Cradle-to-gate greenhouse gas (GHG) burdens for aluminum and steel production and cradle-to-grave GHG benefits of vehicle lightweighting in China

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ABSTRACT

Greenhouse gas (GHG) burdens of steel and aluminum production and life cycle benefits of vehicle lightweighting in China were evaluated. Production of advanced high-strength steel (AHSS) and wrought aluminum (Al) have average cradle-to-gate GHG emissions of 3.9 and 17.5 kg CO₂eq/kg. Lightweighting benefits for eleven passenger car models over five driving cycles (including real-world and regulatory cycles) were determined. Lightweighting using AHSS to replace conventional steel has cradle-to-grave GHG savings in all cases, mainly attributed to savings in material use. Wrought Al has a much higher GHG production burden than AHSS and requires greater fuel savings in the use phase to achieve net cradle-to-grave GHG savings. Maximum GHG savings occur with Al versus AHSS in cases where the powertrain is resized, travel is congested, or lifetime travel distance is long. A typical Beijing peak-hour driving cycle with low speed and frequent stop-and-go has higher fuel reduction values (FRVs) and GHG savings than other cycles. Congested travel conditions make lightweighting a particularly effective emissions reduction strategy in China.

1. Introduction

Vehicle lightweighting is an important strategy to improve fuel economy and mitigate on-road greenhouse gas emissions worldwide. However, lightweight materials such as aluminum, magnesium, and carbon fiber are generally more energy intensive to produce and generate more emissions than conventional materials such as steel and steel alloys. There is a trade-off between increased emissions in the material production phase and decreased emissions in the fuel-cycle (all stages related to fuel production and consumption) (DOE, 2015; Modaresi et al., 2014; Kim and Wallington, 2013; Kim et al., 2015; Lewis et al., 2014; Schmidt et al., 2004). Life cycle assessments accounting for contributions from materials production, component manufacture, vehicle assembly, fuel production, vehicle use, and end-of-life management are required to evaluate the environmental impact of lightweighting.

Transparent and accurate data for critical parameters, such as upstream energy intensity of material production, recycling rates, material substitution rates, and fuel reduction values (FRVs) are needed to estimate the potential energy savings and greenhouse gas (GHG) mitigations from vehicle lightweighting. These parameters need to reflect local supply chain and real-world driving conditions. Most published studies have been conducted for vehicle lightweighting in North America and Europe (DOE, 2015; Luk et al., 2017; Raugei et al., 2015). However, the upstream industry energy intensity and on-road operation conditions in Asia differ significantly from those in North America and Europe. Recent studies in Japan (Palencia et al., 2015), India (Upadhyayula et al., 2019), and China (Sun et al., 2019) demonstrate increasing interest in vehicle lightweighting in Asia.

Palencia et al. (2015) reported up to 92.2% GHG reduction potential...
for vehicle lightweighting combined with electrification compared to a baseline fleet in 2050 but focused on fuel cycle emissions instead of a complete cradle-to-grave analysis. Upadhyayula et al. (2019) report a life cycle GHG emission reduction of 17% from vehicle lightweighting in India, and that material production emission intensity is a critical parameter. However, Upadhyayula et al. (2019) applied unspecified global average GHG burdens for steel and aluminum production. Sun et al. (2019) demonstrated that FRVs strongly impact life cycle GHG emissions, but considered generic fuel consumption values and FRVs which do not provide insight into in-use performance for specific vehicle models in the market. None of the above-mentioned studies provided a transparent process-by-process inventory of energy use and emissions during material production in Asia.

To better understand lightweighting benefits from a real-world perspective, assessments are needed that capture material manufacturing and real-world driving characteristics in Asian countries. There are several reasons why China, in particular, warrants individual analysis.

First, there were 29 million vehicles sold in China in 2017 (ATRC, 2017), and for the eighth consecutive year, China was the largest global vehicle market. To address vehicle-related energy and environmental concerns, policymakers have mandated fuel efficiency improvements. The Ministry of Industry and Information Technology (MIIT) of the Chinese government requires improvement of the fleet average fuel consumption of passenger vehicles from 6.9 L/100 km in 2016 to 5.1 L/100 km in 2020 (MIIT, 2012), a compound annual improvement rate of 7.7%. The roadmap released by SAE-China proposed fuel consumption of 4.0 L/100 km in 2025 and 3.2 L/100 km in 2030 (China Society of Automotive Engineers (China SAE), 2016). Vehicle lightweighting is an effective measure to improve vehicle fuel efficiency and mitigate the use-phase environmental impacts.

