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ORIGINAL PAPER

Comparing the life cycle Greenhouse Gas emissions from vehicle production in China and the USA: implications for targeting the reduction opportunities

Han Hao[1] · **Qinyu Qiao**[1] · **Zongwei Liu**[1] · **Fuquan Zhao**[1] · **Yisong Chen**[2]

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Abstract China is responsible for around one-quarter of global vehicle production. The associated Greenhouse Gas (GHG) emissions have become a major concern to the industrial sustainable development. With the aim of identifying the opportunities of cutting GHG emissions from China's automotive industry, this study estimates the life cycle GHG emissions from vehicle production in China and compares the results with the case in the USA from multiple perspectives. The results reveal that the GHG emissions from the production of a standard internal combustion engine-based passenger vehicle in China are around 9.6 ton per vehicle, 54% higher than the US level of 6.2 ton per vehicle. The power-intensive nature of vehicle production and China's higher GHG emission intensity of power generation are the major reasons behind the difference. Accordingly, total GHG emissions from passenger vehicle production in China were around 173.9 million tons in 2013, accounting for nearly 3% of the GHG emissions from the manufacturing and construction sector. Based on the analysis, it is recommended that China should further optimize the grid mix and reduce the emission intensity of power generation. Besides, emission intensities of steel and aluminum productions should be further reduced through applying energy-efficient technologies and promoting material recycling.

Keywords Vehicle production · Transport · Life cycle assessment · Greenhouse Gas emissions · Energy consumption

✉ Fuquan Zhao
zhaofuquan@tsinghua.edu.cn

[1] State Key Laboratory of Automotive Safety and Energy, Tsinghua University, Beijing 100084, China

[2] School of Automobile, Chang'an University, Xi'an 710064, China

Abbreviations

BF	Blast furnace
BFG	Blast furnace gas
BOF	Basic oxygen furnace
CAAM	China Association of Automobile Manufacturers
CCP	Consumable components production
COG	Coke oven gas
EAF	Electric arc furnace
EV	Electric vehicle
FCV	Fuel cell vehicle
GHG	Greenhouse Gas
HEV	Hybrid electric vehicle
HVAC	Heating, ventilating and air conditioning
ICE	Internal combustion engine
ICEV	Internal combustion engine vehicle
IEA	International energy agency
INDCs	Intended nationally determined contributions
IPCC	Intergovernmental panel on climate change
LCA	Life cycle assessment
NBSC	National Bureau of Statistics of China
NDRC	National Development and Reform Commission
OCP	Original components production
OICA	Organisation Internationale des Constructeurs d'Automobiles
PHEV	Plug-in hybrid electric vehicle
WTW	Well-to-Wheel

Introduction

The automotive industry is a typical traditional manufacturing industry, which has provided the world with billions of vehicles that play a core function in the modern society (Hao et al. 2016b). Driven by sustained economic growth, global vehicle production experienced rapid growth over the past century, reaching historic high of 90.8 million in 2015 (OICA 2016). Despite the benefits vehicles bring to the society, they have caused significant energy and environmental concerns. Especially, as the external effect of vehicle production, millions of tons of Greenhouse Gas (GHG) are emitted into the atmosphere every year (Zhao et al. 2016). As estimated by International Energy Agency (IEA), CO_2 emissions from the manufacturing and construction sector, to which the automotive industry is an important contributor, accounted for 37.4% of global energy-related CO_2 emissions in 2013 (IEA 2015).

China is facing great pressure from the international community to reduce GHG emissions. In 2013, China's anthropogenic CO_2 emissions reached 9.0 billion tons, accounting for 28% of global total (IEA 2015). In the Intended Nationally Determined Contributions (INDCs) China announced in 2015, total CO_2 emissions were promised to peak before 2030. Furthermore, the CO_2 intensity (measured as CO_2 emissions per unit of GDP) in 2030 is expected to decrease by 60–65% compared to the 2005 level (Chinese government 2015). At the same time, China is very representative when analyzing the GHG emissions associated with vehicle production. China's vehicle production experienced fast growth over the past decade, from 2.1 million in 2000 to 24.5 million in 2015 (CAAM 2016). Currently, China's vehicle production represents around one-quarter of global vehicle production (OICA 2016). Considering the fast economic development and urbanization progress, there is still solid further growth potential in China's vehicle production (Hao et al. 2011a, b). Under such a circumstance, China has great need in reducing GHG emissions to realize the promise (Howell et al. 2014) and the automotive industry has been targeted as a priority in the overall GHG reduction scheme (Hao et al. 2014).

From a life cycle perspective, almost all phases of vehicle production are associated with GHG emissions (Xia et al. 2016), including raw material extraction, transportation, material production, transformation, vehicle assembly, disposal, recycling, etc. The majority of GHG emissions are caused by the use of process fuels, such as coal, diesel, electricity, etc. Yet, a small proportion of the GHG emissions are sourced from the consumption of carbon-containing materials.

Due to the fact that GHG emissions from the vehicle use phase, i.e., GHG emissions caused by the use of vehicle fuels, are higher than those from the vehicle production phase, most existing studies on vehicles associated GHG emissions paid more attention to the use phase, or what is normally referred to as the Well-to-Wheel (WTW) stages, while the production phase was studied as a fixed value influencing the performance of vehicles during life time. Such studies typically compared life cycle emissions among vehicles with different propulsion systems, including conventional internal combustion engine vehicles (ICEV), hybrid electric vehicles (HEV), plug-in hybrid electric vehicles (PHEV), battery electric vehicles (EV), fuel cell vehicles (FCV), etc. Bauer et al. (2015) evaluated the environmental impacts of current and future vehicles, finding that EVs and FCVs could help to reduce GHG emissions if non-fossil energy resources were used for electricity and hydrogen production. Hawkins et al. (2013) developed a life cycle assessment (LCA) model for estimating GHG emissions from ICEVs and EVs, revealing that with European or US electricity mix assumed, EVs could help to decrease GHG emissions compared with ICEVs. Wang et al. (2013) compared the emissions from EVs and FCVs in China's context. The results indicated that under the Chinese generation mix, the energy and environmental performances of EVs became worse. Orsi et al. (2016) conducted a research on the emissions, energy use and cost of different vehicles in different regions, finding that compressed natural gas vehicles and EVs are the potential alternatives that help to reduce oil consumption and emissions in the private transport sector.

Meanwhile, numerous studies also focused on estimating GHG emissions from the vehicle production phase. This is a very important complement to the vehicle use phase studies as they significantly extended the study scope and closed the life cycle loop. Nanaki and Koroneos (2013) conducted an environmental and economic comparison of vehicles with three different types of propulsion systems, with both vehicle production and use phases covered. The results indicated that the environmental impacts of EVs depended substantially on the source of electricity. Zamel and Li (2006) analyzed the life cycle GHG emissions from ICEVs and FCVs in Canada. Both vehicle production and vehicle use phases are accounted. They found that the total emissions of an FCV were 49% lower than an ICEV.

Besides, the impact of vehicle light-weighting on the energy consumption and GHG emissions from vehicle production has also attracted attentions from the research community. Dhingra and Das (2014) analyzed the life cycle environmental impacts of engines made of different materials, finding that replacing the steel and cast iron in the engine with other metals such as aluminum and magnesium, which was lighter, could help vehicles achieve better fuel economy. Das (2000) compared the life cycle energy consumption and emissions between vehicles using aluminum and conventional steel. The results indicated that 52 GJ/vehicle life cycle energy savings would occur if steel was replaced by aluminum. Lewis et al. (2014) assessed the reduction potential of emissions from vehicle electrification and weight reduction. The results showed that the greatest emission reductions occurred when steel was replaced by aluminum. Kim et al. (2010) compared the reduced emissions during vehicle use phase with the increased emissions associated with the production of lightweight vehicles, finding that GHG emissions from aluminum light-weighting varied with the place where aluminum was produced and whether recycled aluminum could be used instead of primary aluminum.

