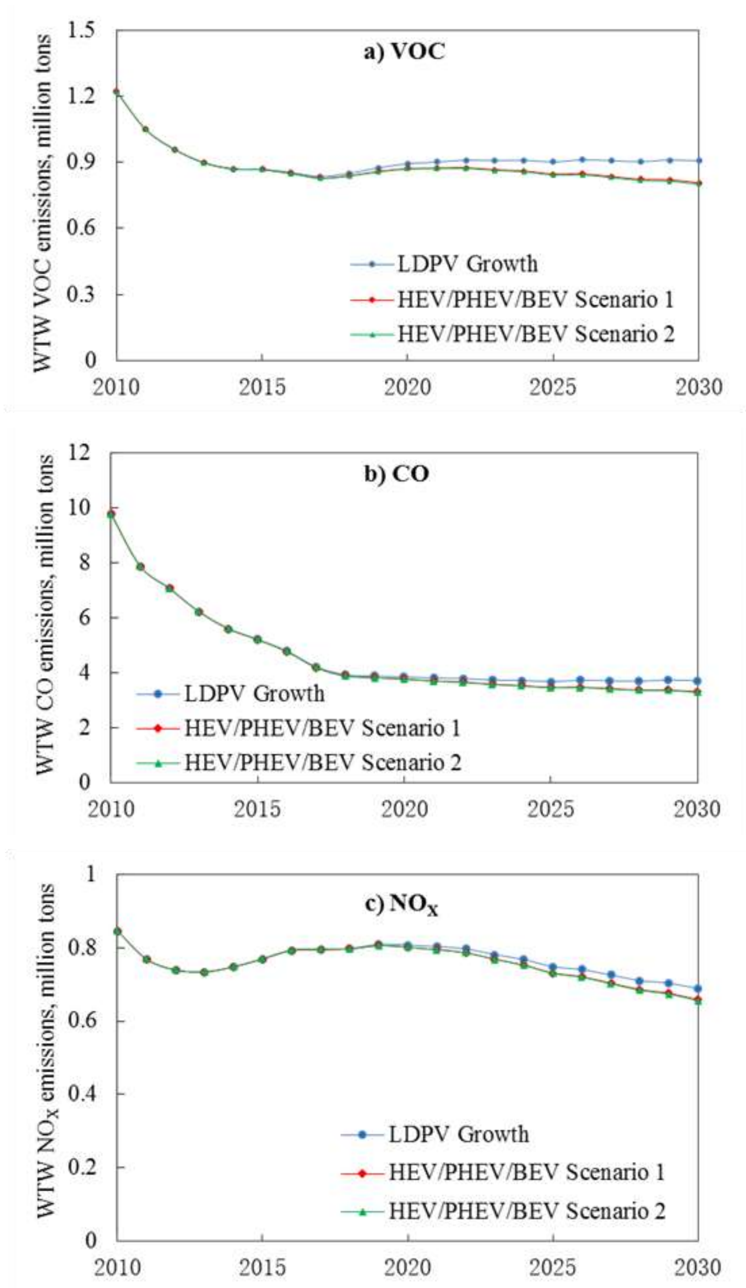

Figure 84: Trends of pollutant emissions for fleet in China: a) VOC, b) CO, c) NOX, d) PM2.5, 2010-2030

Promotion of HEV, PHEV and BEV will also help cut pollutant emissions. However, we clearly observed a further narrower reduction benefit in WTW CO_2 emissions compared to that of WTW fossil energy use. The reduction rate in VOC, CO, NO_X and $PM_{2.5}$ emissions relative to the LDPV growth scenario is 12% (0.1 million tons), 11% (0.4 million tons), 4% (33 thousand tons) and 2% (1.2 thousand tons) in 2030, respectively, for HEV/PHEV/BEV penetration scenarios 1.

4.3. The benefit of energy saving and emissions reduction for fleet in three regions

4.3.1. The benefit of energy consumption and CO_2 emissions for fleet

Figure 85 presents the trends in WTW fossil fuel use of LDPV fleet under three scenarios in three regions from 2010 to 2030. The fossil fuel consumption for the LDPV fleet in YRD and PRD shows a similar trend, while the trend is different in Beijing. In Beijing, fossil energy consumption decreases do to limits placed on private car registration launched in 2011. The WTW fossil fuel use decreases to 4.9 million tons SCE in 2030. In YRD and PRD, for the first decade (2010-2020), the WTW fossil fuel use will continue to increase rapidly in this decade (2010-2020), and then slow significantly and reach peak WTW fossil energy consumption around 2025 in YRD and PRD.