Second, China is the largest producer of aluminum and steel, accounting for 45% and 50% of global production in 2016 (World Aluminium Association, 2019; World Steel Association, 2018). Advanced high-strength steel (AHSS) and aluminum are currently the dominant lightweighting options for the mass application. The China SAE roadmap (China Society of Automotive Engineers (China SAE), 2016) proposes use of 190 kg aluminum per vehicle and 50% of steel to be AHSS by 2020. AHSS is generally compatible with existing manufacturing and materials. Aluminum and Al alloys have well-developed processing technologies for high volume manufacturing. Other lightweight materials, including magnesium and carbon fiber, are used in limited amounts because of high cost and difficulty in manufacturing and are typically viewed as longer-term options (China Society of Automotive Engineers (China SAE), 2016; DOE, 2019). A comprehensive and self-consistent data set of energy use and emissions associated with aluminum and steel production is needed to evaluate and compare the life cycle energy benefit of vehicle lightweighting. Unfortunately, such a data set is lacking for China. A goal of the present work is to provide these data.

Third, driving conditions and vehicle fuel economy vary across countries. For example, the average fuel consumption of new light duty vehicles in 2017 in the U.S., U.K., China, and India is 8.6, 5.8, 7.6, and 5.6 L/100 km respectively (Yang and Bandivadekar, 2017). Such variance further influences vehicle lightweighting benefits. Greater benefits from vehicle lightweighting are observed for trips with higher fuel consumptions (e.g., city cycle compared with highway cycle), and less efficient vehicle models (Luk et al., 2017; Kim et al., 2015; Kim and Wallington, 2013). Vehicle fuel reduction values (FRVs, reported in L/(100 km 100 kg)) can be used to estimate fuel reduction benefits of lightweighting. Previous studies have often used generic estimates of FRVs, in the range 0.15–0.3 and 0.25–0.5 L/(100 km 100 kg) for without and with powertrain adjustment (Luk et al., 2017). These ranges lead to uncertainties in estimating energy benefits of lightweighting and cannot be used to estimate benefits for a specific vehicle model.

To address the research gap indicated above, this study evaluates FRVs and life cycle GHG mitigation benefits of eleven passenger car models in the Chinese market, with considerations of local features such as Chinese-specific driving cycles and life cycle inventory data. Cradle-to-gate energy use and GHG emissions associated with the production of AHSS and aluminum in China are presented and compared with previous studies in Asia, the U.S., and global markets. FRVs for eleven vehicle models are estimated using on-road fuel consumption data and a physics-based FRV model. An evaluation of the GHG benefits of replacing conventional steel with AHSS or aluminum in vehicle body is provided.

2. Method and data

2.1. Scope, assumptions and system boundary

Aluminum and advanced high-strength steel (AHSS) are lightweight alternatives for standard steel that are likely to be widely adopted in passenger cars in the short term (China Society of Automotive Engineers (China SAE), 2016). An estimate of life cycle GHG emissions (E_{LC}) include emissions from material production (E_{mp}), component manufacturing and vehicle assembly (E_{m/a}), vehicle operation (E_{op}), fuel production (E_{fp}), maintenance and repair (E_{mr/r}), and end-of-life (E_{el}) as given by Eq. (1).

\[ E_{LC} = E_{mp} + E_{m/a} + E_{op} + E_{mr/r} + E_{el} \]  