Existing studies have provided a mature framework for analyzing the GHG emissions from vehicle production. However, as revealed by existing studies, the GHG emissions from vehicle production exhibit significant regional disparities. This can be attributed to the differences in various factors, including the emission factors of process fuels, vehicle manufacturing technology, the use of recycled materials, etc. Under such a circumstance, the results obtained in one region's context can be of low relevance to another. Especially, when considering the situation of China, the GHG emissions from vehicle production can be quite different from other countries due to its uniqueness in power generation, material flow, etc. Unfortunately, the GHG emissions from vehicle production in China have not been fully investigated, mainly due to the lack of data and synergy.

To fill such a gap, this study aims at estimating the GHG emissions from vehicle production in China. For this purpose, this study employs a life cycle framework, under which the energy consumption and emissions throughout all phases of vehicle production are taken into consideration. In order to reflect the situation in China, a localized database is established by using the China-specific data from a wide range of literatures. This study aims to answer what roles the different materials, different phases and different process fuels of vehicle production play in the overall GHG emissions. Furthermore, as targeting the reduction opportunities, this study takes the situation in the USA, the country with the second largest vehicle production and top manufacturing techniques (OICA 2016), as benchmarks.

Methods and data

System boundary

This study employs the cradle to gate concept, under which the GHG emissions are considered. This system is established on the basis of the real vehicle manufacture process in China, including material production, material transformation, components production, assembly, and consumable components replacement, as Fig. 1 shows. And several assumptions are imported from GREET. The inputs into this system are categorized into energy input and material input; outputs categorized into GHG emissions and other emissions. As this study focuses on vehicle production, the latter phases of vehicle life cycle, including vehicle distribution, use, and disposal are not covered in the analysis.

Regarding GHG emissions, this study considers both direct emissions and indirect emissions, which complies with the definition of Scope 3 emissions (Greenhouse Gas Protocol 2015). Specifically, GHG emissions associated with the combustion of process fuels within the vehicle manufacturing entities, production of input energy and materials are both considered. Due to the limited impact and data availability, GHG emissions caused by the consumption of carbon-containing materials are not considered in the analysis.

Historically and currently, internal combustion engine (ICE)-based passenger cars have been dominating the vehicle market. Accordingly, GHG emissions from the production of ICE-based passenger cars represented the majority of the GHG emissions from vehicle production. Although new propulsion technologies are expected to gain higher market shares, their impact on vehicle manufacturing industry is quite limited in the near future. Actually, in China, the capacity of new energy vehicles is expected to reach 5 million in 2020 (Chinese State Council 2012), which is about only one-fifth of the level of ICE-based passenger cars in 2015. Furthermore, as the manufacture techniques of new energy vehicles have not been fully developed in China, especially traction batteries, it is not possible to deliver reliable results on the life cycle emissions of new energy vehicles in the far future when they dominate the vehicle market. In addition, as mentioned before, numerous studies focused on the life cycle emissions of vehicles with different propulsion systems. They have analyzed this topic clearly in developed countries, which can be useful references for China. With such consideration, ICE-based passenger car is chosen as the reference vehicle. However, due to lack of relevant studies, the definition for a standard passenger car in China's context is unclear. In this study, the vehicle specification is based on the Automotive System Cost Model (Das 2004), which is adopted by GREET as well, in which a standard mid-size (comparable to the B-class car in China's market context) ICE-based passenger car with full specifications is defined (Burnham 2012). This approximation can be justified by the fact that in the context of automotive industry globalization, the vehicle models introduced into the US market and Chinese market have become more and more synchronized. Besides, by using the same reference vehicle, the results from this study become comparable to the estimations in the US context, which will be further discussed in the results section. The vehicle specification is presented in Table 5 in "Appendix".

Methods

The life cycle GHG emissions from vehicle production can be derived through Eq. (1) to (4).

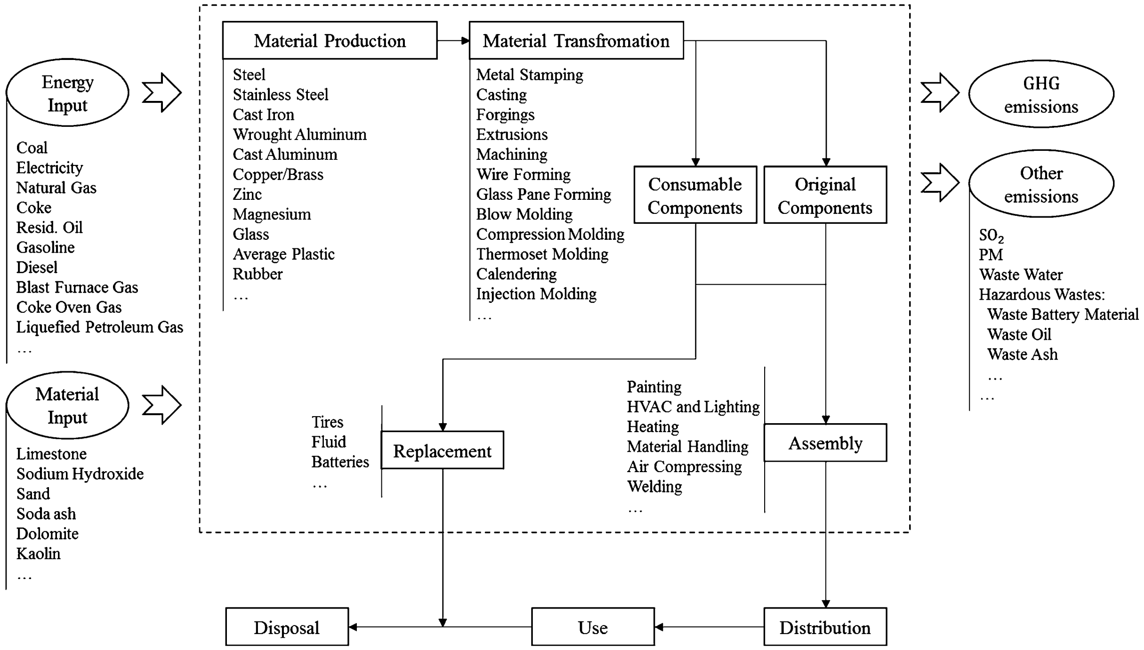

Fig. 1 System boundary defined in this study

$$\mathrm{CE}_{\mathrm{MP/MT}} = \sum_{i}\sum_{j}\mathrm{EF}_{j}\cdot\sum_{k}\mathrm{EC}_{i,j,k} \tag{1}$$

where, $\mathrm{CE}_{\mathrm{MP/MT}}$ is the GHG emissions from the material production/material transformation phase of vehicle production (kg-$\mathrm{CO_2}$eq); EF_j is the life cycle GHG emission factor of process fuel j (kg-$\mathrm{CO_2}$eq/MJ); $\mathrm{EC}_{i,j,k}$ is the consumption of process fuel j for process k within the production/transformation phase of material i (MJ).

$$\mathrm{CE}_{\mathrm{OCP}} = \mathrm{CE}_{\mathrm{MP}} + \mathrm{CE}_{\mathrm{MT}} \tag{2}$$

$$\mathrm{CE}_{\mathrm{CCP}} = \mathrm{CE}_{\mathrm{TI}} + \mathrm{CE}_{\mathrm{BA}} + \mathrm{CE}_{\mathrm{FL}} \tag{3}$$

$$\mathrm{CE}_{\mathrm{VP}} = \mathrm{CE}_{\mathrm{OCP}} + \mathrm{CE}_{\mathrm{CCP}} + \mathrm{CE}_{\mathrm{AS}} \tag{4}$$

where, $\mathrm{CE}_{\mathrm{VP}}$ is the life cycle GHG emissions from vehicle production (kg-$\mathrm{CO_2}$eq); $\mathrm{CE}_{\mathrm{OCP}}$, $\mathrm{CE}_{\mathrm{CCP}}$ and $\mathrm{CE}_{\mathrm{AS}}$ are the GHG emissions from original components production, consumable components production and vehicle assembly (kg-$\mathrm{CO_2}$eq); $\mathrm{CE}_{\mathrm{TI}}$, $\mathrm{CE}_{\mathrm{BA}}$ and $\mathrm{CE}_{\mathrm{FL}}$ are the GHG emissions from the productions of tires, batteries and fluids (kg-$\mathrm{CO_2}$eq).