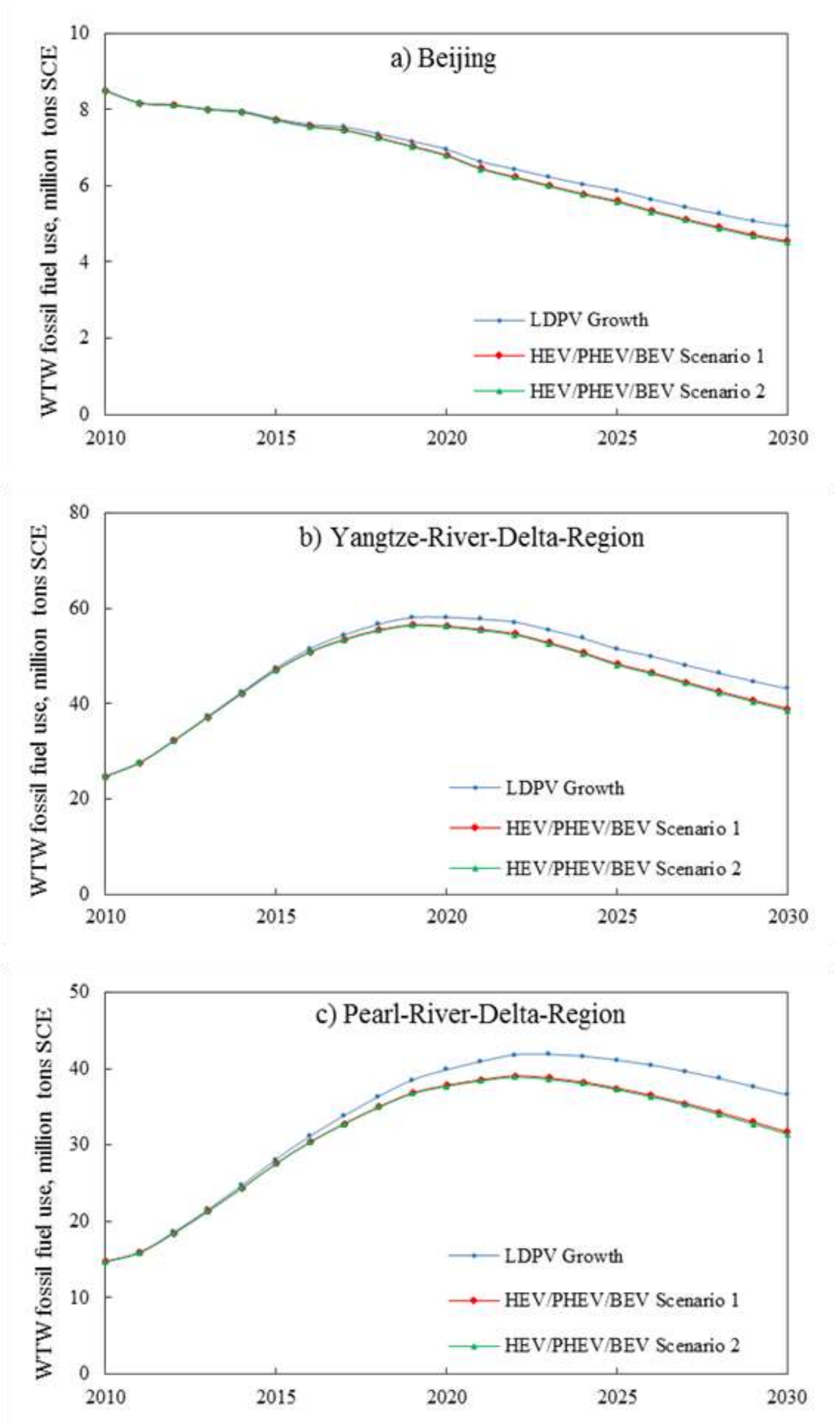

Figure 85: Trends of fossil fuel use from penetration of HEV/PHEV/BEV in a) Beijing, b) Yangtze-River-Delta-Region, c) Pearl-River-Delta-Region, 2010-2030

Promotion of HEV, PHEV and BEV could help further cut WTW fossil energy use. For Beijing, the fossil energy use under HEV/PHEV/BEV scenario 1 and 2 decreases by 8% and 9% in 2030, respectively. For YRD and PRD, the fossil energy use under scenario 1 will reduce 4 and 5 million tons SCE in 2030, and the reduction rate is 10% and 14%, respectively (relative to LDPV growth scenario).

Similar to fossil energy use, the WTW CO_2 emissions for the LDPV fleet in all three regions have the same trend with fossil energy use, as shown in Figure 86. Without penetration of alternative fuels and advanced vehicle technologies, by 2030 the WTW CO_2 emissions under the LDPV growth scenario will reach 11, 95 and 80 million tons, respectively, for these three regions.

Promotion of HEV, PHEV and BEV could help further cut WTW CO_2 emissions in all three regions. Similarly, we clearly observed a further narrower reduction benefit in WTW CO_2 emissions compared to that of WTW fossil energy use. For Beijing, WTW CO_2 emissions under HEV/PHEV/BEV scenario 1 and 2 will decrease by 6% and 7% in 2030, respectively. Promotion of PHEV and BEV in YRD and PRD could achieve more CO_2 reduction benefit due to the much cleaner electricity generation mix in these two regions. Under HEV/PHEV/BEV scenario 2 the reduction rate in WTW CO_2 emissions relative to LDPV growth scenario is 8% and 12% in YRD and PRD, respectively. Clearly, the effort to mitigate fleet WTW CO_2 emissions is tougher than either that for fossil energy use or oil consumption. More control strategies (such as CCS for coal power and more renewable fuels) need to be combined for a more successful vehicle electrification in China in the future.

