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Are the electric vehicles more sustainable than the conventional ones? Influences of the assumptions and modeling approaches in the case of typical cars in China

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ABSTRACT

In order to make the sustainable development of transport system, China has taken actions toward the electrification transition of vehicles. However, whether the electric vehicles (EVs) are more environmentally friendly than the conventional internal combustion engine vehicles (ICEVs) in China is still not clear due to a lack of complete and consistent environmental impact comparison of vehicles. This study takes the vehicle models from BYD Qin Pro series in China as illustrations to compare the environmental impact of battery electric vehicle (BEV) and plug-in hybrid electric vehicle (PHEV) with the conventional ICEV. The environmental profiles of vehicles across the whole life cycle within a set of environmental indicators are analyzed. The key aspects that may heavily influence the environmental impacts, i.e. regional heterogeneity, technology improvement and different modeling methods (attributional vs consequential) choices, are further tested, respectively. The results show that already today the BEV and PHEV powered by the current average Chinese electricity mix offer 23% and 17% reduction of global warming potential (GWP), respectively, compared to the gasoline ICEV. But it is achieved at the expense of considerable increases in mineral resource scarcity and ecological and human toxicity, etc. All of the factors considered show markedly influences on the environmental profiles of EVs, even up to 51% of GWP differences for different modeling methods application. And they all have the possibility to reverse the environmental priorities of some impact categories among ICEV, BEV and PHEV.

1. Introduction

With the requirement of energy saving and environmental protection, electric vehicles (EVs) have been promoted vigorously by many countries in recent years and become important parts of the modern transportation system (EC, 2017; METI, 2018). In order to reach the goal of the Paris Climate Agreement that peak the national carbon dioxide emissions before 2030, the Chinese government has released a series of policies to prioritize the promotion of new energy vehicles (NEVs), including battery electric vehicles (BEVs), plug-in hybrid electric vehicles (PHEVs), and fuel cell electric vehicles (FCVs) to reduce the emissions from automotive industry (MIIT, 2017; MOF, 2019). As a result, NEVs in China grow rapidly in numbers in recent years and makes China the largest market for NEVs. According to the statistics, the total NEV

sales volume in China in 2018 was above 1 million, along with 2.61 million of NEV stock (NMPSC, 2019). Among all kinds of NEV stock, BEVs account for more than 80%. In addition, the Chinese government has set a goal to reach 5 million of cumulative output of BEVs and PHEVs in 2020 (CSC, 2012), and the target in China is to reach 30% of new vehicle sales as NEV by 2030 (IEA, 2019).

In the context of rapid development of EVs and transition to electric mobility, whether the EVs outperform the conventional internal combustion engine vehicles (ICEVs) from the environmental perspective has attracted extensive attention. Many scholars have conducted the comparative environmental impact analysis of EVs and ICEVs. Kawamoto et al. (2019) compared the life cycle greenhouse gas (GHG) emissions of BEV and ICEV by using different electricity mix of US, European Union (EU), Japan, China, and Australia. They concluded that

in regions where renewable electric energy was widely used, the total GHG emissions of BEV were less than those of ICEV. Hawkins et al. (2012) and Lombardi et al. (2017) compared the life cycle environmental impact of EVs and ICEV under European context. The results indicated that EVs powered by the European electricity mix provided global warming potential (GWP) reduction potential relative to conventional diesel or gasoline vehicles, but showed potentials for significant increases in e.g. human toxicity, eutrophication, and metal depletion impacts. In the work of Egede et al. (2017), the environmental performance of EVs and conventional vehicles were compared under different regions. They demonstrated that concerning GWP, there were many regions in the world where BEVs were not performing better than ICEVs when considering the full life cycle. Bauer et al. (2015) conducted a comparative life cycle assessment (LCA) of different kinds of vehicles under both current and future technologies scenarios. They found that EVs under 2030 scenarios had the potential to reduce their environmental footprint, given that more non-fossil energy was applied for electricity production. Under the Chinese context, Qiao et al. (2019) performed a comparative analysis of the life cycle GHG emissions of BEV with ICEV, adapting key life cycle inventor (LCI) data for vehicle manufacturing and operation according to the actual situation in China. They indicated that BEV had lower GHG emissions than ICEV. Wu et al. (2018) compared the life cycle GHG emissions of BEVs and ICEVs under 2010, 2014 and 2020 scenarios considering future changes of electricity generation technologies and mix. The results indicated that BEVs showed gradual reduction of the total life cycle GHG emissions in 2020, relative to ICEVs. Shen et al. (2019) and Wu et al. (2019) explored the effect of regional electricity mix on the GHG emissions of BEVs. Both studies indicated that BEVs exhibited substantial GHG mitigation benefit in comparison to ICEVs in southern provinces in China. In the work of Shi et al. (2019) and Li et al. (2019), the emissions of BEVs relative to air pollution, such as SO_2, NO_x, primary $PM_{2.5}$ and PM_{10} were analyzed and compared with the ICEVs. The results showed that for current situation (2015), BEVs held obvious advantages in CO_2, VOCs, CO, NO_X and $PM_{2.5}$ emissions reduction, while the PM_{10} emissions of BEVs were higher than those of conventional ICEVs (Shi et al., 2019).

Existing research is valuable and helps to understand the environmental impact of EVs. However, the comparative environmental profiles of EVs with ICEVs is not analyzed sufficiently and consistently. Most of the available research fails to compare EVs with their counterparts with comparable size, weight, traction power, etc. and in some cases, vehicles with different years of technologies are even compared (Cerdas et al., 2017; Lombardi et al., 2017). This may lead to misleading results in the evaluation of the impact reduction potential of EVs. In addition, GHG emissions are usually the only impact category considered, which may ignore the potential burden-shifting to other environmental impacts. Furthermore, the factors that may heavily influence the environmental impact comparison, such as regional heterogeneity, technology improvement and different modeling methods choices are not addressed sufficiently. Although the heterogeneity in the regional grids and climatic conditions might have profound effect on the environmental impact of vehicles, as indicated by Shen et al. (2019) and Wu et al. (2019), they are not considered simultaneously in previous studies. Moreover, as vehicle technologies are under rapid development, such as lightweighting and energy efficiency improvement, the environmental performance of EVs as well as ICEVs is expected to improve significantly in the coming years. Nevertheless, a foresight comparative perspective concerning the future technological progress of vehicles, as well as electricity mix improvement is not applied consistently in previous studies. Regarding the modeling method choice in LCA, attributional one using average energy mix is commonly applied in the previous vehicle comparative studies (Bauer et al., 2015; Hawkins et al., 2012; Qiao et al., 2019). It normally considers a product system as isolated from the rest of the technosphere or economy and addresses what environmental impact is attributed to the product. However, all product systems are interacted more or less, and for large scale system changes