The system is optimized in order to simplify the calculation and reveal the estimation of GHG emissions from different divisions more clearly. Based on a series of researches from Valipour's team, the importance and practicability of this kind of method were proved in different fields, especially hydrodynamics, including irrigation system design (Valipour 2012a) and further simulation (Mahdizadeh et al. 2015), precipitation analysis (Valipour 2016), surface irrigation simulation (Valipour 2012b) and further design (Valipour et al. 2015), and new water lift devices analysis (Yannopoulos et al. 2015).

Data

As mentioned above, the major intended contribution of this study is to estimate the China-specific GHG emissions from vehicle production. This is mainly realized by localizing the database. The database contains thousands of inputs, such as process energy efficiency, the shares of process fuels, material efficiency, emission factors of process fuels, etc. Due to data availability, it is almost impossible to localize the whole database. Instead, this study focuses on localizing some key data inputs. Specifically, first, the GHG emissions associated with steel production is determined by using the China-specific data. This is the most important step because steel alone accounts for 62% of total vehicle weight. Second, the GHG emissions associated with aluminum production is calculated by using China-specific data. This is not only based on the consideration that aluminum production accounts for a significant share of GHG emissions from vehicle production, but also due to the fact that aluminum production is power-intensive, which introduces considerable regional disparity considering the uniqueness of power generation in China. Third, the GHG emission factors of the process fuels are localized, because these emission factors have an overall influence on the calculations of the model. In the following section, the sources and treatment methods of these localized data inputs are introduced in detail. Other data, if not noted, are adopted from the GREET model.

Material composition of the reference vehicle

As mentioned above, the material composition of the reference vehicle is determined by referring to the vehicle specification, as shown in Table 1. It can be found that the use of materials for vehicle production is quite concentrated. Steel alone accounts for 62.3% of total vehicle weight. The top five materials add up to over 90% of total vehicle weight. In this regard, this study puts major effort on analyzing GHG emissions associated with these dominating materials.

Material production

The major data sources used for compiling the database for the consumptions of process fuels during the production of different materials are presented as follows.

Steel: Steel production comprises the processes of iron ore extraction and processing, coke production, sintering, pelletizing, blast furnace–basic oxygen furnace (BF–BOF), continuous casting, hot rolling, cold rolling and coating/cutting. The Electric Arc Furnace (EAF) process is applied in some factories. For the iron ore extraction and processing processes, existing data in China's context covers only the total energy consumption and the electricity consumption (Editorial Board of China Steel Yearbook 2015), which is not detailed enough to support the analysis. Instead, data from the GREET model are employed. For the coke production process, data from Weng (2009) are employed, which was based on the investigation of over 20 Chinese coke producers. For other processes, data are localized based on the reported data from one of the biggest steel manufacturers in China (Jing et al. 2014).

Table 1 The weight distribution of materials consumption of the reference vehicle

Material	Share by weight (%)	Weight/kg
Steel	62.26	804.9
Average Plastic	11.12	143.7
Iron	10.89	140.8
Cast Aluminum	4.63	59.9
Glass	2.87	37.1
Rubber	2.26	29.2
Wrought Aluminum	2.20	28.4
Copper/Brass	1.87	24.2
Others	1.91	24.6
Total	100	1292.8

Fluid, tires and batteries are not included

Aluminum: Aluminum production comprises the processes of bauxite mining, anode production, alumina production, aluminum smelting and ingot casting. Hao et al. (2016a) estimated the GHG emissions from primary aluminum production in China, finding that the national average GHG emissions from China's primary aluminum production were 16,500 kg-CO_2eq/t ingot in 2013, which is much higher than the global average. Relevant data are incorporated into the database of this study.

Other materials: For other materials, this study uses the data from the GREET model. There are three reasons for this approximation: It is hard to get the detailed data from China's factories; the manufacturing technologies are very similar between China and the USA (such as different kinds of plastics); the other materials only account for a small proportion of vehicle weight.

Material transformation

When it comes to material transformation, data include the energy consumption of transformation processes, transportation and storage. This study assumes that all transportation is by road using a standard diesel truck with the load of 9.3 t (ANL 2015) and the average distance from the production plant to the transformation plant is 200 km. The assumption about storage in this study is the imported from GREET-2015. As data for many processes are difficult to gather, the surrogate-based method established by Sullivan et al. (2013) is employed. For the transformation processes that are unclear in terms of energy consumption and GHG emissions, other processes containing similar physical or chemical courses are employed as surrogates. For example, the process of aluminum stamping is surrogated by steel stamping, which shares almost the same physical course. By doing this, much more sufficient data can be obtained to populate the database. The major material transformation processes and the surrogate processes are presented in Table 2. The details are discussed as follows.

Steel: Steel transformation consists of two major processes: stamping and machining, which are for virgin steel and recycled steel, respectively. In China's context, virgin steel accounts for about 90% of total steel consumption, while recycled steel accounts for the other 10% (Yang et al. 2010). This share is used as the basis for separating virgin steel and recycled steel consumptions. Regarding the consumptions of process fuels, due to data availability in China's context, data from the GREET model are employed.

Aluminum: For cast aluminum, the processes consist of casting and machining. While for wrought aluminum, the

Table 2 Material transformation processes and the surrogate processes

Material	Transformation	Surrogate process
Steel	Material stamping	Steel stamping
	Machining	Steel Machining
Aluminum	Material stamping	Steel stamping
	Casting	Aluminum casting
	Extrusion	Aluminum extrusion
	Machining	Steel machining
Iron	Casting	Iron casting
	Forging	Iron forging
	Machining	Steel machining
Plastics	Extrusion	High-density polyethylene pipe extrusion
	Compression molding	Compression molding rubber
	Calendering	Polyvinyl chloride calendering
	Injection molding	Polypropylene parts molding
Copper	Wire forming	Copper wire forming
Glass	Glass pane forming	Float glass forming
Polymers	Blow molding	High-density polyethylene bottles blow molding
Rubber	Compression molding	Compression molding rubber
	Injection molding	Polypropylene parts molding
Polymer resins	Thermoset molding	Polyurethane foams molding

processes consist of stamping (cold rolled) and extrusion. The weight shares of these two kinds of aluminum are shown in Table 1. This study uses the data from the GREET model.

Other materials: For other materials, the transformation is shown in Table 2. For copper, the process is mainly wire forming. For glass, the process is mainly float glass forming. For rubber, two major processes are included: compression molding accounts for 89%, while injection molding accounts for 11%. For plastics, different kinds of plastics are treated with different processes. For magnesium, casting and molding are the major processes, which are based on data from the GREET model.

Consumable components

Tires: For the tires, this study uses the data from a specific radial tire producer in China (Yang et al. 2014). The processes of exploitation, material production, part manufacturing and tire production are all considered in the calculation. It is assumed that the tires are replaced for three times during the life cycle of a vehicle.