The present study focuses on the material production, vehicle operation, and fuel production stages. Differences in energy use and emissions contributions from other stages (component manufacturing and vehicle assembly, maintenance and repair, and end-of-life) between baseline and lightweighted vehicles are minor (Keoleian and Sullivan, 2012; Kelly et al., 2015), and hence not included in the comparative analysis. GHG emissions from material production (E_{mp}) include on-site process fuel combustion emissions during material production, upstream emissions from process fuel production, and non-combustion emissions in the coking process and metal refining (e.g., perfluorocarbons during aluminum production). Characterization of E_{mp} was based on the Greenhouse gases, Regulated Emissions, and Energy use in Transportation (GREET) Model developed by Argonne National Laboratory (GREET, 2017) using updated material production inventory data determined in this study. GHG emissions from vehicle operation (E_{op}) includes tailpipe emissions and emissions from gasoline production. Tailpipe emissions were calculated from the fuel consumption (L/100 km) with changes between the baseline vehicle and lightweighted options determined using FRVs (fuel reduction values, L/(100 km 100 kg)). GHG emissions from gasoline production were calculated from fuel consumption determined in the present study, and fuel-cycle emissions factors taken from our previous work (Ke et al., 2017; Wang et al., 2015; Wu et al., 2012).

For simplicity, we do not consider secondary mass reductions, i.e., mass savings achieved in vehicle supporting parts when the vehicle mass is reduced, because it is less significant than primary mass reduction, highly uncertain, and requires complex modeling to estimate (Alonso et al., 2012). There are two approaches to account for the impact of material recycling. The recycled content approach (or cut-off approach) captures recycling benefits in the material production phase based on the share of secondary material used. The end-of-life approach accounts for recycling potential by applying recycling credits. We adopted the recycled content approach, consistent with the GREET model.

2.2. Aluminum (Al) production

Virgin Al production includes bauxite mining, alumina extraction, anode fabrication, electrolysis/ingot casting, and material transformation (casting or forging). Secondary Al processes include scrap
transportation, scrap preparation, remelting, machining, and material transformation (casting or forging) (see Fig. A1 for system boundary). In 2015, the production of primary and secondary aluminum in China was 31.5 and 4.6 million tons, respectively. Secondary aluminum accounted for 19.8% of total cast production and 12.5% of total wrought production (China Nonferrous Metal Industry Association, CNMIA, 2016). Domestic scrap Al does not meet the demand for secondary Al in China. In 2015, 2.08 million metric tons of aluminum scrap was imported and used to produce 1.24 million metric tons of secondary Al, accounting for 27% of total secondary Al production in China.

Table 1 presents process-by-process energy consumption of aluminum production in China. Industry data in China were obtained from approximately 70 aluminum production enterprises from the Chinese Aluminum Association (CAA) and used to calculate the material and energy consumption database for the two most energy-intensive processes; alumina extraction and electrolysis (China Aluminum Association CAA, 2017). Energy use data for other processes were taken from official statistics and literature as listed in Table 1. Comparing the results for China with those for the U.S. from the GREET_2017 model, alumina extraction is much more energy intensive, mainly due to lower quality bauxite, and electrolysis is less energy intensive, because most Al production facilities are new, or recently upgraded, and incorporate the latest technology.

GHG emission intensity for material production is determined by the energy intensity and share of fuels in each process (oil, electricity, natural gas, coal, etc., see Fig. A1 for details). Primary aluminum production is an energy-intensive process with electrolysis accounting for the majority of energy consumption. We used the industry-specific electricity mix for aluminum production, rather than national average grid mix, because the former better represents electricity consumed for aluminum electrolysis (Colett et al., 2016). Provincial primary aluminum production and electricity consumption data were collected in 31 Chinese provinces and used to adjust electricity sources specifically for aluminum production. As Fig. A1 shows, coal-fired electricity share of aluminum electrolysis is estimated to be 82% in 2015, which is higher than for the average national electricity mix (67%, Chinese Electric Yearbook, 2016). Shandong, Xinjiang, and Henan are the top three provinces in terms of aluminum production and the coal-fired electricity shares are higher in these provinces than the national average.

The electrolysis of alumina gives rise to emissions of tetrafluoromethane (CF4) and hexafluoroethane (C2F6). CO2 and C2F6 are perfluorocarbons (PFCs) with very long atmospheric lifetimes and very high global warming potentials. The CO2-equivalent PFC emission factor (EF_{PFC}, kg CO2eq/t-Al) was calculated using Eq. (2)

\[
EF_{PFC} = 6630 \times EF_{CF4} + 11100 \times EF_{C2F6}
\]  

(2)

where 6630 and 11100 are the 100-year time horizon global warming potentials of CF4 and C2F6 (Hodnebrog et al., 2013), and \( EF_{CF4} \) and \( EF_{C2F6} \) are the corresponding emission factors in kg/t-Al. The PFCs emission factors depend on the electrolysis technology and operation conditions. Point Fed Pre Bakes technology is currently used in China and we used the emission factors provided by the Chinese Aluminum Association, \( EF_{CF4} = 0.034 \text{ kg/kg-Al} \) and \( EF_{C2F6} = 0.0034 \text{ kg/kg-Al} \).