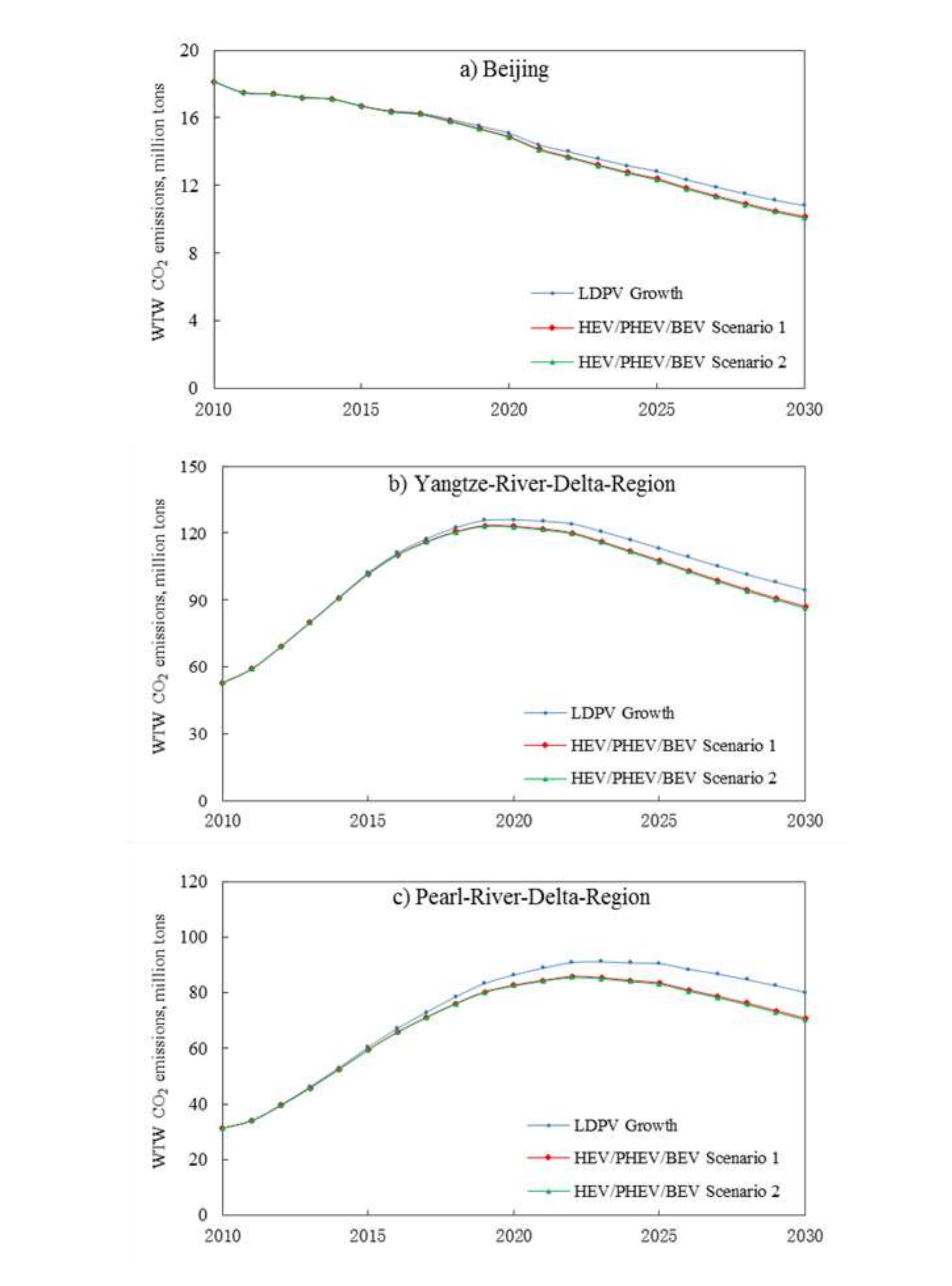

Figure 86: Trends of CO2 emissions from penetration of HEV/PHEV/BEV in a) Beijing, b) Yangtze-River-Delta-Region, c) Pearl-River-Delta-Region, 2010-2030

4.3.2. The benefit of pollutants emissions for fleet

Figure 87, Figure 88, Figure 89 illustrate the trends of pollutant emissions of LDPVs in Beijing, YRD and PRD. Thanks to a series of stringent vehicle emission controls adopted in Beijing (e.g., Euro V and Euro VI emission standards, restrictions on LDPV use and purchase, etc.), the estimated vehicle emissions in Beijing continue to decrease in the next decades. The WTW VOC, CO, NO_X and $PM_{2.5}$ emissions decrease to 4.9 million tons SCE in 2030.

For YRD and PRD, the emissions of VOC, NO_X and $PM_{2.5}$ will continue to increase rapidly in this decade (2010-2020), and then slow significantly and reach peak WTW fossil energy consumption around 2025 while the emissions of CO will keep decreasing through 2030.

Promotion of HEV, PHEV and BEV could also help further cut WTW pollutant emissions in all three regions. Similarly, we clearly observed a further narrower reduction benefit in Beijing compared to that in YRD and PRD. For Beijing, WTW VOC, CO, NO_X and $PM_{2.5}$ emissions under HEV/PHEV/BEV scenario 1 decrease by 11%, 10%, 6% and 3% in 2030, respectively. Promotion of PHEV and BEV in YRD and PRD could achieve more reduction benefit due to the much cleaner electricity generation mix in these two regions. Under HEV/PHEV/BEV scenario 1 the reduction rate in VOC, CO, NO_X and $PM_{2.5}$ emissions relative to LDPV growth scenario is 12%, 11%, 7% and 4% in YRD, 12%, 11%, 10% and 6% in PRD, respectively.

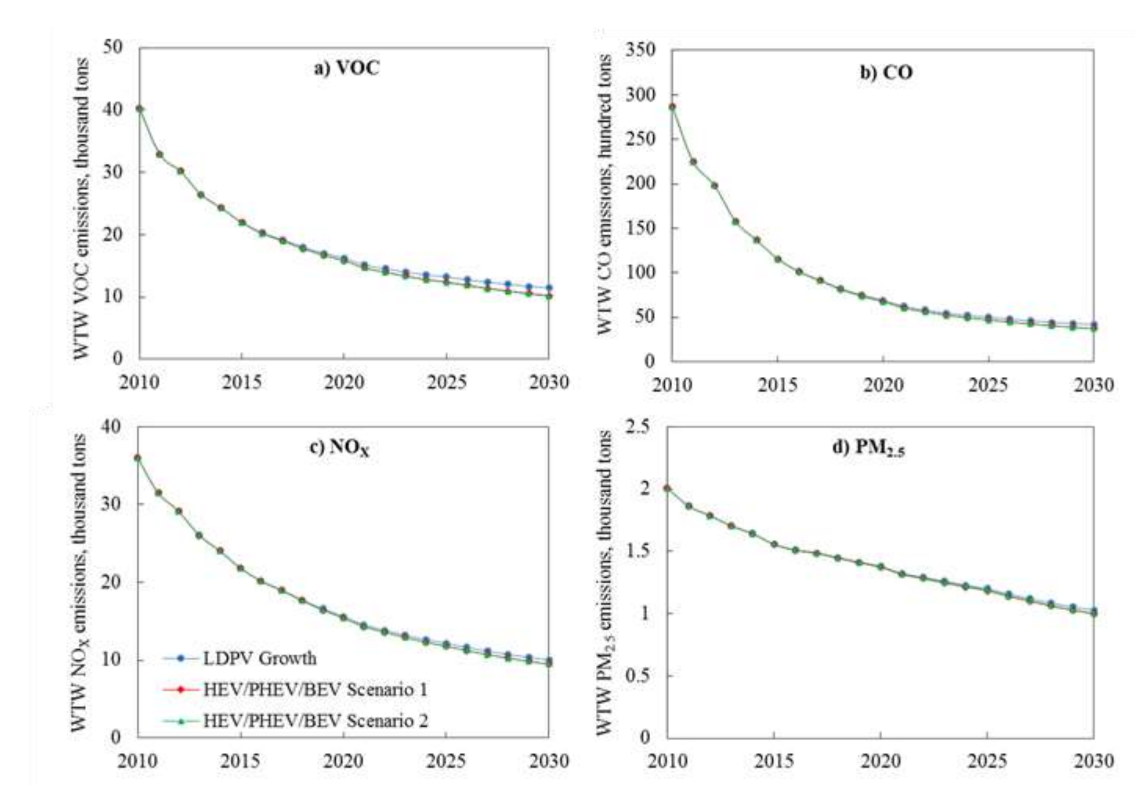

Figure 87: Trends of pollutant emissions from penetration of HEV/PHEV/BEV in Beijing