Fluids: The fluids used on vehicles include engine oil, brake fluid, transmission fluid, powertrain coolant, windshield fluid, adhesives, etc. Due to data availability in China's context, this study assumes that GHG emissions from fluids production in China are the same with the level in the USA. The numbers of their replacements during the life cycle of a vehicle are assumed by referring to the GREET model.

Batteries: For ICE-based vehicles, the battery used on vehicle is normally a small-capacity lead-acid battery that is used for lighting and other electric appliances. As the GHG emissions from battery production is quite low, this study uses the data from the GREET model. It is assumed that the lead-acid battery is replaced for two times during the life cycle of a vehicle.

Vehicle assembly

Vehicle assembly can be divided into six major processes: painting, HVAC and lighting, heating, material handling, air compressing and welding (Sullivan et al. 2010). This study uses the data from the GREET model.

GHG emission factors

The life cycle GHG emission factors of different kinds of process fuels are shown in Table 3. CO_2, CH_4 and N_2O are taken into consideration, and the convert factors are 1, 25 and 298. Most emission factors are China-specific values, which are compiled based on multiple data sources. For electricity, this paper estimates the nationwide average emission factor by using the capacity-weighted average of each province's emission factors. It should be noted that the GHG emission factors of coke, blast furnace gas (BFG) and coke oven gas (COG) only consider the direct emissions from fuel combustion because the indirect emissions have already been accounted in the steel production phases.

Table 3 Life cycle GHG emission factors of different process fuels

Process fuel	Emission factor ($g\text{-}CO_2eq/MJ$, $g\text{-}CO_2eq/kWh$)	Data sources
Coal	97.5	Chen (2014), IPCC (2006)
Electricity	834.5	Ma et al. (2014), NBSC (2016)
Natural Gas	64.8	Chen (2014), IPCC (2006)
Coke	107.0	NDRC (2014), IPCC (2006)
Residual Oil	91.7	Chen (2014), IPCC (2006)
Gasoline	87.7	Chen (2014), IPCC (2006)
Diesel	90.7	Chen (2014), IPCC (2006)
BFG	260.1	IPCC (2006)
COG	44.5	IPCC (2006)

Results and discussions

Total emissions

The calculation results of the life cycle GHG emissions from vehicle production in China are presented in Table 4. As a comparison, the results from the GREET model, which reflect the US situation, are also provided (ANL 2015). It can be found that the GHG emissions from vehicle production in China are 9.6 t per vehicle, 54% higher than the US level, 6.2 t per vehicle. This substantial difference is caused by several factors, which are further discussed in the following sections.

However, errors exist in the estimation, which are from three major sources: (1) this study assumes that all the steel (as well as other materials) plants in China adopt the same manufacture techniques, while some small steel plants are still using former techniques and causing more GHG emissions; (2) this study assumes that the vehicle production is evenly distributed in each region of China and then applies average GHG emission factors, while materials are mass-produced in several specific provinces; (3) this study assumes that the GHG emission factors are fixed in spite of the different combustion modes in different regions, while the amount of CH_4 and N_2O emissions vary among different combustion modes.

Emissions from vehicle components

The GHG emissions from vehicle components are presented in Fig. 2. Total GHG emissions from vehicle production are categorized into three groups, original components production, consumable components production and assembly. It can be found that original components production (OCP) is the dominating source of GHG emissions, which accounts for around 75% of total GHG emissions. The shares of GHG emissions from consumable components production (CCP) and assembly are similar at the level of around 14%. For all three categories of GHG emissions, China has higher values than the USA The GHG emissions from OCP are 7224 kg in China and 4141 kg in the USA, implying a difference of 3083 kg, contributing to 91.9% of the overall difference. The differences in GHG emissions from consumable components production and

Table 4 Life cycle GHG emissions from vehicle production

$kg\text{-}CO_2eq/vehicle$	China			The USA		
	Material production	Material transformation	Total	Material production	Material transformation	Total
Original components	6504.6	719.8	7224.4	3562.5	578.7	4141.2
Steel	4444.0	426.4	4870.4	2635.1	325.0	2960.1
Aluminum	1457.4	55.1	1512.5	371.9	53.6	425.5
Other materials	603.2	238.3	841.5	555.6	200.0	755.6
Consumable components		1371.2			1252.6	
Tires		472.1			353.5	
Fluid		852.9			852.9	
Batteries		46.2			38.7	
Assembly		1001.3			847.4	
Total		9596.9			6241.2	

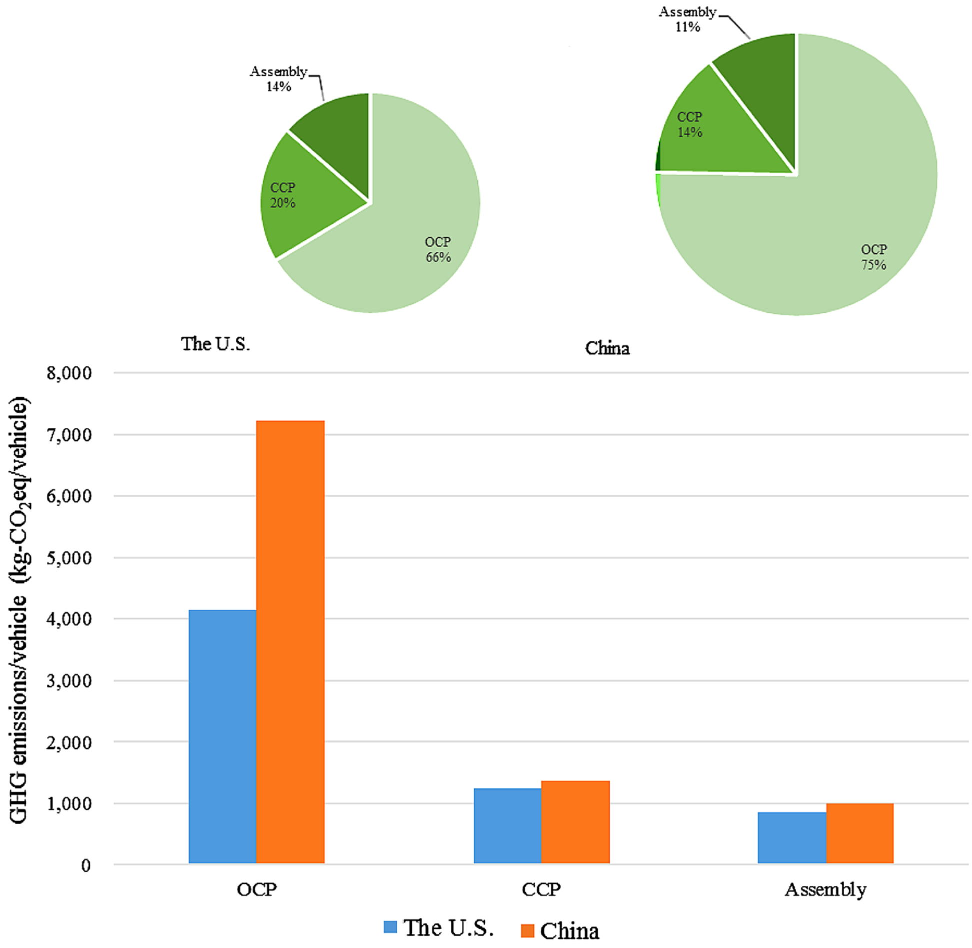

Fig. 2 Effect of components on emissions (percentage and amount). *Note*: OCP consists of the production of body system, powertrain system, transmission system and chassis system, which are expected to face no replacements during life time. CCP consists of the production of batteries, fluids and tires, which are expected to be replaced for several times during life time

assembly are quite small between China and the USA. With such regard, in the following sections, all GHG emissions are dedicated to OCP.