this becomes a problem (Hauschild et al., 2017). If the vehicle comparisons should serve for decision making on the future introduction of EVs in China, consequential method using long-term marginal electricity mix needs to be used consistently as new electricity production capacity would be installed in the long run in response to the projected demand growth of electricity as a result of the large-scale adoption of EVs. Unfortunately, the impacts arising from those choices are rarely reported.

To address these research gaps, three advanced and best-selling Chinese vehicle models from BYD Qin Pro series in China, i.e. gasoline based ICEV, PHEV, and BEV are taken as illustrations to reveal the comparative environmental profiles of vehicles. The whole life cycle with a set of environmental indicators are incorporated in the environmental analysis. The effect of regional heterogeneity, technology improvement of vehicles, as well as electricity mix and different modeling methodology (attributional vs consequential) are further analyzed, respectively. Based on these, the key points in the comparative assessment of vehicles are highlighted and corresponding recommendations are given.

2. Materials and methods

2.1. Goal and scope definition

To compare the life cycle environmental impact of the EVs i.e. BEV and PHEV with the conventional ICEV, three "A" size segment (these would be "C" segments in EU) vehicle models of BYD Qin Pro-series, named Qin Pro, Qin Pro DM and Qin EV500 are selected as the representative products considering their high market share in China. The technical parameters of the selected vehicles are shown in Table 1. Same sizes and technology levels with similar output power make them comparable. The main differences of the three vehicles lie in the powertrain systems, where fuel tanks, ICE, electronic combustion control systems, etc. of ICEV are replaced with lithium nickel manganese cobalt oxide (NMC) Li-ion battery with its temperature control system, electric motor and electrical system controls, etc. of BEV, while PHEV owns all of the mentioned parts in its powertrain system. The mass of PHEV and BEV is a little larger than the ICEV mainly due to the additional configuration of battery and its accessories. The lifetime mileages of vehicles and batteries are assumed as 150,000 km in the baseline according to the mileage assurance of BYD. PHEV carries an NMC battery with 14.38 kWh capacity enabling to provide an 82 km of all-electric driving and also a combustion engine to extend its driving range.

Table 1
Technical parameters of the selected vehicles in China.

Type	ICEV	PHEVs	BEVs
Model name	*BYD Qin Pro*	*BYD Qin Pro DM*	*BYD Qin EV500*
L/W/H (mm)	4765 × 1837 × 1500	4765 × 1837 × 1495	4765 × 1837 × 1515
Wheel base (mm)	2718	2718	2718
Vehicle curb weight (kg)	1380	1690	1650
Engine maximum power (kW)	118	118	-
Engine maximum torque (N. m)	245	245	-
Motor maximum power (kW)	-	110	120
Motor maximum torque (N. m)	-	250	280
Fuel tank capacity (L)	50	39	-
Fuel use (L/100 km)	6.5	4.3	-
Battery capacity (kWh)	-	14.38	56.4
Range in electric mode (km)	-	82	420
Electricity use in electric mode (kWh/100 km)	-	17.5	13.4
Vehicle/battery lifetime (km)	150,000	150,000	150,000

Followed by Faria et al. (2012), the use scenario considered for PHEV is assumed as 85% in electric mode (every week of commute) and 15% in range extended mode (such as longer trips at weekends). The calculation of energy and fuel consumption of PHEV is detailed in Supporting Information (SI) (Note 1). Variations in use for commute or longer trips are tested in the sensitivity analysis.

To make the comparison reasonable and fair, the functional unit of this paper is defined as one kilometer (km) driven by vehicles under current Chinese standard test procedure. Fuel or electricity life cycle and vehicle life cycle are considered in this study, as shown in Fig. S1 in SI. Specifically, vehicle life cycle includes the extraction of raw materials, components manufacturing, vehicle assembly, maintenance and end-of life (EoL) treatment. Fuel or electricity life cycle includes the production and distribution of fuels or electricity i.e. well to tank (WTT) and the vehicle in-use phase (also known as tank to wheels (TTW)).

In the baseline of the vehicle comparison, an attributional modeling approach is applied. Average LCI data is used for background processes modeling and system expansion is adopted to deal with the multifunctionality of processes. Correspondingly, avoided burden i.e. EoL recycling (EOLR) approach is applied for the vehicles recycling accounting. With this approach, the recycling credits are calculated based on the potential rate of primary materials replaced (Atherton J, 2007). While the recycled content (RC) or cut-off approach, which assigns recycling credits based on the recycled material contents in materials used (Johnson et al., 2013), is tested in Section 4.1. Moreover, consequential modeling regarding the long-term marginal energy mix is additionally established and analyzed, as shown in Section 4.5. Product systems of the three vehicles are modelled in SimaPro, where Ecoinvent v.3.4 is used as a background data source within which most key parameters are adapted according to the Chinese real conditions. ReCiPe characterization method at midpoint level is applied for the impact assessment, considering eighteen impact categories, including GWP, stratospheric ozone depletion potential (SODP), ionizing radiation potential (IRP), ozone formation potential, human health (OFP-HH), fine particulate matter formation potential (PMFP), ozone formation potential, terrestrial ecosystems (OFP-TE), terrestrial acidification potential (TAP), freshwater eutrophication potential (FEUP), marine eutrophication potential (MEUP), terrestrial ecotoxicity potential (TEP), freshwater ecotoxicity potential (FEP); marine ecotoxicity potential (MEP), human carcinogenic toxicity potential (HCTP), human non-carcinogenic toxicity potential (HNCTP), land use (LU), mineral resource scarcity potential (MRSP), fossil resource scarcity potential (FRSP), water consumption (WC).