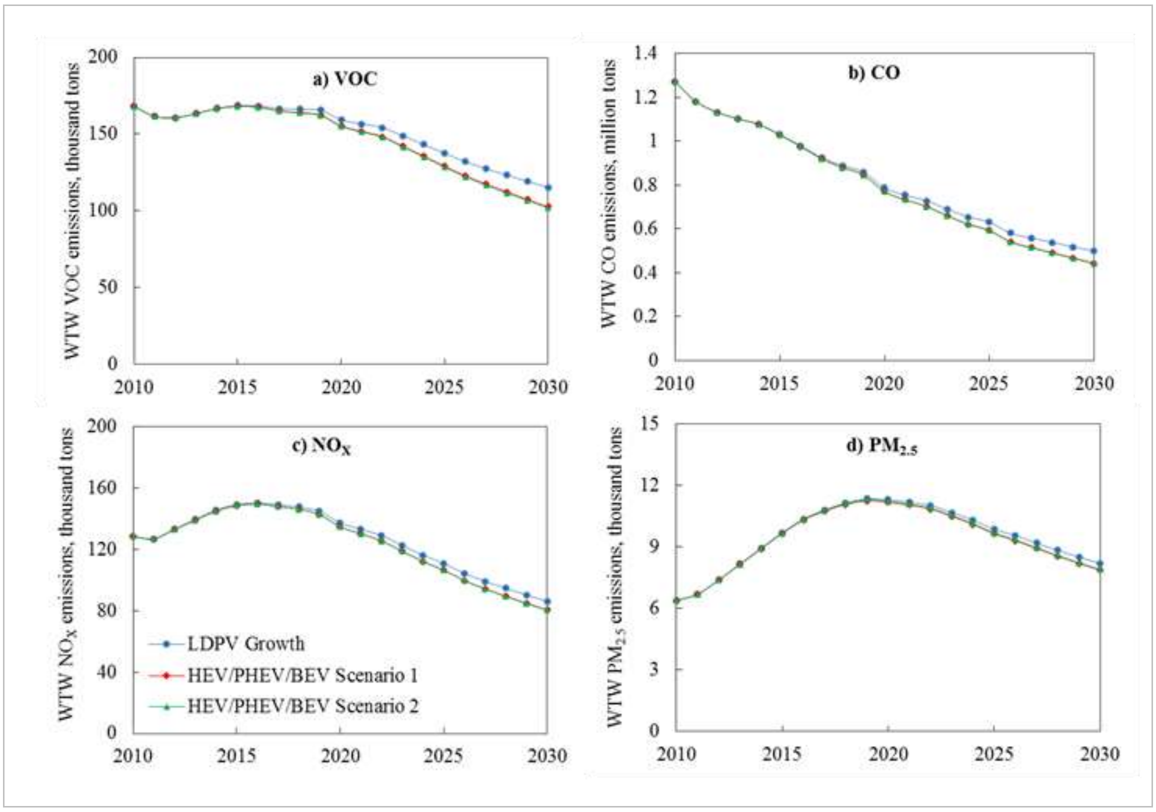

Figure 88: Trends of pollutant emissions from penetration of HEV/PHEV/BEV in YRD

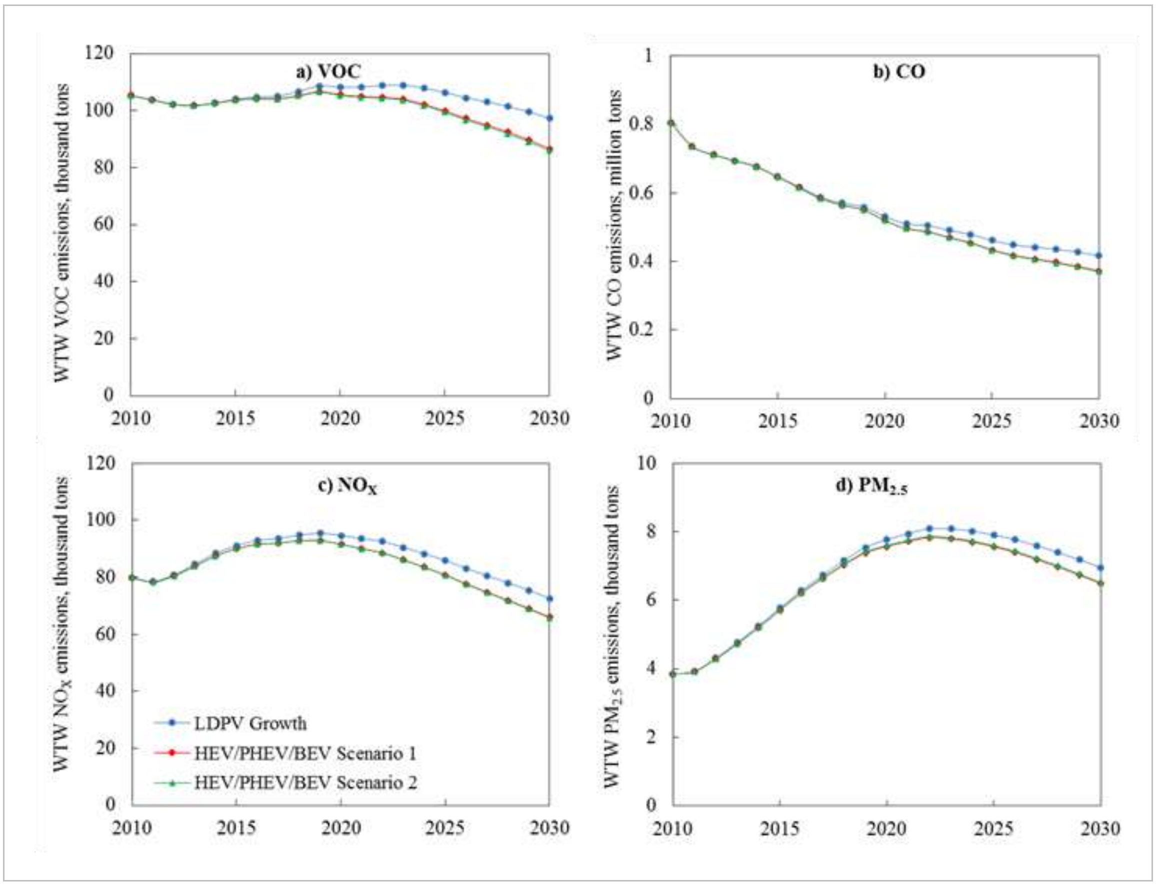

Figure 89: Trends of pollutant emissions from penetration of HEV/PHEV/BEV in PRD

5. Conclusions

5.1. Oil security

Vehicle electrification can greatly ease the oil energy crisis.

From the perspective of fuel-cycle analysis, the petroleum consumption on the upstream stage is much lower than that on the downstream stage, due to a slight oil-based electricity share in China's average generation mix and high refining efficiency of conventional oil refinery industry. Petroleum consumption in the vehicle operation stage can account for about 90% of total life cycle petroleum consumption. However, all of the three kinds of electric driving vehicles of interest--HEV, PHEV15, and PHEV50 can significantly reduce WTW petroleum consumption by 30%, 40%, and 50%, compared to the conventional ICEV, respectively. BEV can almost eliminate petroleum consumption because its driving force comes from electricity, while the oil-based electricity share in China's average generation mix is negligible.

5.2. Fossil energy and CO_2 mitigation

The reduction of fossil fuel consumption and CO_2 emissions of electric vehicles depend on a series of complex factors such as the generation mix, coal-based generating technologies, vehicle fuel economy, and manufacture of vehicle materials.