2.3. Advanced high strength steel (AHSS) production

The energy intensity for production of AHSS is generally assumed to be the same as that of conventional steel (DOE, 2015). Therefore, we chose to evaluate AHSS based on statistics for production of conventional steel products in China. Two processes, basic oxygen steelmaking (BOS) and electric arc furnaces (EAF), account for virtually all steel production worldwide. Processes in BOS include ore mining, coke production, sintering, pelletizing, blast furnace, basic oxygen furnace, and ingot casting. Processes in EAF include scrap transportation, melting in an electric furnace, and ingot casting. The EAF process in China uses 550–600 kg scrap to produce one t of crude steel on average, significantly lower than what is observed in other countries, and relies heavily on hot liquid pig iron input (Vercammen et al., 2017).

Steelmaking in China is dominated by the BOS process, accounting for 94% of national steel production in 2016 (World Steel Association, 2018). The EAF has limited production volume mainly due to the high cost of scrap steel and energy for producing hot liquid pig iron. We consider the Bos exclusively below. Steel ingots are converted into steel sheets through hot rolling and cold rolling processes, and then a zinc coating is applied in a galvanizing process (see Fig. A2 for system boundary). Table 2 shows the process-by-process energy consumptions for steel production in China. Energy consumption in major processes were calculated using data from China Steel Yearbook (2016), which provides industry average statistics from China Steel & Iron Association (CSIA) member enterprises that accounted for 84.3% of national production in 2015. Energy use statistics in the China Steel Yearbook are based on information from steel enterprises and do not include coke production. Energy consumption data for coke production from Liu et al. (2016) were used in this study.

As seen from Table 2, steel production in China has similar energy intensity to those in the U.S., with slight differences mainly attributed to the different allocation of by-products. The coking process in China, however, is much more energy intensive than the U.S., because small-scale facilities (chamber height of 5.5 m or less) with lower energy efficiency accounts for 48% total coke production in China (China Coking Industry Association, 2016), while typical coke plants are larger, 6–8 m high, in western countries (Hein and Kaiser, 2012).

2.4. Lightweighting fuel reduction for real-world driving conditions

2.4.1. FRV modeling

A physics-based assessment model (Kim et al., 2015) was used to
estimate FRVs of eleven passenger cars over five different driving cycles. FRVs are usually evaluated without, or with, powertrain resizing. The latter assumes resizing of the powertrain such that the performance of the lightweighted vehicle matches that of the baseline vehicle. Resizing results in additional fuel savings and FRVs with powertrain resizing are larger than those determined without powertrain resizing.

Fig. 2 presents the FRV model framework used in the present study. Real-world and regulatory driving cycles in China, and on-road fuel consumption profiles obtained from eleven passenger cars were used in the FRV model of Kim et al. (2015). Details of the FRV modeling and the calculation sheet are given in Appendix.

### 2.4.2. Real-world and regulatory driving cycles in China

We have recently proposed two cycles to represent real-world driving in Beijing, namely the Beijing Peak Hour Cycle (BPHC) and Beijing Off-peak Hour Cycle (BOHC) (Ma et al., 2019). Representativeness of the real-world driving cycles was insured by using large-scale GPS data sets from 459 privately-owned passenger sedans and SUVs in Beijing, covering nearly 17,000 sampling days and 3.3 million km travelled (He et al., 2016). The cycles are shown in Fig. 3 and described in detail elsewhere (Ma et al., 2019).

We also include a standard driving cycle developed by the China Automotive Technology & Research Center (CATARC, 2018), the China Light-duty vehicle Test Cycle (CLTC). The CLTC was developed based on large-scale real-world driving profiles collected in 41 cities in China with total driving distance of 22.6 million kilometers. Note that the CLTC is a national average driving cycle whereas the driving conditions in megacities such as Beijing and Shanghai could be different. Two regulatory driving cycles, the New European Driving Cycle (NEDC) and the Worldwide harmonized Light vehicles Test Cycles (WLTC), are included as well. The NEDC is the current cycle used in China fuel consumption regulations, and the WLTC has been adopted in the upcoming China 6 emission standard for criteria pollutants.