Emissions from materials

Figure 3 shows the GHG emissions from materials. To highlight the difference, GHG emissions associated with steel and aluminum are separated out as two single categories, while GHG emissions associated with other materials are aggregated into one single category. For both China and the USA, steel and aluminum together contribute to over 80% of total GHG emissions from OCP.

When comparing China and the USA, huge differences can be found both in steel- and aluminum-associated GHG emissions. The steel-associated GHG emissions are 4870 kg in China and 2960 kg in the USA, implying a difference of 1910 kg, contributing to 62% of the overall difference. The major reason behind this difference lies in the different compositions of steel production processes. In China, about 90% steel facilities are based on the BF–BOF process producing primary steel, while the other 10% based on the EAF process producing secondary steel (Li and Zhu 2014). As a comparison, in the USA, the situation is 73.6% BF–BOF process versus 26.4% EAF process (ANL 2015). The EAF process has much lower GHG emissions than the BF–BOF process (Serrenho et al. 2016).

The aluminum-associated GHG emissions are 1513 kg in China and 426 kg in the USA, implying a difference of 1087 kg, contributing to 35% of the overall difference. The major reason behind this difference is that the production of aluminum is power-intensive, and the emission factor of power generation in China is much higher than that in the USA (Lin et al. 2016).

Emissions from phases

Figure 4 presents the GHG emissions from the material production phase and the material transformation phase. The activities within the material production phase occur in the upstream factories, such as steel plants. The activities within the material transformation phase happen partially in upstream plants and partially in the vehicle manufacturing factories. It can be found that for both China and the USA, GHG emissions from the material production phase account for over 85% of total CO_2 emissions.

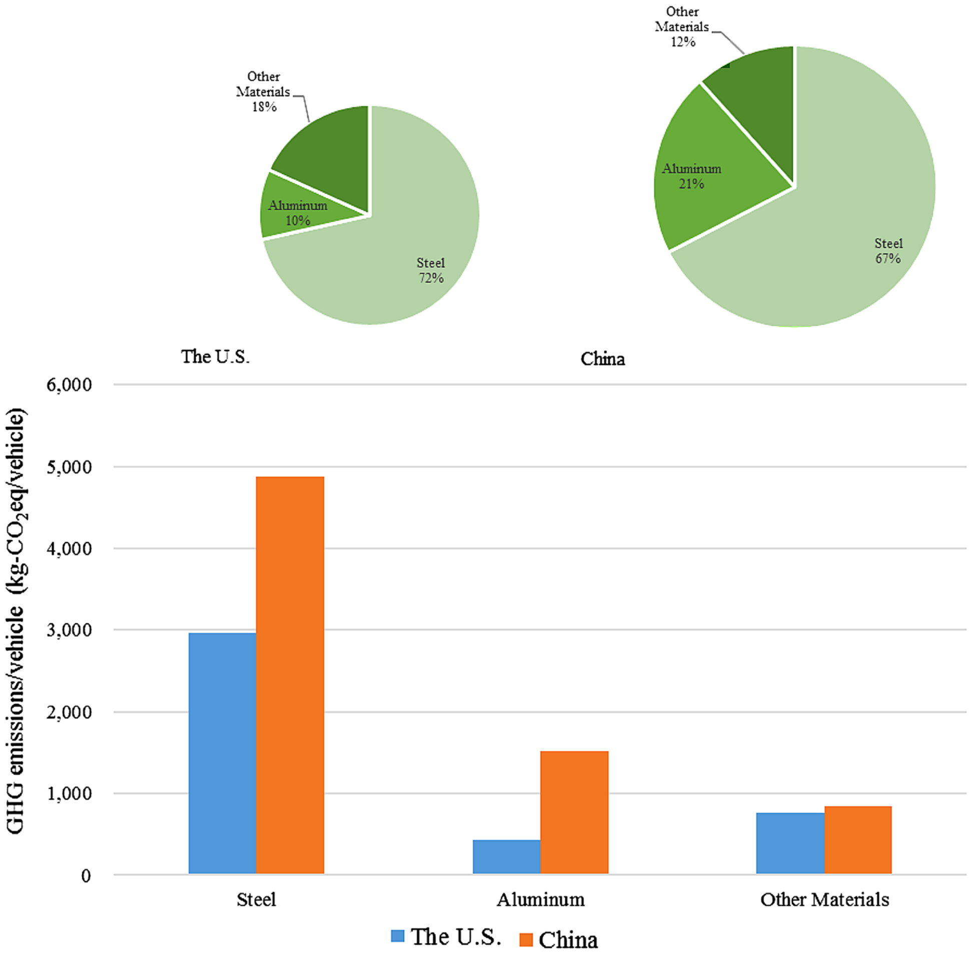

Fig. 3 Effect of materials on emissions (percentage and amount). *Note*: the effect of materials is dedicated to GHG emissions from original components production. Other Materials consist of iron, plastic, copper, glass, rubber and others

For both of the two phases, the GHG emissions in China are higher than the levels in the USA. Relatively, the difference within the material production phase is more significant. The GHG emissions from the material production phase in China and the USA are 6505 and 3563 kg, implying a difference of 2942 kg, contributing to 95% of the overall difference. Thus, the GHG emissions gap of vehicle production should be mostly attributed to the upstream industry rather than the vehicle manufacturing industry itself.

Emissions from process fuels

As mentioned above, GHG emissions covered in this study are the GHG emissions from the combustion of process fuels. Therefore, it is possible to observe GHG emissions from the process fuel perspective, as presented in Fig. 5. It can be found that electricity is the largest source of GHG emissions both in China and the USA, which account for around 40% of total GHG emissions. The GHG emissions from electricity consumption in China are 2812 kg, 70% higher than the US level, 1651 kg. The difference is mostly caused by the fact that power generation in China is much more GHG-intensive than the USA.

Regarding GHG emissions from other process fuels, significant disparities also exist. In the USA, electricity, coke and natural gas are the top three sources of GHG emissions. As a comparison, in China, coal becomes the second largest source of GHG emissions. The GHG emissions from coal use are 2099 kg in China and 423 kg in the USA, implying a difference of 1676 kg, contributing to 54% of the overall difference. This is accompanied by the fact that GHG emissions from natural gas use in China are only 8% higher than the level in the USA. This reflects the difference in the energy structure between these two countries.

Policy implications

Figure 6 summarizes the GHG emissions composition from vehicle production by components, materials, phases and process fuels. OCP dominates when considering GHG emissions from vehicle production both in China and the