For all kinds of vehicles of interest, the WTW fossil fuel consumption will decrease sharply in the next two decades with the development of vehicle technologies, generating efficiency, and manufacturing technologies. HEV, PHEV50, and BEV can save fossil energy by 20%, 11%, and 13%, compared to ICEV, respectively. In the primary stage of vehicle electrification, the reduction effects of BEV on fossil fuel consumption in vehicle operation stage can be offset by high energy consumption in the manufacturing processes of key vehicle components like the battery. As the continuous growth of electric vehicle market and the technology development of manufacturing industry, HEV, PHEV50 and BEV can achieve larger reduction effects by 21%, 19%, and 40%, respectively, compared to ICEV in 2030.

The energy consumption of the fuel-cycle accounts for more than 80% of full life-cycle. Although the contribution of the vehicle-cycle is relatively low, it still has a remarkable impact on the integrated fuel consumption. For instance, the breakeven point of fossil fuel consumption of BEV is beyond 250 thousand kilometres, compared to HEV in JJJ with a high coal electricity share, which means the BEV has no advantages on fossil fuel consumption compared to HEV within the life-time driving distance of 200 thousand kilometres. If we only concern about the fuel-cycle consumption, driving a BEV in JJJ will consume a similar quantity of fossil fuel compared to HEV. But for some regions with a high clean energy share, the breakeven point of fossil fuel consumption of BEV, compared to HEV will show up earlier, within the life-time driving distance of 200 thousand kilometres.

The full life-cycle emissions of CO_2 turn out to be a different situation from fossil fuel consumption. In the baseline year--2010, HEV, PHEV15, PHEV50 and BEV reduce fuel-cycle CO_2 emissions by 30%, 30%, 10% and 10%, respectively, compared to ICEV. Similar to the fossil fuel consumption, the reduction effects of BEV on CO_2 emissions in the vehicle operation stage can be offset by a dominant coal-based electricity share and high energy consumption in the manufacturing processes of key vehicle components like battery. From the perspective of full life-cycle analysis, BEV will increase CO_2 emissions by 8% more than ICEV on the national average level in 2010, while HEV will reduce CO_2 by 19%, compared to ICEV. The reduction effects of vehicle electrification will occur after 2020, while HEV, PHEV50 and BEV can reduce CO_2 emissions by 20%, 13% and 27%, respectively, compared to ICEV in 2030.

This study simulated when and which batteries modules are relatively environmental friendly from a life cycle perspective, including lead acid (Pb-Ac), nickel metal hydride (Ni-MH) and four lithium ion (Li-ion) battery types. Technological progress is needed to lower the life cycle analysis in China 2030. In 2010 (the reference year), Li-ion batteries need 210-240 MJ life cycle energy to produce 1kg module and emit 20-23 kg CO_2 equivalent simultaneously, compared to 114MJ/kg, 11kg CO_2 for Ni-MH and 42MJ/kg, 3.7kg CO_2 for Pb-Ac. The energy and emission data were collected from material production (including recycling use) and battery production whose detailed industrial processes were considered. The battery module contributes 52% of total vehicle cycle energy use of a middle size passenger car. Comparing the results of China and the Western world, batteries in China are more energy intensive and in the operation stage more miles must be accumulated to achieve the life cycle benefit of electrified vehicles. Although $LiNi_{1/3}Co_{1/3}Mn_{1/3}O_2$ batteries did not have the lowest weight specific vehicle cycle energy use, the per module energy use for the same power demand is favourable because of high energy reserve capacity. According to the scenarios in 2030, Li-ion batteries would cut 72-74% of energy use to produce one kg of battery. Until then the per module vehicle cycle results of li-ion battery is more competitive than that in the reference year.

5.3. Air pollutant emissions mitigation

Penetration of electric vehicles can greatly reduce or even eliminate the air pollutant emissions in the vehicle operation stage, which has significant impacts on the improvement of urban air quality. However, in terms of a fuel-cycle perspective, the mitigation of emissions is dependent on the particular air pollutant. . For light-duty vehicles, the penetration of electric vehicles has a great advantage in reducing VOC and CO emissions, but may increase NO_X, $PM_{2.5}$ and SO_2 emissions significantly. To achieve life-cycle benefit for all major air pollutants, cleaner and renewable electricity needs to be heavily promoted in the future.

WTW VOC and CO emissions mainly come from the vehicle operation stage. All of the HEV, PHEV, and BEV can significantly reduce VOC and CO emissions, and the regional variation of VOC and CO from these vehicles in China is moderate. For VOC emissions, HEV, PHEV and BEV can reduce this pollutant by ~30%, ~50% and ~90%, respectively, compared to the ICEV; while for CO emissions, the reduction ratio of HEV, PHEV and BEV is ~40%, ~60% and ~95%, respectively, compared to ICEV. The VOC and CO emissions will decrease gradually with the development of vehicle emission standards over time.

$PM_{2.5}$ and SO_2 emissions mainly are generated in the upstream stages such as feedstock exploitation, transmission and distribution, and manufacturing process. HEV has considerably lower $PM_{2.5}$ and SO_2 emissions, while PHEV and BEV will remarkably increase the primary $PM_{2.5}$ and SO_2 emissions. The $PM_{2.5}$ and SO_2 emissions will gradually decrease as the development of generation efficiency and the clean coal-based electricity share increase over time. To significantly reduce the full life-cycle $PM_{2.5}$ and SO_2 emissions, we need more clean energy generation and penetration of dust removal and desulfurization technology in power plants when promoting PHEV and BEV in China.

Similar to $PM_{2.5}$ and SO_2 emissions, NO_X emissions mainly come from the upstream stages. The regional variation of NO_X emissions in China is remarkable, due to the variation of generation mix throughout the country. For instance, the regions with high coal-based electricity share such as Northeast China and the Jing-Jin-Ji region have higher NO_X emissions than the regions with a high clean energy electricity share such as Central China and Pearl River Delta region. HEV can reduce NO_X emissions about 30% compared to ICEV, while the NO_X emissions of PHEV and ICEV are comparable. But the NO_X emissions of BEV are higher by 40% to 100% among those regions than NO_X from ICEV. After 2020, in those regions with high clean energy electricity share such as Central China and the Pearl River Delta region, BEV could lower NO_X emissions compared to ICEV.

6. Discussions and Outlook

The main results and conclusions in respect to energy consumption and the environmental impact of electric vehicle promotion in China based on different vehicle technologies were presented in the above chapters. In this chapter, relevant policies and impeding factors for the climate friendly development of electro-mobility will be discussed.