### Table 2

<table>
<thead>
<tr>
<th>Process Energy Use for Steel Production in China and the U.S., MJ/t of Steel</th>
<th>China (this study)</th>
<th>U.S. (GREET default)</th>
<th>Data source for China</th>
</tr>
</thead>
<tbody>
<tr>
<td>Iron Ore Extraction and Processing (per t of Iron Ore)</td>
<td>$0.393 \times 10^3$</td>
<td>$2.09 \times 10^3$</td>
<td>China Steel Yearbook, 2016</td>
</tr>
<tr>
<td>Coke Production (per t of Steel)</td>
<td>$3.93 \times 10^4$</td>
<td>$1.81 \times 10^4$</td>
<td>Liu et al., 2016</td>
</tr>
<tr>
<td>Sintering (per t of Steel)</td>
<td>$2.21 \times 10^3$</td>
<td>70</td>
<td>China Steel Yearbook, 2016</td>
</tr>
<tr>
<td>Blast Furnace (per t of Steel)</td>
<td>$2.56 \times 10^3$</td>
<td>$2.07 \times 10^3$</td>
<td>China Steel Yearbook, 2016</td>
</tr>
<tr>
<td>Basic Oxygen Furnace (per t of Steel)</td>
<td>$6.69 \times 10^3$</td>
<td>$8.04 \times 10^3$</td>
<td>China Steel Yearbook, 2016</td>
</tr>
<tr>
<td>On-site Generation and Other Steam Uses and Losses (per t of Steel)</td>
<td>$1.22 \times 10^4$</td>
<td>$-4.8 \times 10^3$</td>
<td>China Steel Yearbook, 2016</td>
</tr>
<tr>
<td>Hot Rolling (per t of Hot Strip)</td>
<td>$1.79 \times 10^4$</td>
<td>$1.55 \times 10^3$</td>
<td>China Steel Yearbook, 2016</td>
</tr>
<tr>
<td>Skin Mill (per t of Hot Rolled Steel)</td>
<td>—</td>
<td>$0.049 \times 10^3$</td>
<td>see footnote</td>
</tr>
<tr>
<td>Cold Rolling (per t of Cold Rolled Steel)</td>
<td>$2.49 \times 10^3$</td>
<td>$1.63 \times 10^3$</td>
<td>China Steel Yearbook, 2016</td>
</tr>
<tr>
<td>Galvanizing (per t of Galvanized Steel)</td>
<td>$1.61 \times 10^3$</td>
<td>$0.81 \times 10^3$</td>
<td>China Steel Yearbook, 2016</td>
</tr>
<tr>
<td>Stamping</td>
<td>$1.87 \times 10^4$</td>
<td>$1.00 \times 10^3$</td>
<td>see footnote</td>
</tr>
</tbody>
</table>

Note: CSIA = China Steel and Iron Association. These data reflect purchased energy inputs and do not account for energy associated with feedstocks and ancillary materials (e.g., lime). Energy input from energy carriers (e.g., blast furnace gas) is combined with purchased energy for CSIA data but listed separately in GREET model. Due to the complexity of steel production, some processes were combined in CSIA statistics. Skin mill is merged into hot rolling. Stamping is considered as steel processing in CSIA statistics. The shares of different fuels (coal, natural gas, electricity, coke oven gas, etc.) for each process were estimated using industry average data reported by Liu et al. (2016) and Qiu et al. (2007). See Fig. A2 for details.
Key cycle parameters for the cycles are compared in Table 3. BPHC and BOHC have the same duration of 1400 s, which is the average trip duration estimated from He et al. (2016). The BPHC represents driving conditions during peak hours and has an average speed of 23.9 km/h and an idle time of 26%. The BOHC has slightly higher average speed (28.8 km/h) and is similar to the CLTC. The NEDC has higher average speed than the two Beijing cycles and is much milder in terms of two travel dynamics parameters (relative positive acceleration, RPA, and the 95th percentile multiple of speed and positive acceleration, $S \times a_{pos[95]}$). The WLTC is intermediate in terms of aggressiveness and has travel dynamics similar to CLTC. However, it has a lower idle time fraction than the Beijing cycles and includes an extra-high speed phase (maximum speed = 131 km/h), which is not representative of typical driving patterns in China.