2.2. LCIs

2.2.1. Vehicle life cycle

As mentioned above, the vehicle product system includes material extraction, component production, vehicle assembly, maintenance and EoL disposal. LCIs collection is detailed as follows. Material composition of the vehicles are imported from the GREET models (ANL, 2019). To adapt the data, we have scaled the mass of each material based on the real mass of the investigated vehicles in this study, as shown in Table S1. Material content of Li-ion battery and its manufacturing emissions are taken from Majeau-Bettez et al. (2011), who established a transparent LCI of NMC battery in their study. Production of positive material of $LiNi_{0.4}Co_{0.2}Mn_{0.4}O_2$ is adjusted based on the emission data from one Chinese leading battery production enterprise (Xie et al., 2015), as shown in Table S2 in SI. The material composition of 12V lead-acid batteries of ICEV and PHEV is imported from GREET model, while that of iron phosphate lithium-ion (LFP) battery for BEV is taken from Majeau-Bettez et al. (2011). Primary materials of steel, aluminum and copper, etc. are applied in the modeling to avoid double counting when the EoL recycling method is adopted. The production of primary steel follows the blast furnace and basic oxygen furnace (BF-BOF) route, whose emissions is from Ecoinvent 3.4. It should be noted that, in

practice, if steel scrap is also an input of the steelmaking route, an extrapolation method (WSA, 2017) can be used to calculate the emissions of 100% primary steel production. The database for materials manufacturing in Europe is used, in which the main energy sources such as electricity and hard coal are replaced with Chinese data. The LCI modeling of electricity mix in China is described in detail (see Note 2 in SI).

Vehicle assembly involves painting, heating, air compressing, welding, material handling, ventilation and air conditioning (HVAC) and lighting (ANL, 2019). The energy consumption associated with these processes is incorporated in the LCA modeling and the corresponding data reported in the GREET model is used since it is not available in China, as shown in Table S3 in SI. Moreover, it is assumed that lead-acid battery, tires and fluids are replaced during the vehicle maintenance stage, as shown in Table S1.

For the EoL vehicles, they are pre-treated to remove the residual oil, airbags, refrigerants, etc. followed by the dismantling of batteries and tires for further recovering. The remaining part of the vehicles are mechanically shredded, which allows a part of iron and copper scraps to be picked directly. The automotive shredded residue (ASR) is treated by using magnetic and heavy media separation techniques to screen and collect additional iron, aluminum and copper scraps (Hao et al., 2017; Li et al., 2016). And then, these collected materials are used to produce secondary materials, which avoid impact from primary materials production. The impacts of recovery of the recycled materials is taken from Ecoinvent v3.4. Considering 90% of iron scraps go into basic oxygen furnace (BOF) process and the remaining go into electric arc furnace (EAF) process in China, the corresponding ratios of recovering steel by BOF and EAF processes are applied. More details for the calculation of EoL steel credit are shown in Fig. S2 in SI. It is worth noting that, in practice, the metals quality may degrade during the recycling process, due to the inclusion of impurities (Johnson et al., 2013). Replacing primary materials with recycled ones without considering quality degradation in this work would somehow exaggerate the recycling credits. Further recovery of plastic, glass, fine mineral and other nonmetallic materials is hard to achieve, because the most advanced techniques are required and are not likely to be applied by Chinese enterprises even in 2025 (Hao et al., 2017). Thus, they are treated by landfilling in this study. The detailed LCI of vehicle recycling provided by Hao et al. (2017) is applied in this study. Advanced hydrometallurgical technology enabling the recycling of both copper and aluminum is employed for NMC battery recycling considering it is the future trend of battery recycling (Xie et al., 2015). The corresponding inventory that is adapted in this study is shown in Table S4 in SI. The LFP battery recycling is based on Cheng (2019).

2.2.2. Fuel life cycle

Fuel life cycle of vehicles involves the production and distribution of fuels i.e. WTT and the in-use phase of fuels i.e. TTW. BEV does not produce exhaust emissions, while ICEV and PHEV do due to fuel combustion. The amount of electricity consumption of BEV and PHEV, as shown in Table 1, is based on the Chinese standard test procedure, same as the New European Driving Cycle (NEDC). An average charging efficiency of 90% is adopted to account for the electricity losses during charging the battery (Canals Casals et al., 2016; Qiao et al., 2019). The electricity emission intensity of the average electricity mix in 2019 is developed based on the Chinese statistics, detailed in SI (Note 2). The default emission data of gasoline production comes from Ecoinvent, where the energy-related data is adapted to Chinese conditions. The ICEV studied in this paper follows the China V emission standard, which is equivalent to the Euro 5 standard. Therefore, the exhaust emissions during ICEV operation are scaled based on the emission database of Euro 5 mid-size gasoline ICEV vehicles in Ecoinvent.

The non-exhaust emissions of the three vehicles from tire, road and brake wear are also considered. Considering the brake wear of BEV tends to be lower than ICEV due to its regenerative brake, a conservative

estimate of zero brake wear emissions for BEV is assumed (Timmers and Achten. 2016). While the rest of non-exhaust emissions are assumed as the same with the ICEV, which come from the default data from Ecoinvent.

3. Results

3.1. Overview of the life cycle environmental impact comparison

Fig. 1 shows the comparative LCA results of the three vehicle technologies for all the impact categories included in the ReCiPe 2016 Midpoint methodology. The impacts for each impact category are normalized to the highest impact in each category. Vehicle life cycles are segmented into several stages, i.e. vehicle production, operation, maintenance, and EoL. Regarding the significant contribution of battery on the impact of vehicle production and EoL, it is displayed individually. Vehicle operation is composed of WTT and TTW, which together represent the contribution of the fuel life cycle. Detailed numerical results for each category of the three vehicles are provided in the Table S8-S10 in SI.

BEV shows reduction potential for GWP, SODP, OFP, TAP and FRSP relative to ICEV with the average Chinese electricity mix in China, primarily due to it offers advantages on the corresponding emissions in fuel life cycle than ICEV. However, BEV and PHEV show potentials for burden shifting especially for IRP, FEUP, MRSP and all the impact categories associated with toxicity. The nuclear energy-based power used

in WTT stage and the vehicle especially battery production of BEV and PHEV are the main drivers. The total environmental impact of PHEV always lies between BEV and ICEV except for TAP and HCTP. The detailed cause-effect analysis of these impact categories is shown in Section 3.3.