6.1. Generation mix

China's Twelfth Five Year Plan on Energy Development released in early 2013 put forward ambitious goals to actively develop renewable energy sources such as hydrogen, solar and wind power. According to the plan, the share of non-fossil fuels will be raised to 11.4% by the end of 2015, the share of natural gas increased to 7.5%, while that of coal will be reduced to 65%. Just a few days before this report was finished, the Energy Development Strategy Action Plan (2014-2020) was released by the China's State Council, promising more efficient, self-sufficient, green and innovative energy production and consumption by 2020. According to the plan, the share of non-fossil fuels in the total primary energy consumption will be up to 15% by 2020, from 9.8% in 2013. In addition, the share of natural gas will be raised to above 10%, while that of coal will be reduced to below 62%. To achieve this goal, production of both shale gas and coal bed methane are supposed to reach 30 billion cubic meters by 2020. Furthermore, China officially announced the emission peak of greenhouse gases would occur about 2030 for the first time, and promised the share of non-fossil fuels in the total primary energy consumption will rise to 20% by 2030.

Already today China is the single largest investor in renewable energies. China is undoubtedly going to enforce its strength on promoting renewable and clean energy generation in future, bearing in mind the barely satisfactory achievements in the past decade. The reasons are evident—limited domestic fossil fuel reserves, serious environment pollution, and increasing pressure in respect to the fulfilment of greenhouse gas reduction target subject to international agreements.

Another sensitive issue related to China's future energy structure is the renewed investment in nuclear power plants in eastern coastal areas and the planning of inland nuclear power plants. After the Fukushima disaster many countries slowed down, and even suspended their construction of nuclear power plants, including China. However, *China's Security Plan of Nuclear Power and Development* indicates to steadily recover the construction of coastal nuclear power plants in the future. Beyond all safety aspects, this will have a considerable impact on the climate impact of energy production in China and consequently be an important determinant for the life-cycle emissions of electric vehicles.

In this study, we projected that coal-based generation would account for 71% and 60% of the total generation mix by 2030 in the conservative and ambitious scenario, respectively. In the baseline year 2010, the share of coal in the total energy consumption was 71.9%, while the coal-based generation share in the total generation mix was about 79%. According to our projection, the share of coal in the total energy consumption in 2030 will be 50%~60%, which is roughly in accordance with the 2020's target of 62% in the Energy Development Strategy Action Plan (2014-2020). In general, the projection for the average generation mix on the national level in China will not change considerably. However, the regional generation mixes are becoming more and more uncertain due to different development prospects of various renewable energy sources in China. While the methodology developed through this study is adequate, it should frequently be revised with current government policies and progress in respect to planned energy transition.

6.2. Integration of renewable energy into electric vehicle charging systems

As mentioned before, a high share of low-carbon renewable energy sources is key to achieve the undoubtedly high emission reduction potential of electric vehicles from the perspective of the life-cycle. However, the expansion of low-carbon energy production is not just about investing into renewable energy power plants and infrastructure. It is also about making clean energy available for electric-vehicles. To do so, renewable energies have to be integrated into the charging network. In this respect, several complications must be taken into consideration.

Firstly, the spatial coupling of power generation and vehicle charging events need to be considered. Cities are the driver behind the promotion of electric vehicles. Short distances, many charging opportunities and generally higher average income of city residents in China are driver for electro-mobility in high-density urban areas. Beyond that, city administrations are providing substantial subsidies for electric-vehicles in China. However, renewable energy sources are typically not located in cities. Almost all large-scale wind power generation fields and solar power generation fields are located at the outskirts and even hundreds or thousands of miles away from cities. A vast and open access power network is required to make use of the power from remote renewable energy sources.

Secondly, temporal coupling of power generation and vehicle charging is to be considered, especially for energy sources with endogenous temporal changeability such as wind, solar and tidal power. In the absence of smart grid technologies and vehicle charging management, adequate energy storage capacity has to be developed to store and temporally distribute surplus power that is generated during the off-peak.

Chinese state and local governments have provided a number of purchasing incentives to narrow the gap of the original retail prices between electric vehicles and their conventional counterparts. However, those incentives are typically targeting the purchase and not the operational costs. The Chinese government still has to create incentives for private vehicle owners to charge electric vehicles with renewable energy power at a competitive price. Given the infrastructure requirements in the current renewable energy development stage, this might require subsidies for renewable power suppliers or a disincentive for coal-based energy production (e.g. through taxation).

6.3. User acceptance

Research works such as this LCA focus on the energy and environmental impact of electric vehicles, while users care about costs, usability, performance, appearance, and even trend. With monetary purchasing subsidies, super credits, tax exemptions and local incentives for industry and consumers the list of support programmes from the Chinese government is long. However, to establish a complete and mature market for electric vehicles in China, user acceptance must be further considered.

The designed electric driving range is considered to affect the user acceptance for a specific electric vehicle to a great extent. But unlike the conventional gasoline and diesel vehicles, driving range of electric vehicle is limited due to relatively low energy density of battery, compared to liquid fuels. Drivers of conventional vehicles do not have to be concerned about running out of fuels because the driving ranges of their vehicles are commonly hundreds of miles while the fuelling station network is extensive. Therefore, not only the limited driving range, but also the lack of charging facilities of electric vehicles has adverse effect on the user acceptance. In addition, car buyers would also concern about the lifetime costs of driving an electric vehicle with comparison to that of conventional vehicles. As mentioned before, governments have made great efforts on cutting down the retail price of electric vehicles to a competitive level for conventional vehicles by providing various preferential and subsidy policies. It is now up to the vehicle producers to ensure that driving ranges are competitive.

6.4. Limitations and unsolved issues of this study

Marginal generation mix versus average generation mix

In this study, we employed the average generation mix on both regional and national level to evaluate the energy and environmental impact of electric vehicles. It might be questioned that electric vehicles are extra electricity carriers over the average power load when they are charged on the terminals of power grid. Therefore, to be exact, the marginal generation mix should be used to evaluate the energy and environmental impact of electric vehicles, rather than average generation mix.

To obtain the marginal generation mix of charging electric vehicles, we need complete data of all power plants involved and detailed information of the response and dispatch mechanism of the power grid. But they are very difficult to access in China and also too much beyond the framework of this study. In addition, the involvement of more renewable power generation such as wind power and solar power will complicate the power dispatch in the future, because they can't serve the base load with their great endogenous variability. Therefore, it is not realistic at this stage to develop an electricity dispatch model to highly characterize the marginal generation for charging electric vehicles. From the macro perspective, China's annual incremental generation mix is close to the average generation mix because coal-fired power is still dominant and the share of renewable energy is still low due to the expected considerable development target of renewable energy can't be achieved in a short time without a long-period construction of energy market and supporting facilities. In conclusion, we adopted the average generation mix to evaluate the life-cycle energy and environmental impact of those new energy vehicles owing to the unavailability of electricity dispatch model and the current situation of Chinese power structure.