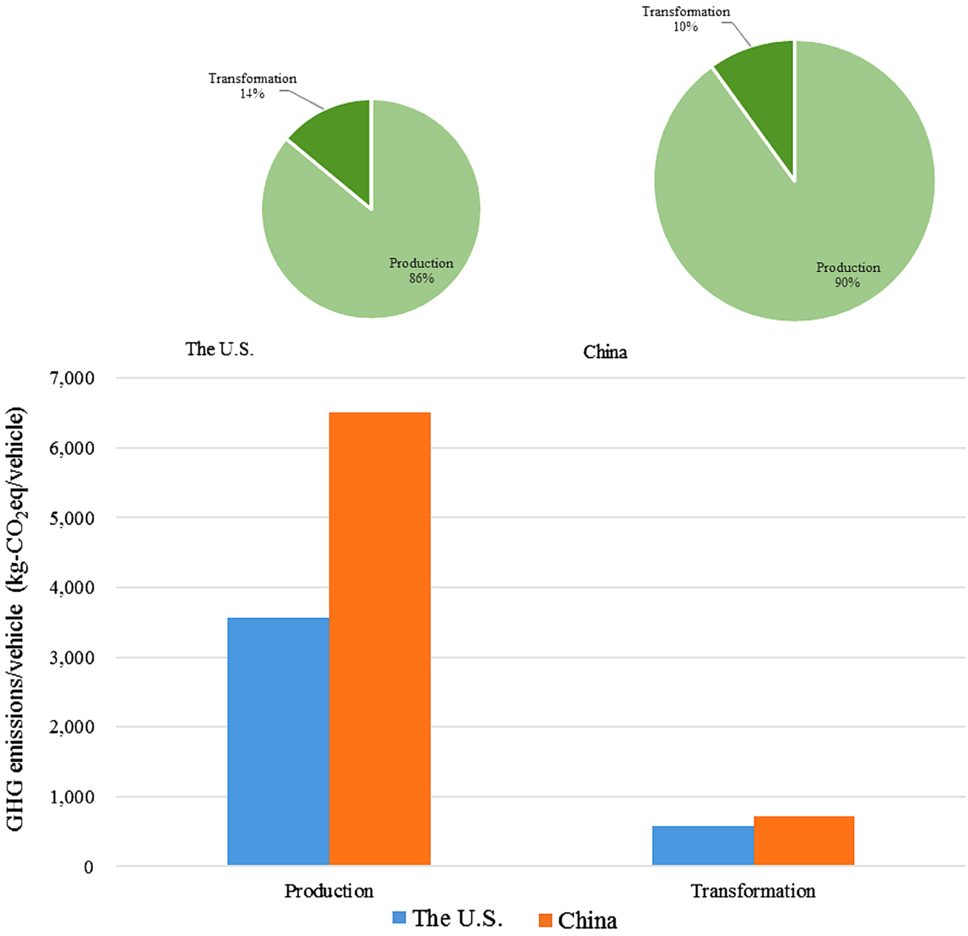

Fig. 4 Effect of phases on emissions (percentage and amount). *Note*: the effect of phases is dedicated to GHG emissions from original components production. Production consists of the processes before the output of materials (i.e. Iron ore extraction and processing, coke production and steel production). Transformation consists of the processes from material output to components assembly

USA. And over 80% of the GHG emissions from OCP production are caused by steel and aluminum. At the same time, the GHG emission level of steel and aluminum in China is much higher than the level in the USA. That is to say, huge reduction potentials exist in China based on the development of steel and aluminum industries, while more attention should be paid to the production phase. From the processing fuel point of view, besides the huge amount of coal consumed during steel production, GHG emissions from electricity in China account for a larger proportion than in the USA, revealing that the energy structure plays an important role as well. Such information is of high relevance to policy makers seeking opportunities to reduce GHG emissions from the automotive manufacturing industry. Using the estimated number in this study as a basis, the GHG emissions from the production of passenger vehicles in China were around 173.9 million tons in 2013, accounting for nearly 3% of the GHG emissions from the manufacturing and construction sector (IEA 2015). If the GHG intensity of vehicle production in China can be as low as the level in the USA, 60.8 million tons of GHG emissions can be cut. This number will become more considerable with China's vehicle production growing higher in the future. Therefore, it is important for China to take measures to reduce the GHG emissions from vehicle production.

The efforts for reducing GHG emissions from vehicle production should be focused on two aspects. First, the GHG emission intensities of steel and aluminum productions should be further reduced. For steel production, China should promote the use of recycled steel as raw materials, coupled with the development of EAF process. Currently, the share of recycled steel used for steel production in China is only 11%, compared to 90% in Turkey, 70% in the USA, 56% in the EU, and the world average of 37% (Wübbeke and Heroth 2014). Steel production in China is 15–20% more energy-intensive than the top runners in the world (Wang et al. 2007). For aluminum production, China should place the aluminum production capacity in power-clean regions. As suggested by Hao et al. (2016a), due to the disparity in power generation, provincial GHG emissions from primary aluminum production range from 8.2 t-CO$_2$eq/t ingot (Qinghai) to 21.7 t-CO$_2$eq/t ingot (Inner Mongolia). Besides, aluminum recycling should be further promoted. The emission intensity of secondary aluminum production is only 1/24 of primary aluminum production in China's context.

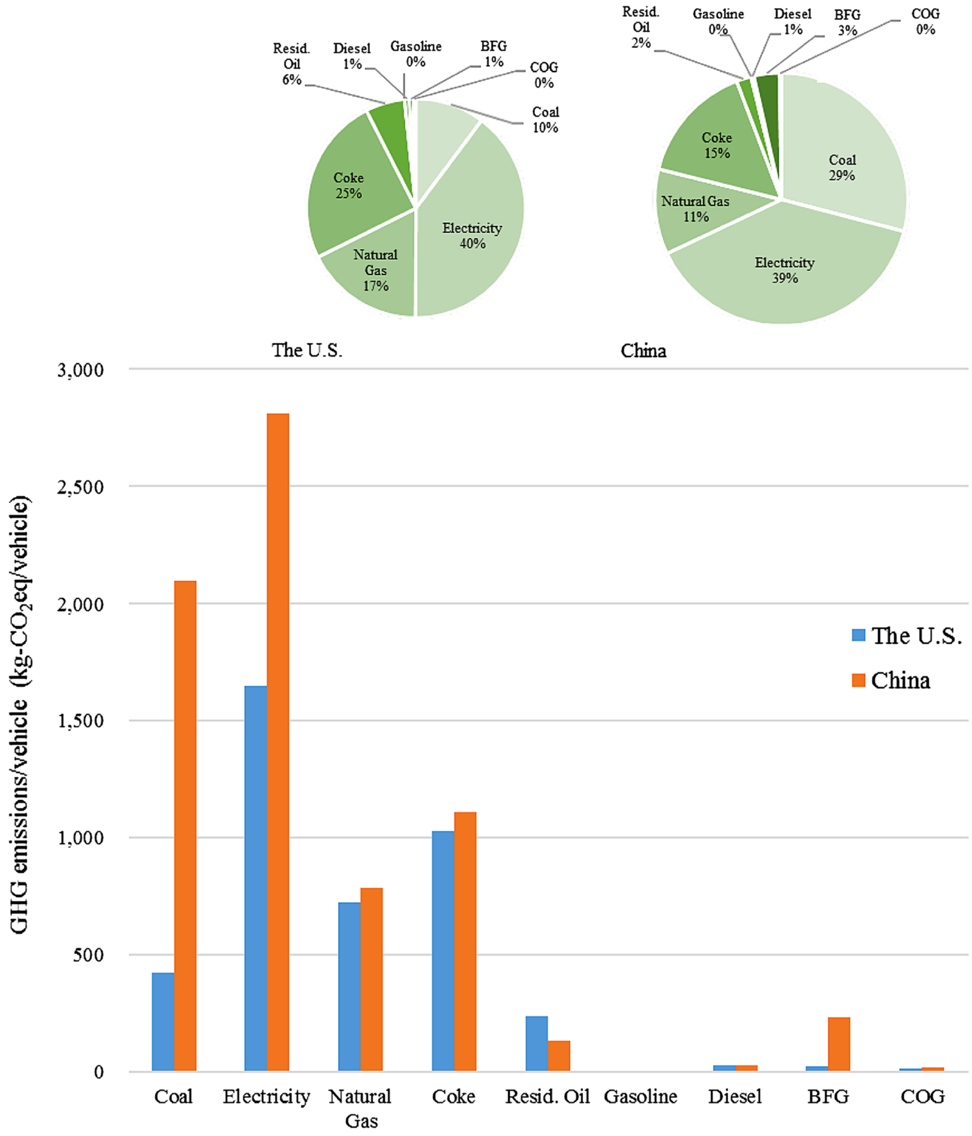

Fig. 5 Effect of fuels on emissions (percentage and amount). *Note*: the effect of fuels is dedicated to GHG emissions from original components production