For all three vehicle technologies, the fuel life cycle dominates GWP, SODP, IRP, LU, and FRSP, while the remaining more than half of the impact scores are caused primarily by the activities associated with the production of the vehicles including batteries. Maintenance accounts for a tiny part of the total impact for all vehicle types and impact categories.

The production of BEV and PHEV is more environmentally intensive than that of ICEV for all impact categories, mainly due to battery production. Even without the battery, BEV production shows higher impacts than the ICEV, due to the use of more environmentally intensive materials such as copper and aluminum in the BEV powertrain. For the impact categories where BEV and PHEV are superior to the ICEV, like GWP, lower operation emissions caused by the fuel life cycle offset the additional burden of vehicle production.

Recycling of vehicles can to some extent reduce the environmental impact of the vehicle production. When the recycling crediting of the vehicles is accounted for, the impact of vehicles production is partially offset as shown in Fig. S3 in SI.

3.2. GWP

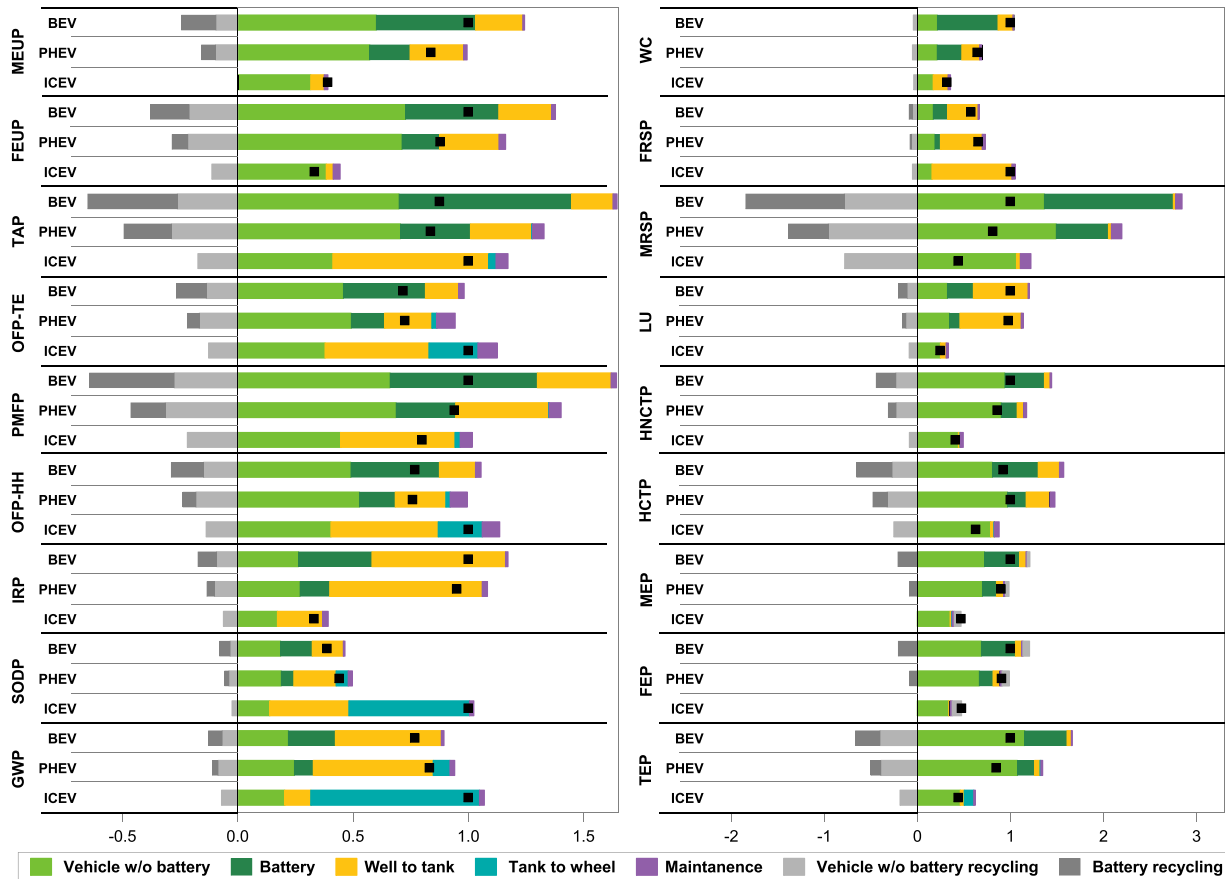Fig. 2 shows the life cycle GWP of the three vehicle technologies in

Fig. 1. Impact scores for 1 km driving with each of the three vehicle technologies. Results have been normalised to the highest total impact for each category. Black dots represent total impact by adding positive and negative impact contributions. Global warming potential (GWP), stratospheric ozone depletion potential (SODP), ionizing radiation potential (IRP), ozone formation potential, human health (OFP-HH), fine particulate matter formation potential (PMFP), ozone formation potential, terrestrial ecosystems (OFP-TE), terrestrial acidification potential (TAP), freshwater eutrophication potential (FEUP), marine eutrophication potential (MEUP), terrestrial ecotoxicity potential (TEP), freshwater ecotoxicity potential (FEP); marine ecotoxicity potential (MEP), human carcinogenic toxicity potential (HCTP), human non-carcinogenic toxicity potential (HNCTP), land use (LU), mineral resource scarcity potential (MRSP), fossil resource scarcity potential (FRSP), water consumption (WC).
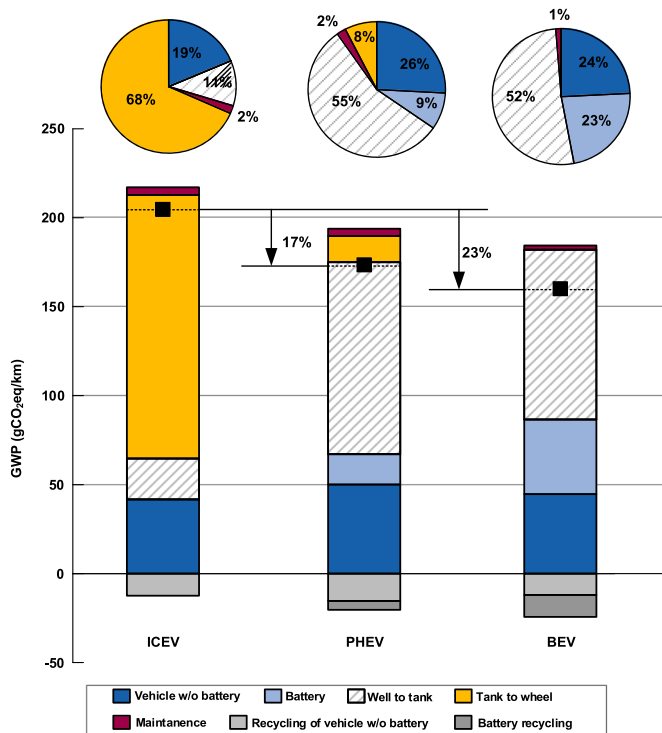
Fig. 2. Life cycle GWP per kilometer of the three vehicle technologies.