Full life-cycle analysis on vehicle and fuel systems

In this study, we conducted the localised fuel-cycle analysis for various vehicle technologies and material cycle of different batteries of electric vehicles, which is obviously not a proper full life-cycle for various vehicle and fuel systems. However, energy consumption and pollutant emission data of production process of key vehicle components and vehicle assembling process are very difficult to obtain in China. There are two ways to get these very sensitive data named by the manufacturers. One is through joint research projects with vehicle manufacturers; and the other one is the officially published data from those authorities. We still keep trying hard to get adequate high-quality energy consumption, and air pollutant emission data, even cost information of vehicle manufacturing and disposal phases through all the ways we have, in order to conduct a complete full life-cycle analysis on energy consumption and pollutant emissions of different vehicle and fuel systems in China.

Air quality impact

The major results of this study are the life-cycle energy consumption, CO2 and air pollutant emissions of different new energy vehicle technologies in China. To a certain extent, air pollutant emissions are preliminary results for air quality impact analysis. Further study will focus on the air quality simulation on Chinese regional and city level to evaluate the air quality impact of promoting large-scale electric vehicles in China.

Influences of electric vehicle promotion on life-cycle air pollutant emissions mainly focus on the emissions from power plants on the upstream energy sector and from transportation sector in the downstream vehicle operation stage. Apparently, vehicle electrification will shift the emissions from transportation sector to power sector, especially for air pollutants highly related to fossil fuel combustion, such as NO_X, particle matters, and SO_X. Therefore, promoting electric vehicles will significantly reduce the pollutant emissions from transportation sector in urban areas, compared to conventional gasoline and diesel vehicles. However, in the meantime, it will generate extra emissions from upstream energy sector, specifically from power plants which commonly locate at the outskirts. In addition, emission reduction effects on transportation in urban areas mostly occur in the daytime, while the emission increases from energy supply sector often occur at night since electric vehicles are expected to be charged at night, considering more accessibility of charging facilities at home and relatively lower electricity price during this period over a day. Therefore, the emissions will change not only on space, but also on time, which makes it complicated to address the air quality impact of promoting large-scale electric vehicles in China.

The next phase of our study is to employ air quality model to simulate the change of air pollutant concentrations caused by emission changes due to promotion of large-scale electric vehicles in China, which is believed to address the environmental impact of electric vehicle more properly and exactly.

7. Conclusion

The LCA results show: Promoting electro-mobility is not a low-hanging fruit in the battle against climate change. The long-term mitigation potential of EVs could come at significant abatement and environmental costs today. Nonetheless the LCA also shows that a low-carbon automotive sector is not attainable without EVs. A low-carbon transformation of the road transport sector is only attainable if EVs are established in the market, if transport demand management is effectively leading to a modal shift and if the share of renewable energy increases substantially.

What are the implications of these results for policy making in China? A number of concrete conclusions were presented in chapter 6. However, a comprehensive systemic climate and environmental approach is required beyond the promotion of e-mobility. Now is the time to develop innovative intermodal mobility solutions, improve battery recycling, transform the energy sector and install dust removal and desulphurisation technology. If these preconditions are met, electro-mobility in China will activate its huge climate and environmental protection potential.

The question remains: Are the substantial investments in e-mobility justified from an environmental point of view? There does not seem to be much of an alternative currently. Without any doubt the demand for motorised individual mobility in China will not slow down significantly any time soon. The technological lock-in effect of not investing e-mobility now - to enable it to develop at the same pace as the renewable energies - could therefore be very costly in the long run. Long development cycles in the automotive industry, long innovation cycles for traction batteries and a costly charging infrastructure require early collaboration across industry and the State. If government, car manufacturers, and other stakeholders, such as charge point operators and utilities, fail to drive the market expansion now, the passenger road transport sector is likely not ready for the necessary decarbonisation when the power sector is.

List of tables

Table 1 Six regional grid systems and the provinces included ... 6

Table 2: Energy efficiency, shares of process fuels and transportation modes for two pathways ... 8

Table 3: Projection of national average generation mix in China, 2030 ... 12

Table 4: Removal efficiency and penetration rate of emission control technologies of power plants in China, 2010-2030 ... 16

Table 5: Chinese fuel consumption standards for new passenger cars ... 17

Table 6: Energy saving effect and penetration target of energy saving technologies of passenger car 19

Table 7: Projection of fuel economy of light duty passenger vehicles in China ... 20

Table 8: Component compositions of each part of vehicle ... 25

Table 9: Tier 3 standard limits for vehicle emission factors under FTP test ... 61

Table 10: Top 10 regional wind curtailment areas in China, 2011 ... 69

Table 11: Penetration scenarios for electric vehicles in Beijing ... 70

Table 12: Charging patterns for electric vehicles ... 70

Table 13: Results of charging requirement for different scenarios ... 71

List of Figures

Figure 1: Logistics of methodology fundamentals, tools and major data inputs/outputs ... 4

Figure 2: Wind generation capacity distribution of China in 2011 ... 5

Figure 3: Gasoline pathway ... 8

Figure 4: Electricity pathway ... 8

Figure 5: Electricity generation development of China, 1980-2010 ... 10

Figure 6: Electricity generation mix at national and regional levels in China, 2010 ... 11

Figure 7: Generation mix in China, 2010-2030 ... 11

Figure 8: Generation mix in six regions, 2010-2030 ... 13

Figure 9: Emission factors of air pollutants from power plants in China, 2010-2030 ... 16

Figure 10: Sales-weighted fuel economy and market share of the top 60th sales of LDPV models in China ... 18

Figure 11: Comparisons of fuel economy between BEVs and the corresponding ICEVs, 2010 ... 20

Figure 12: Fuel economy projections of ICEV, HEV, PHEV, and BEV, 2010-2030 ... 21

Figure 13: Emission factors on driving stage of light duty gasoline vehicles ... 23

Figure 14: Emission reduction rate of HEV and PHEV compared with light duty gasoline vehicles under China IV emission standard ... 23