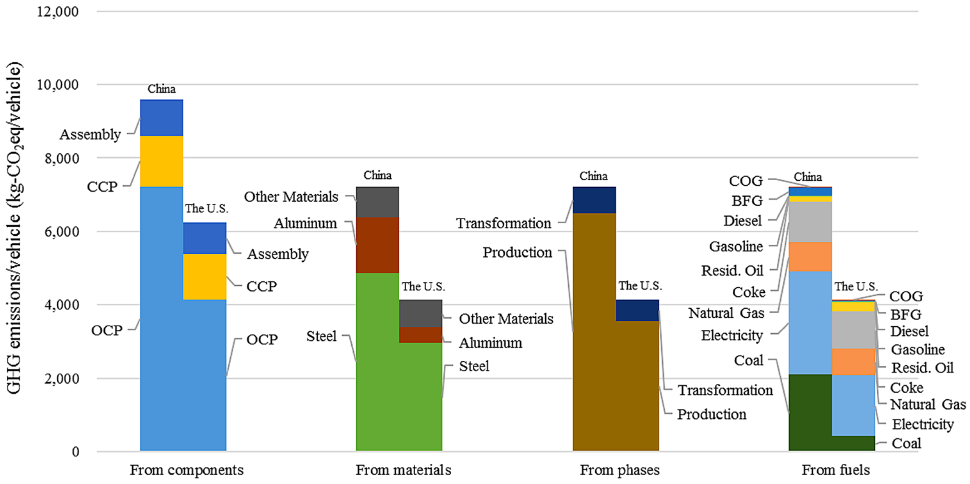

Fig. 6 A summary of the composition of GHG emissions from vehicle production. *Note*: the effect of materials, phases and process fuels are dedicated to GHG emissions from OCP

Second, China should reduce the GHG emission factors of process fuels, especially electricity. The life cycle GHG emission factor of electricity in China is currently much higher than the level in the USA. Besides, China should also take advantage of the regional grids which are lower in GHG emission intensity. The marginal CO_2 emission factor of power generation ranged from 809.5 g/kWh (Eastern Grid) to 1128.1 g/kWh (Northeastern Grid) (NDRC 2015).

Furthermore, China could consider some other carbon reduction techniques such as carbon sequestration, which can help to reduce the GHG emissions from the whole process of vehicle manufacture, especially power generation and steel production.

On the other hand, from the GHG emissions per passenger point of view, public transportation can contribute to the reduction of GHG emissions from production as well. For instance, the GHG emissions from the production of a diesel bus with 86 passenger capacity are about 149 t-CO_2eq in the USA (McKenzie and Durango-Cohen 2012), which means only 1.7 t-CO_2eq per passenger if fully loaded, about 30% less than the GHG emissions from the production of a passenger car in the USA estimated in this study.

Conclusions

In this study, the life cycle GHG emissions from vehicle production in China are estimated and compared with the case in the USA from multiple perspectives. The results reveal that the GHG emissions from the production of a standard ICE-based passenger vehicle in China are around 9.6 t per vehicle, 54% higher than the US level of 6.2 t per vehicle. The power-intensive nature of vehicle production and China's higher GHG emission intensity of power generation are the major reasons behind the difference. In comparison, the difference of GHG emissions from the use phase of an ICEV between China and the USA is quite small due to the fixed combustion mode. For example, the emission factor of gasoline in China is 87.7 g-CO_2eq/MJ, consisting of 18.1 for fuel production and 69.6 for combustion, while the numbers in the USA are 81.8, 12.7 and 69.1 (ANL 2015). This situation would cause a 7% difference during the use phase.

Despite the significant policy implications this study reveals, further steps are needed to obtain more precise estimations. Although this study uses China-specific data as much as possible to reflect the localized situation, some hard-to-obtain data are still based on the GREET model, which reflects the US situation. Such data include the production and transformation of several materials, the energy consumption of vehicle assembly, batteries, fluids, etc. The data basis should be further enhanced with more data collected. A GREET model-fashioned database for vehicle production should be established. With such database, more opportunities of GHG emissions reduction from vehicle production can be identified to help the government shape more appropriate policies.

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Appendix: Vehicle specification

See Table 5.

Table 5 The material consumption of the reference vehicle

Vehicle system	Components	Materials
Body System	Body-in-white	100% steel
	Body panels	100% steel
	Glass	100% glass
	Exterior trim	93.6% plastic, 4.3% steel, 1.5% rubber, 0.6% organic
	Body sealers/deadeners	100% rubber
	Door module	65.3% plastic, 32.6% organic, 1.8% steel, 0.3% glass
	Seating and restraint system	58% steel, 39% plastic, 3% organic
	Heating, Ventilation, Air conditioning (HVAC) module	56.2% steel, 21.5% wrought Al, 16.7% copper, 2.4% plastic, 2% rubber, 0.5% zinc, 0.7% other
	Interior electronics	59% plastic, 41% copper
	Others	/

Table 5 continued

Vehicle system	Components	Materials
Powertrain System	Engine unit	50% cast iron, 30% cast Al, 10% steel, 4.5% plastic, 4.5% rubber, 1% copper
	Fuel cell stack	62.8% carbon fiber composite, 23.2% wrought Al, 5.4% PFSAa, 5.0% carbon paper, 1.5% steel, 1.4% PTFEa, 0.6% carbon/PFSAa suspension, 0.1% platinum
	Powertrain thermal system	50% steel, 50% plastic
	Exhaust system	99.985% steel, 0.015% platinum
	Others	/
Transmission system	Transmission unit	30% steel, 30% wrought Al, 30% cast iron, 5% plastic, 5% rubber
Chassis System	Cradle	100% steel
	Driveshaft/axle	100% steel
	Braking system	60% iron, 35% steel, 5% friction material
	Wheels	100% steel
	Steering system	80% steel, 15% wrought Al, 5% rubber
	Others	/
Batteries	Pb–Ac battery	69.0% lead, 14.1% water, 7.9% sulfuric acid, 6.1% plastic, 2.1% fiberglass, 0.8% other
Fluids (excluding fuel)	Engine oil, Power steering fluid, Brake fluid, Transmission fluid, Powertrain coolant, Windshield fluid, Adhesives	

References

Argonne National Laboratory (ANL) (2015) The Greenhouse Gases, regulated emissions, and energy use in transportation model. Chicago University, U.S. greet.es.anl.gov. Accessed March 2016

Bauer C, Hofer J, Althaus HJ, Duce AD, Simons A (2015) The environmental performance of current and future passenger vehicles: life cycle assessment based on a novel scenario analysis framework. Appl Energy 157:871–883

Burnham A (2012) Updated vehicle specifications in the GREET vehicle-cycle model. Chicago University, U.S. greet.es.anl.gov/publication-update-veh-specs. Accessed March 2016

Chen YS (2014) A study on life cycle ecological benefits assessment of automotive parts (unpublished publicly). Ph.D. Academic Dissertation, Hunan University, China, pp 42–48

China Association of Automobile Manufacturers (CAAM) (2016) China Automotive Industry Yearbook 2000–2015. Beijing, China

Chinese Government (2015) Enhanced actions on climate change: China's intended nationally determined contributions. Beijing, China. www.sdpc.gov.cn/xwzx/xwfb/201506/t20150630_710204.html. Accessed March 2016

Chinese State Council (2012) Planning for the development of the energy-saving and new energy automobile industry (2012–2020). Beijing, China. www.gov.cn/zwgk/2012-07/09/content_2179032.htm. Accessed September 2016

Das S (2000) The life-cycle impacts of aluminum body-in-white automotive material. J Miner Metals Mater Soc 52(8):41–44

Das S (2004) A comparative assessment of alternative powertrains and body-in-white materials for advanced technology vehicles. SAE Technical Paper. 01-0573