terms of g CO_2eq per kilometer. Based on the present average Chinese electricity mix the life cycle GWP of BEV and PHEV are 156 and 169 g CO_2eq/km, respectively, with 23% and 17% of GWP reduction compared with ICEV. Vehicle operation emissions are the main contributors of the life cycle GWP for all the three vehicles. Specifically, they account for 79%, 63%, and 52% of the total GWP of ICEV, PHEV, and BEV, respectively. The tailpipe GHG emissions in TTW stage dominates the GWP of the ICEV, whereas WTT associated with coal-based electricity production emissions dominates GWP for BEV and PHEV. Emissions from vehicle production are another hotspot for the BEV, accounting for nearly half of its life cycle GWP. It is estimated to be 86 g CO_2eq/km with the selected vehicle technologies, which is twice the 41 g CO_2eq/km of ICEV production. Although the GWP of vehicle production of BEV and PHEV is higher, it is compensated by the low carbon emissions during vehicle operation.

The total GWP crediting of EoL BEV recycling is 26 g CO_2eq/km. If the recycling crediting is integrated i.e. the recovered materials are applied for the vehicle manufacturing, the GWP of BEV production decreases to 60 g CO_2eq/km. In such a context, the proportion of BEV production emissions is reduced to 39% of the total life cycle GWP.

3.3. Other impact categories

As shown in Fig. 1, BEV and PHEV outperform the ICEV in terms of the SODP. The CFC emissions from Chinese hard coal-based electricity production and gasoline production are the dominators of BEV and ICEV, respectively, during their TTW stages. In the production stage, polyurethane and copper production are the main sources of SODP-related emissions.

More than half of life cycle IRP of BEV is caused by the use stage consumption of nuclear electricity, while the radioactive waste treatment during gasoline production is the major contributor for ICEV. Considering that 4.8% of current Chinese electricity comes from nuclear, the larger fraction of electricity use during the BEV and PHEV operation causes their IRP to be greater than that for the ICEV.

For BEV and PHEV, the vehicle life cycle dominates the impacts in

PMFP, OFP-HH, OFP-TE, TAP, FEUP and MEUP, while the fuel life cycle especially for the WTT stage dominates those of ICEV (except for FEUP and MEUP). The impacts of BEV and PHEV production can be traced back to the copper production, while waste natural gas, heavy fuel oil burning, etc. during the refinery operation of gasoline are the main sources to the use stage emissions of the ICEV. Among all of these impact categories, BEV and PHEV do not show benefit in PMFP, FEUP and MEUP.

The toxicity-related impacts (TEP, FEP, MEP, HCTP, HNCTP) follow the same trend, as shown in Fig. 1. For the BEV, it is are generally more than 200% of the ICEV scores, which shows a potential for significant burden shifting towards toxicity for BEV. Vehicle production emissions dominate the life cycle toxicity impact for all three types of vehicles. A further process contribution analysis identifies the disposal treatment of the sulfidic mine tailings related to the copper production chain as the hotspot of BEV. The remaining impact is mainly caused by the disposal treatment of spoils from hard coal mining for hard coal-based electricity production. The additional consumption of copper for the electric drivetrain of BEV, as well as the battery production, causes its greater toxicity impact than ICEV. The freshwater and marine eutrophication of BEV are also dominated by the vehicle production stage, and again the source is copper production and hard coal-based electricity processing.

MRSP is another category that potentially brings burden-shifting for BEV application. The additional consumption of metals such as copper and steel for BEV production leads to an MPSP score, which is more than 2 times of that of the ICEV. As regards to FRSP, BEV outperforms the ICEV with the present Chinese electricity mix. It can be expected that more clean energy-based electricity use will lead to significant FRSP mitigation for the BEV.

4. Discussions

4.1. Sensitivity and uncertainty analysis

This section aims to understand the robustness of the results against the uncertain parameters associated with the assumptions in this work. In the LCI modeling, material content of the vehicles is taken from the GREET model, which is expected to produce a certain amount of discrepancy. Hence, a sensitivity analysis is conducted, where the material composition of BEV and ICEV based on Golf 6 from Fabienne (2011), as shown in Table S5 in SI, is used as a counterpart to test the sensitivity. Furthermore, energy consumed by battery manufacturing and recycling in the baseline scenario from Xie et al. (2015) represents the corresponding levels of 2015 in China. Decreases of energy consumption during battery manufacturing and recycling by 10% are considered, respectively. Lifetime mileages of vehicle and battery are two other uncertainty sources. A 10% increase of the vehicle lifetime is analyzed regarding the extended lifetime of the vehicles in China while a 10% decrease of battery lifetime is tested considering the addition burden from possible battery replacement. Due to the life cycle electricity and fuel consumption of the PHEV are highly dependent on the use scenarios. The sensitivity analysis in terms of a 10% longer or 10% shorter trips for commuting is analyzed. Moreover, the sensitivity of the application of recycled content approach is also tested. Overall, all the parameters of the three vehicles involved in the sensitivity analysis are summarized in Table 2.