Figure 15: Weight share of major components for different vehicle fleets (except batteries) 24
Figure 16: Battery energy demand of different vehicle fleets, 2010 25
Figure 17: Major components and materials in a typical Li-ion battery 26
Figure 18: Component composition of a typical Li-ion battery cell...26
Figure 19: Material composition of major subparts in a Li-ion battery cell..27
Figure 20: Material composition of four different Li-ion battery cells 27
Figure 21: Material composition of Ni-MH (Left) and Pb-Ac (Right) batteries 27
Figure 22: Energy and material flow of lithium compounds in China...28
Figure 23: Energy and material flow of nickel compounds in China...29
Figure 24: Energy and material flow of copper compounds in China...29
Figure 25: Energy and material flow of aluminium compounds in China 30
Figure 26: Share of process fuels and upstream fossil energy use for producing typical vehicle materials 30
Figure 27: Contribution rate of each process in battery module production 31
Figure 28: Technical Progress of Production Processes of Key Materials..32
Figure 29: Regional variations in historical LDPV growth, 2002-2009 33
Figure 30: Projected LDPV stock in China, 2010-2030..34
Figure 31: Projected LDPV stock of the three developed regions, 2010-2030.......................................35
Figure 32: Survival rates for LDPVs with vehicle age in China..36
Figure 33: The trends for average annual VKT of light-duty vehicles...37
Figure 34: WTW petroleum consumption of LDPV in China, 2015-2030...39
Figure 35: WTW fossil fuel consumption of LDPV in China, 2015-2030..40
Figure 36: WTW CO2 emissions of LDPV in China, 2015-2030 41
Figure 37: WTW VOC emissions of LDPV in China, 2015-2030 42
Figure 38: WTW CO emissions of LDPV in China, 2015-2030 43
Figure 39: WTW NOX emissions of LDPV in China, 2015-2030..44
Figure 40: WTW PM2.5 emissions of LDPV in China, 2015-2030 45
Figure 41: WTW SO2 emissions of LDPV in China, 2015-2030..46
Figure 42: WTW petroleum consumption of LDPV in six regions of China, 2015-2030.....................48
Figure 43: WTW fossil fuel consumption of LDPV in six regions of China, 2015-2030......................49
Figure 44: WTW CO2 emissions of LDPV in six regions of China, 2015-2030 51
Figure 45: WTW VOC emissions of LDPV in six regions of China, 2015-2030 52
Figure 46: WTW CO emissions of LDPV in six regions of China, 2015-2030 53
Figure 47: WTW NOX emissions of LDPV in six regions of China, 2015-2030..................................55
Figure 48: WTW PM2.5 emissions of LDPV in six regions of China, 2015-2030 56

Figure 49: WTW SO2 emissions of LDPV in six regions of China, 2015-2030 58

Figure 50: Generation mix in Beijing, 2010-2030 60

Figure 51: Comparison of vehicle emission factors under old and new standards in Beijing, 2015-2030 61

Figure 52: WTW petroleum consumption of LDPV in Beijing and JJJ, 2015-2030 62

Figure 53: WTW fossil fuel consumption of LDPV in Beijing and JJJ, 2015-2030 62

Figure 54: WTW CO2 emissions of LDPV in Beijing and JJJ, 2015-2030 63

Figure 55: WTW VOC emissions of LDPV in Beijing and JJJ, 2015-2030 63

Figure 56: WTW CO emissions of LDPV in Beijing and JJJ, 2015-2030 64

Figure 57: WTW NOX emissions of LDPV in Beijing and JJJ, 2015-2030 64

Figure 58: WTW PM2.5 emissions of LDPV in Beijing and JJJ, 2015-2030 65

Figure 59: WTW SO2 emissions of LDPV in Beijing and JJJ region, 2015-2030 65

Figure 60: Power generation development in China, 2000-2012 66

Figure 61: a) Generation share in 2012 and b) Generation capacity share in 2012 66

Figure 62: Wind generation capacity distribution of China in 2011 67

Figure 63: The potential wind power electricity exporters for Beijing 67

Figure 64: Electricity production and consumption of provinces of China in 2010 68

Figure 65: a) Vehicle-cycle fossil energy use and b) CO2 emissions of major materials, 2010-2030 ... 73

Figure 66: Vehicle-cycle energy use and CO2 emissions of cell materials, 2010-2030 74

Figure 67: Comparison of vehicle cycle results of a) energy use and b) CO2 emissions of three typical batteries, 2010 75

Figure 68: Vehicle-cycle a) energy use and b) CO2 emissions of battery EV, 2010-2030 76

Figure 69: Vehicle-cycle a) energy use and b) CO2 emissions of NCM modules equipped on different vehicle fleet, 2010-2030 76

Figure 70: Contribution fraction of vehicle-cycle energy use of each sub components, 2010 77

Figure 71: Contribution rate of each process in battery module production, 2010 and 2030 78

Figure 72: Vehicle-cycle energy use of ICEV, HEV, PHEV50 and BEV, 2010-2030 79

Figure 73: Vehicle-cycle CO2 emissions of ICEV, HEV, PHEV50 and BEV, 2010-2030 79

Figure 74: Full life-cycle energy use of ICEV, HEV, PHEV50 and BEV, 2010-2030 80

Figure 75: Contribution of each stage to total life-cycle fossil energy use of ICEV, HEV, PHEV50 and BEV, 2010-2030 80

Figure 76: Full life-cycle CO2 emissions of ICEV, HEV, PHEV50 and BEV, 2010-2030 81

Figure 77: Accumulative full life-cycle fossil energy use during lifetime of a model-year 2010 sedan with different technologies 82

Figure 78: Accumulative full life-cycle CO2 emissions during lifetime of a model-year 2010 sedan with different technologies 83

Figure 79: Total PHEV/EV and HEV sales in 25 cities, as of Dec. 2012 ... 84

Figure 80: Annual light duty vehicle sales ratio of new energy vehicles in major cities and nationwide 85

Figure 81: Share of ICEV, HEV, PHEV and BEV to total new LDPV sales, a) Scenario 1 and b) Scenario 2 ... 87

Figure 82: Trends of a) Fossil fuel and b) CO2 emissions from penetration of HEV/PHEV/BEV in China ... 88

Figure 83: WTT and TTW CO2 emissions of LDPV fleet with various scenarios in China, 2010-2030 89

Figure 84: Trends of pollutant emissions for fleet in China: a) VOC, b) CO, c) NOX, d) PM2.5, 2010-2030 ... 90

Figure 85: Trends of fossil fuel use from penetration of HEV/PHEV/BEV in a) Beijing, b) Yangtze-River-Delta-Region, c) Pearl-River-Delta-Region, 2010-2030 ... 92

Figure 86: Trends of CO2 emissions from penetration of HEV/PHEV/BEV in a) Beijing, b) Yangtze-River-Delta-Region, c) Pearl-River-Delta-Region, 2010-2030 ... 94

Figure 87: Trends of pollutant emissions from penetration of HEV/PHEV/BEV in Beijing ... 95

Figure 88: Trends of pollutant emissions from penetration of HEV/PHEV/BEV in YRD ... 96

Figure 89: Trends of pollutant emissions from penetration of HEV/PHEV/BEV in PRD ... 96

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