2.4.3. Real-world energy consumption

Vehicle fuel consumption measured in real-world driving is higher than type-approval values. Our previous study collected on-road fuel consumption data from eleven vehicles using on-board-diagnostics-II (OBD-II) scan tools (Lu et al., 2018). The OBD-II provides access to real-time diagnostic signals from the engine control unit (ECU). In addition to instantaneous vehicle speed and acceleration, OBD-II also provides real-time mass airflow and air-fuel ratio to do a quite accurate estimate of fuel rate. The eleven vehicles tested were manufactured from 2015 to 2016, including compact, mid-size, and full-size passenger cars (see Table A5 for displacement and vehicle weight). The vehicles were tested over routes designed to cover different regions, road types, and travel time.

To evaluate the distance-based fuel consumption, an operating binning method was applied, which uses speed and vehicle specific power (VSP) to represent micro operating conditions and relates them to instantaneous fuel rate. VSP (kilowatts per t) is defined as the instantaneous power demand of the vehicle divided by its mass, and is the sum of loads resulting from aerodynamic drag, acceleration, rolling resistance, and hill climbing. The following expression was used to calculate VSP (Jiménez-Palacios, 1999; Zhang et al., 2014).

\[
VSP = v \times (1.1 \times a + 9.81 \times \sin(\text{grade}) + 0.132) + 3.02 \times 10^{-4} \times v^3
\]

VSP (kW/t) is calculated vehicle specific power; $v$ (m/s) is instantaneous vehicle speed recorded by OBD-II; $a$ (m/s$^2$) is instantaneous vehicle acceleration; and grade is the angle of inclination of the road, which is set to 0 in this study.

Fig. 3. Beijing Peak Hour Cycle (BPHC), and Beijing Off-peak Hour Cycle (BOHC) (Ma et al., 2019). Peak hour is 7–9 am and 5–7 pm on weekdays, and off-peak hour is all other times.

### Table 3

<table>
<thead>
<tr>
<th>Parameters</th>
<th>Unit</th>
<th>BPHC</th>
<th>BOHC</th>
<th>CLTC</th>
<th>NEDC</th>
<th>WLTC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Duration</td>
<td>s</td>
<td>1400</td>
<td>1400</td>
<td>1800</td>
<td>1180</td>
<td>1800</td>
</tr>
<tr>
<td>Distance</td>
<td>km</td>
<td>9.3</td>
<td>11.0</td>
<td>14.5</td>
<td>10.9</td>
<td>23.3</td>
</tr>
<tr>
<td>Average speed</td>
<td>km/h</td>
<td>23.9</td>
<td>28.8</td>
<td>29.0</td>
<td>33.3</td>
<td>46.5</td>
</tr>
<tr>
<td>Average running speed</td>
<td>km/h</td>
<td>33.1</td>
<td>38.5</td>
<td>37.7</td>
<td>43.6</td>
<td>53.0</td>
</tr>
<tr>
<td>Speed $[90]$</td>
<td>km/h</td>
<td>71.0</td>
<td>86.0</td>
<td>114.0</td>
<td>120.0</td>
<td>131.3</td>
</tr>
<tr>
<td>Idle time percentage</td>
<td>%</td>
<td>26</td>
<td>24</td>
<td>23</td>
<td>24</td>
<td>12</td>
</tr>
<tr>
<td>Average acceleration</td>
<td>m/s$^2$</td>
<td>0.507</td>
<td>0.506</td>
<td>0.415</td>
<td>0.480</td>
<td>0.477</td>
</tr>
<tr>
<td>Average deceleration</td>
<td>m/s$^2$</td>
<td>0.570</td>
<td>0.558</td>
<td>0.464</td>
<td>0.683</td>
<td>0.514</td>
</tr>
<tr>
<td>RPA $^a$</td>
<td>m/s$^2$</td>
<td>0.21</td>
<td>0.20</td>
<td>0.17</td>
<td>0.11</td>
<td>0.15</td>
</tr>
<tr>
<td>$S \times a_{pos[95]}$ $^b$</td>
<td>m$^2$/s$^2$</td>
<td>6.1</td>
<td>9.2</td>
<td>6.8</td>
<td>6.3</td>
<td>9.2</td>
</tr>
</tbody>
</table>

Note: $^a$ RPA stands for relative positive acceleration.