Fig. S4-S6 in SI show the sensitivity analysis results of all the uncertain parameters considered. The changes of almost all of the impact categories due to the variations of the considered parameters lie within a 10% interval, except that the application of RC method leads to the relative variations of several impact categories of the three vehicles exceed 20% and even more than 150% for MRSP. It is mainly because the average recycled content of materials in China is at a very low level (recycled content contained in steel, aluminum and copper only accounts for 20%, 16% and 26%, respectively (MOC, 2019)), which is far lower than the recycling rates accounted in the EOLR method. The LCA

Table 2

Uncertain parameters considered in the sensitivity analysis for the three technologies of vehicles.

Uncertain parameter considered	Baseline	Sensitivity analysis compared to baseline		
		ICEV	PHEV	BEV
Vehicle material proportions	Taken from GREET model, as shown in Table 2	Fabienne (2011)[1]	-	Fabienne (2011)[1]
Energy consumed by battery manufacturing	As shown in Table S2 in SI	-	− 10%[2]	− 10%[2]
Energy consumed by battery recycling	As shown in Table S3 in SI	-	− 10%[2]	− 10%[2]
Vehicle life time (km)	150,000	+ 10%	+ 10%	+ 10%
Battery life time (km)	150,000	-	− 10%[3]	− 10%[3]
Ratio of longer trips	15%	-	± 10%	-
Different recycling counting methods	EoL recycling	Recycled content	Recycled content	Recycled content

Note:

[1] The vehicle material proportions for ICEV and BEV taken from Fabienne (2011) are shown in Table S5 in SI, respectively.

[2] The energy consumed by battery manufacturing and recycling after they are decreased by 10% is shown in Table S6 and S7 in SI.

[3] 10% reduction of battery life time means 10% increase of battery mass.

results of the three vehicles using the RC recycling method are further shown in Fig. S7. Overall, for all of the tested changes, the environmental advantages of BEV and PHEV over ICEV is not changed, supporting their robustness. However, the application of RC recycling method can reverse the comparative results of some impact categories between BEV and PHEV, thus it needs to be treated properly in practice (Johnson et al., 2013).

4.2. Benchmarking against other studies

The comparative results of vehicle GWP obtained in this study together with other results from the literature is shown in Table S11 in SI. The life cycle GWP per kilometer of the three vehicles in our study is generally lower than their counterparts in the previous studies. Differences in key technical parameters of vehicles, electricity mix and application of different EoL recycling accounting methods might be the drivers of these discrepancies.

Other impact categories are rarely reported for vehicle life cycle environmental impact assessment. Bauer et al. (2015), Bohnes et al. (2017) and Hawkins et al. (2012) incorporated the analysis of terrestrial acidification, human and ecological toxicity, and mineral resource depletion, etc. into their work. Due to the differences in life cycle impact assessment (LCIA) methodologies and the fact that only the internally normalised LCA results are given in these papers, they cannot be used for the direct benchmarking of impact scores. In these studies, BEV showed potential burden-shifting to freshwater and marine eutrophication, human and ecological toxicity, and mineral resource depletion compared to ICEV. This is aligned with the findings in this work.

4.3. Influences of regional grids and climate conditions

The environmental impact comparison of the three vehicles in the baseline is based on the average Chinese electricity mix under the standard ambient temperature at 20°C. However, in practice, there is a large difference in the environmental burdens of electricity generation in different regions in China, which makes the environment impact of EVs charging in different regions a big difference (Shen et al., 2019). Moreover, the climatic conditions across China varies significantly.

Vehicles in frigid locations such as northern China are likely to have greater life cycle burdens (e.g. for heating the vehicles, and for lower efficiency of battery systems) than vehicles driven in temperate locations (Wu et al., 2019). This section, thus, addresses how the heterogeneity in grids and climate conditions might affect the results. Six regional grids in China, including north, northeast, northwest, central, east and south grids, are considered. To reveal the upper limits of influences caused by regional climate conditions, the situation in Harbin, one of the coldest cities in China, is analyzed. In this paper, we follow Wu et al. (2019) to calculate the relative variations of fuel consumption rate of the three vehicles under the ambient temperature of Harbin with their baseline fuel consumption rate.

Fig. S8 shows the environmental impact variations of charging BEV and PHEV with different regional grids. It can be seen that north grid offers maximum increase of GHG emissions (21% for both BEV and PHEV), which makes PHEV no longer superior to ICEV in GWP. While the EVs charged by south grid provides 18 % GWP reduction, at the expenses of increasement of IRP, etc. More fossil fuel-based electricity generated in north gird, as shown in Fig. S9, makes its GWP larger than that of other grids in China. Fig. S10 shows the environmental impact variations of the three vehicles under the ambient temperature of Harbin. Due to the total fuel and electricity consumption rates of vehicles increase not that much in cold regions in China (20% and 6% increases for fuel and electricity consumption rates, respectively), the relative variations are negligible when all life cycle stages are counted (most of impact categories increase lower than 10%). Thus, the regional climate conditions do not change the comparative results of the three vehicles.

4.4. Future perspective on technology development

Due to the technology development, vehicle and battery weight, fuel and electricity use and electricity mix are expected to be reduced over time. Therefore, technology development over time is considered under the foresight perspective of 2030. Two scenarios are developed for each improvement. One is a moderate, i.e. business-as-usual scenario, which is based on the prediction of China's future automobile development under the current policies. The other one is an aggressive scenario. Improvements of vehicles and power system considered in this work is shown in Table 3. Specific consideration of improvement of each parameter is elaborated (see Note 3 in SI).

Fig. 3 shows the life cycle GWP per kilometer of the three vehicles under the 2030 scenario. The expected reduction of vehicle and battery mass, improvement of fuel economy and structural change of electric mix can mitigate the GWP of all of the three vehicles. Among all the improvement pathways, fuel use reduction will be the dominator for future GWP mitigation of all the three vehicles both in the moderate and aggressive scenarios. Compared to the current situation, the GWP is projected to be reduced by 15%, 18% and 16% for BEV, PHEV and ICEV, leading to life cycle GWPs as low as 132, 138 and 171 gCO_2eq/km, respectively, in the moderate scenario. In the aggressive scenario, the GWP of the three vehicle technologies will be further mitigated due to fuel use reduction by up to 125, 131 and 158 gCO_2eq/km for BEV, PHEV and ICEV respectively.