Dhingra R, Das S (2014) Life cycle energy and environmental evaluation of downsized vs. lightweight material automotive engines. J Clean Prod 85:347–358

Editorial Board of China Steel Yearbook (2015) China steel yearbook 2015. Metallurgical Industry Press, Beijing

Greenhouse Gas Protocol (2015) What is the difference between direct and indirect emissions. Washington DC, U.S. www.ghgprotocol.org/calculation-tools/faq. Accessed March 2016

Hao H, Wang H, Yi R (2011a) Hybrid modeling of China's vehicle ownership and projection through 2050. Energy 36:1351–1361

Hao H, Wang H, Ouyang M (2011b) Fuel conservation and GHG (Greenhouse gas) emissions mitigation scenarios for China's passenger vehicle fleet. Energy 36(11):6520–6528

Hao H, Geng Y, Wang H, Ouyang M (2014) Regional disparity of urban passenger transport associated GHG emissions in China: a review. Energy 69:783–793

Hao H, Geng Y, Hang W (2016a) GHG emissions from primary aluminum production in China: regional disparity and policy implications. Appl Energy 166:264–272

Hao H, Geng Y, Sarkis J (2016b) Carbon footprint of global passenger cars: scenarios through 2050. Energy 101:121–131

Hawkins TR, Singh B, Majeau-Bettez G, Strømman AH (2013) Comparative environmental life cycle assessment of conventional and electric vehicles. J Ind Ecol 17(1):53–64

Howell S, Lee H, Heal A (2014) Leapfrogging or stalling out? Electric vehicles in China. HKS Working Paper No. RWP14-035. Harvard University, U.S. ssrn.com/abstract=2493131. Accessed March 2016

Intergovernmental Panel on Climate Change (IPCC) (2006) 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Institute for Global Environmental Strategies, Hayama, Japan. www.ipcc-nggip.iges.or.jp/public/2006gl. Accessed March 2016

Internation Energy Agency (IEA) (2015) CO_2 emissions from fuel combustion. Paris, France. www.iea.org/statistics/relateddatabases/co2emissionsfromfuelcombustion/. Accessed March 2016

Jing R, Cheng JCP, Gan VJL, Woon KS, Lo IMC (2014) Comparison of greenhouse gas emission accounting methods for steel production in China. J Clean Prod 83:165–172

Kim HJ, McMillan C, Keoleian GA, Skerlos SJ (2010) Greenhouse gas emissions payback for lightweighted vehicles using aluminum and high-strength steel. J Ind Ecol 14(6):929–946

Lewis AM, Kelly JC, Keoleian GA (2014) Vehicle lightweighting vs. electrification: life cycle energy and GHG emissions results for diverse powertrain vehicles. Appl Energy 126:13–20

Li Y, Zhu L (2014) Cost of energy saving and CO_2 emissions reduction in China's iron and steel sector. Appl Energy 130:603–616

Lin B, Omoju OE, Okonkwo JU (2016) Factors influencing renewable electricity consumption in China. Renew Sustain Energy Rev 55:687–696

Ma C, Li S, Ge Q (2014) Greenhouse gas emission factors for grid electricity for Chinese provinces. Resour Sci 36(5):1005–1012

Mahdizadeh KM, Gholami SMA, Valipour M (2015) Simulation of open-and closed-end border irrigation systems using SIRMOD. Arch Agron Soil Sci 61(7):929–941

McKenzie EC, Durango-Cohen PL (2012) Environmental life-cycle assessment of transit buses with alternative fuel technology. Transp Res Part D Transp Environ 17(1):39–47

Nanaki EA, Koroneos CJ (2013) Comparative economic and environmental analysis of conventional, hybrid and electric vehicles—the case study of Greece. J Clean Prod 53:261–266

National Bureau of Statistics of China (NBSC) (2016) Annual provincial power generation. Beijing, China. data.stats.gov.cn/easyquery.htm?cn=C01. Accessed May 2016

National Development and Reform Commission (NDRC) (2014) Chinese low carbon development and training materials of listing of provincial greenhouse gas. Beijing, China. qhs.ndrc.gov.cn/gzdt/201403/t20140328_604827.html. Accessed March 2016

National Development and Reform Commission (NDRC) (2015) 2014 Chinese Regional power grid baseline emission factor. Beijing, China. cdm.ccchina.gov.cn/zyDetail.aspx?newsId=52507&TId = 160. Accessed March 2016

Organisation Internationale des Constructeurs d'Automobiles (OICA) (2016) Global vehicle production. Paris, France. www.oica.net/. Accessed March 2016

Orsi F, Muratori M, Rocco M, Colombo E, Rizzoni G (2016) A multi-dimensional well-to-wheels analysis of passenger vehicles in different regions: primary energy consumption, CO_2 emissions, and economic cost. Appl Energy 169:197–209

Serrenho AC, Mourão ZS, Norman J, Cullen JM, Allwood JM (2016) The influence of UK emissions reduction targets on the emissions of the global steel industry. Resour Conserv Recycl 107:174–184

Sullivan JL, Burnham A, Wang M (2010) Energy-consumption and carbon-emission analysis of vehicle and component manufacturing. Argonne National Laboratory (ANL)

Sullivan JL, Burnham A, Wang M (2013) Model for the part manufacturing and vehicle assembly component of the vehicle life cycle inventory. J Ind Ecol 17(1):143–153

Valipour M (2012a) Sprinkle and trickle irrigation system design using tapered pipes for pressure loss adjusting. J Agric Sci 4(12):125

Valipour M (2012b) Comparison of surface irrigation simulation models: full hydrodynamic, zero inertia, kinematic wave. J Agric Sci 4(12):68

Valipour M (2016) Optimization of neural networks for precipitation analysis in a humid region to detect drought and wet year alarms. Meteorol Appl 23(1):91–100

Valipour M, Sefidkouhi MAG, Eslamian S (2015) Surface irrigation simulation models: a review. Int J Hydrol Sci Technol 5(1):51–70

Wang K, Wang C, Lu X, Chen J (2007) Scenario analysis on CO_2 emissions reduction potential in China's iron and steel industry. Energy Policy 35(4):2320–2335

Wang D, Zamel N, Jiao K, Zhou Y, Yu S, Du Q (2013) Life cycle analysis of internal combustion engine, electric and fuel cell vehicles for China. Energy 59:402–412

Weng XH (2009) Research about the problems exist in the statistics of comprehensive energy consumption per unit product of coke in China. Metall Econ Manag 2009(4):22–26

Wübbeke J, Heroth T (2014) Challenges and political solutions for steel recycling in China. Resour Conserv Recycl 87:1–7

Xia X, Li J, Tian H, Zhou Z, Li H, Tian G (2016) The construction and cost-benefit analysis of end-of-life vehicle disassembly plant: a typical case in China. Clean Technol Environ Policy. doi:10.1007/s10098-016-1185-0

Yang N, Huang Q, He L, You X, Wu L (2010) Domestic and international status and development trend of steelmaking compact process. China Metall 4:005

Yang D, Jingru L, Jianxin Y, Ding N (2014) Carbon footprint analysis of radial tyres of passenger cars. Chin J Popul Resour Environ 24:110–114

Yannopoulos SI, Lyberatos G, Theodossiou N, Li W, Valipour M, Tamburrino A (2015) Evolution of water lifting devices (pumps) over the centuries worldwide. Water 7(9):5031–5060

Zamel N, Li X (2006) Life cycle analysis of vehicles powered by a fuel cell and by internal combustion engine for Canada. J Power Sour 155(2):297–310

Zhao F, Hao H, Liu Z (2016) Technology strategy to meet China's 5 L/100 km fuel consumption target for passenger vehicles in 2020. Clean Technol Environ Policy 18(1):7–15