Compared to ICEV, significant GWP mitigation can be achieved by BEV and PHEV through the improvement of the electricity mix and reduction of battery mass. In each scenario for vehicle improvement in 2030, BEV retains the advantages in GWP over PHEV and ICEV. However, if the improvement of the three vehicles is not achieved simultaneously, for example, the reduction of electricity use of BEV is at standstill while the improvement of fuel economy of PHEV evolves, the comparative results will be reversed and PHEV will be the best choice regarding GWP reduction.

In terms of the toxicity-related impact categories, they are reduced in all the improvement scenarios in 2030 for all the three types of vehicles, as shown in Fig. S11 in SI. Reductions of vehicle and battery mass are the main contributors for their expected mitigation. However, in contrast

Table 3

The improved parameters associated with the process chains of the three vehicles in 2030.

Parameters	Baseline			Moderate			Aggressive			References
	BEV	PHEV	ICEV	BEV	PHEV	ICEV	BEV	PHEV	ICEV	
Vehicle mass w/o battery (kg)	1300	1548	1380	1160	1385	1238	1036	1239	1111	ACNPMCCS (2015)
Battery mass (kg)	350	142	-	140	57	-	112	45	-	As above
Energy consumption in use stage										
Fuel use (L/100 km)	-	4.3	6.5	-	3.3	5.3	-	3.3	4.8	CSAE (2016)
Elec. use in electric mode (kWh/100 km)	13.4	17.5	-	10	13	-	9	11.8	-	ACNPMCCS (2015)
Electricity mix	Average elec. mix in 2019			Average elec. mix in 2030-CPS (share)			Average elec. mix in 2030-NPS (share)			IEA (2018)
Coal-based thermal power	62.2%			54.1%			49.2%			
Natural Gas-based thermal power	3.2%			8.6%			7.6%	-		
Oil-based thermal power	0.1%			0.1%			0.1%			
Bioenergy	1.5%			2.5%			2.7%			
Nuclear Power	4.8%			7.3%			7.9%			
Hydropower	17.8%			13.8%			14.6%			
Wind Power	5.5%			7.5%			9.4%			
Solar Power	3.1%			5.9%			8.3%			
Others	2%			0.2%			0.3%			

Note: CPS denotes current policy scenario and NPS denotes new policy scenario.
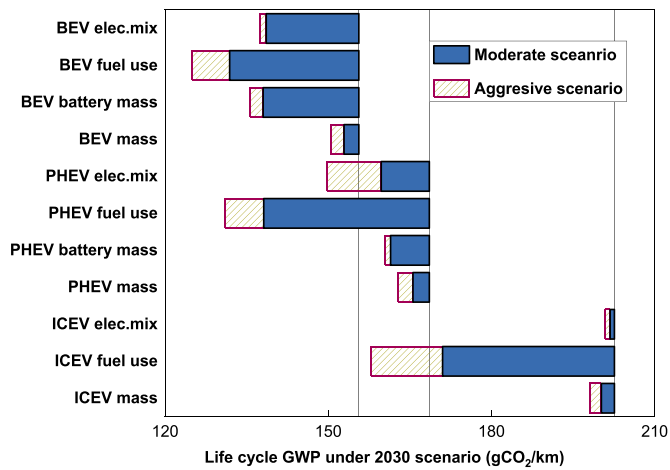
Fig. 3. Life cycle GWP per kilometer of the three vehicles under 2030 scenario. The reductions of GWP under moderate and aggressive scenarios in 2030 are shown relative to the baseline scenario of 2019.

with ICEV, BEV still shows inferior environmental performance in toxicity. All the other impact categories are also found to be improved under the different improvement scenarios in 2030 excepting IRP, MSRP and LU under the scenario of electricity mix change as a result of expansion of nuclear, wind and solar power in 2030.

4.5. Average vs long-term marginal electricity mix

Long-term marginal electricity mix reflects the projected changes in electricity capacity induced by the increased demand. It is composed of the energy sources with a growing production between the reference year and the time horizon (Vandepaer et al., 2018). In this paper, a temporal scope from 2019 to 2030 is considered. Therefore, the increased electricity capacity induced by EVs from 2019 to 2030 is regarded as long-term marginal electricity mix. Theoretically, the use of energy system models or energy simulation is the most accurate approach to catch the causal relations between the increased electricity demand of EVs and electricity capacity. However, its application is restricted by sophisticated modeling and difficulties in data acquisition (Vandepaer et al., 2018). The energy model with the future projection of electricity capacity from official release is an accessible way to get the data, and thus it is used in this study as a compromise. It should be noted that the method applied in this paper represents the average electricity capacity changes caused by various demand rather than those caused by

EVs adoption. Follow the 2030 electricity scenarios mentioned in Section 4.4, the long-term marginal electricity mix (2019-2030) are calculated, as shown in Table S17. The coal-based power is reduced sharply while clean energy power is increased significantly both under CPS and NPS scenarios. The comparative LCA results of the average (2019) and long-term marginal electricity mix (2019-2030) is shown in Table S18 in SI.

Fig. 4 shows the life cycle GWP per kilometer of the three vehicles with average and marginal electricity mix. Since greater proportion of clean energy is applied in marginal electricity scenario, the life cycle GWP of all the three vehicles is reduced compared to the average electricity scenario. Among them, BEV decreases the most in the two marginal scenarios, up to 119 (23% reduction) and 77 (51% reduction) gCO$_2$eq/km for CPS and NPS scenarios, respectively, mainly due to the significant reduction of GWP in WTT stage. As a result, it accordingly emits 40% and 61% lower GWP than ICEV. The LCA results for other impact categories are shown in Fig. S12 in SI. The impact categories such as FRSP, PMFP and FEUP are also decreased significantly because the proportions of coal-based power are decreased in the two marginal electricity mixes, while the IRP, MSRP and LU show the potentials for increasement due to the increased contribution from nuclear, wind and solar power in marginal electricity scenarios. Overall, the consequential modeling regarding the application of marginal electricity mix dose not reverse the comparative results of the three vehicles across all of the
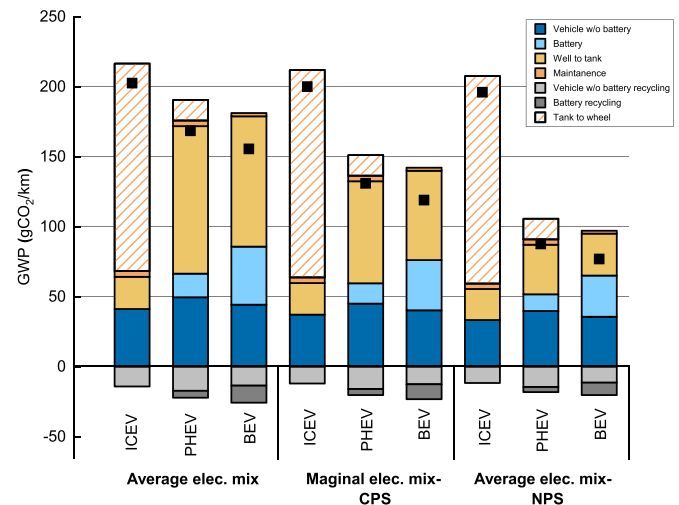
Fig. 4. Life cycle GWP per kilometer of the three vehicles with average (2019) and marginal (2019-2030) electricity mix.

impact categories (except for PMFP). The contribution of vehicle including battery production on the whole life cycle GWP of BEV and PHEV is further increased, in the extreme case (for BEV under NPS energy scenario), from 39% to 58%, which shifts the environmental hotspot from WTT to vehicle production stage.

4.6. Key points for comparative assessment of vehicles and decision-making

Complete and consistent comparative assessment of vehicles is the key to understanding the real environmental benefit of EVs and further supporting decision-making. Summarizing experience from this study, a number of key points with some recommendations are given for comparative assessment of different vehicle types.

Firstly, vehicles with comparable size, output power and technical performance are required for the comparison. It is important to minimize the differences in vehicle parameters except for the powertrains, so that the results can tell the differences caused only by the powertrains and reveal the real reduction potential of EVs. Furthermore, in addition to GWP, a broader array of impact categories should be considered to identify the potential burden shifting and most importantly, to support decision-making. In this study EVs outperform ICEV in GWP, at the expense of significant increases in IRP, freshwater and marine eutrophication, MRSP and all the impact associated with toxicity. Thus, whether EVs are the beneficial choices or not are strongly dependent on the weights of different impact categories set by decision-makers. For example, if the aim is to reduce the GWP, EVs could be more favorable choice than the ICEVs. While if more weights are given to ecotoxicity and mineral resource scarcity, etc., EVs are no longer competitive comparing with ICEVs.

Secondly, a foresight perspective considering the technology development of vehicles as well as the decarbonisation of power system needs to be considered in the comparison, in order to provide a comprehensive understanding of the environmental profiles of the vehicles. As discussed above, in 2030, the environmental profiles of the vehicles are expected to be improved a lot. BEV and PHEV have potentials to continually maintain their advantages over ICEV in terms of GWP. But the preference of BEV over PHEV in 2030 highly depends on the future technology promotion effect.

Last but not least, the environmental impact of vehicles shows markedly differences when different LCI modeling methodologies, i.e. attributional and consequential methods are applied. Although the comparative results across the impact categories (except for MRSP) by using the two modeling methods, regarding the application of average or long-term marginal electricity, are not changed, the relative impact reduction potential is totally different and, in some cases, the environmental hotspot is changed. Thus, the choice of LCI modeling methodologies needs to be dealt with carefully based on the specific decision context. In line with the International Life Cycle Data System (ILCD) guideline (EC-JRC, 2010), which is elaborated by European Commission to guide the LCA analysis, the consequential modeling is generally used for the decision-making in which the background production systems can be significantly influenced by the choice to be made. In the comparative LCA analysis of vehicles, the consequential LCI modeling is applicable if it is under the decision context of the future introduction of the EVs in large scale in China. Otherwise, the attributional LCI modeling is recommended.

5. Conclusions and outlook

This paper compares the life cycle environmental burden of "A" class BEV, PHEV and ICEV in China based on GREET model and key technical parameters of three BYD vehicles. The results show that BEV and PHEV powered by the current average Chinese electricity mix provide 23% and 17% reduction of GWP, respectively, as compared to the gasoline ICEV. The environmental benefit of BEV and PHEV relative to ICEV is

also seen in SODP and FRSP. However, BEV and PHEV present potentials for significant increases in IRP, FEUP, MEUP, MRSP and all the impact categories associated with toxicity, as a consequence of emissions along with vehicle, especially battery production and nuclear-based electricity production. The comparative results are further found to be heavily affected by regional heterogeneity, technology improvement and different modeling methods choices. North grid provides maximum increase of GHG emissions, which leads to PHEV no longer superior to ICEV in GWP. The projected improvements of vehicle technologies and electricity system to 2030 can further reduce the life cycle GWP of the three vehicles. BEV has great potentials to continually outperform ICEV in GWP in 2030 while its advantages over PHEV are highly dependent on whether its energy efficiency improvement keeps pace with that of PHEV. In addition, by using consequential modeling, specifically, long-term marginal electricity mix, the life cycle GWP of all the three vehicles is reduced. In the aggressive marginal scenario, BEV decreases the most, even up to 51% reduction. As a result, it accordingly emits 60% lower GWP than ICEV. The application of marginal electricity mix also offers the possibility to reverse the environmental priority of PMFP among vehicles. Therefore, the selection of different LCI modeling methods in comparative LCA study should be paid much attention and based on the specific contexts.

This paper provides an illustration for the environmental impact comparison of EVs and ICEVs, which offers a robust basis for the continuous improvement of the environmental impact of the current vehicles as well as the decision-making of future EVs deployment in China. The latter would require a change in focus from the functionality of one vehicle towards a fleet-based vehicle environmental burden analysis for China covering the whole vehicle market. This will be the topic of future work.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Supplementary materials

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