$^b$ $S \times a_{pos[95]}$ is the 95th percentile multiple of speed and positive acceleration.

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Driving conditions were allocated into 28 operating mode bins based on instantaneous VSP and speed (v), including a deceleration or braking bin, an idling bin, and others to represent cruise or acceleration driving modes (Zhang et al., 2014) (see Table A4 for detail).

The distance-based fuel consumption of each driving cycle was calculated from the average fuel rate for each operating model bin and the time spent in each bin:

\[ FC_{ijk} = \left( \sum_{j} P_{ij,k} \times \frac{1}{T_{i,k}} \sum_{j=1}^{T} FR_{i,j,k} \right) \times \frac{1}{\rho} \times T_{i,k} \times \frac{1}{D_{i,k}} \times 100 \]

where \( FC_{ij} \) (L/100 km) is fuel consumption of vehicle \( j \) under driving cycle \( k \); \( P_{ij} \) is percentage of operating mode \( i \) in driving cycle \( k \); \( T_{i} \) is the number of data points for vehicle \( j \) in operating mode \( i \); \( FR_{i,j,k} \) is the instantaneous fuel rate in operating mode \( i \) at second \( t \); \( \rho \) (g/L) is the density of gasoline, 725 g/L for 93 Research Octane Number (RON) gasoline; and \( T_{i,k} \) and \( D_{i,k} \) are travel time and total distance of vehicle \( j \) for driving cycle \( k \). Table A5 shows the distance-based fuel consumption of eleven vehicles tested over five driving cycles, which were used to calculate the FRVs and FRVs*.

2.5. Fuel-cycle GHG emissions

The fuel-cycle GHG emissions for combustion of one liter of gasoline is calculated as:

\[ EF_{i} = \left( \frac{EF_{i}}{\delta} + EF_{fuel} \right) \times LHV_{gas} + \frac{\phi_{CO_{2}} \times LHV_{gas}}{1244} \]

where \( EF_{i} \) is the fuel-cycle GHG emission factor, g CO₂eq/L gasoline; \( EF_{i}/\delta \) is the GHG emissions for producing crude, 11.7 g CO₂eq/MJ; \( \delta \) is loss factor, 0.86 MJ gasoline/MJ crude; \( EF_{fuel} \) is the GHG emission for producing gasoline (gasoline blendstock), 13.4 g CO₂eq/MJ; \( LHV_{gas} \) is the heating value of gasoline, 116,090 btu/gal = 32.4 MJ/L; \( \phi_{CO_{2}} \) is the carbon content by mass of gasoline, 86.3%; \( \rho_{gas} \) = 0.749 kg/L. The values above are China-specific data taken from Wang et al. (2015) and Ke et al. (2017). 1244 is the ratio of the atomic mass of carbon to the molecular weight of CO₂. Using Eq. (5) gives \( EF_{i} \) = 3241 g CO₂eq/L.

3. Results

3.1. Cradle-to-gate GHG emissions for AHSS and aluminum

The cradle-to-gate GHG emission results for AHSS and aluminum are given in Fig. 4. We consider wrought aluminum (W. Al) as the lightweight material, which has GHG emissions of 17.5 t CO₂eq/t W. Al, based on a share of 87.5% for primary Al. For primary Al ingot, the GREET 2017 model and IAI (2017) reported 8.3 and 17.6 t CO₂eq/t Al ingot in the U.S. and globally, which are 57% and 10% lower than our estimation for China (19.5 t CO₂eq/t Al ingot). This is mainly a result of much more carbon-intensive electricity used in China. IAI (2017) reported 20.1 t CO₂eq/t Al ingot in China, higher than our estimation by 3%, mainly because they assumed 90% coal-based electricity rather than 82% estimated in the present study. Sun et al. (2019) reported 15.1 t CO₂eq/t primary Al ingot, 23% lower than our estimation, referring to a China Automotive Life Cycle Database (CALCD), but the material production system boundaries and process-by-process energy use and emission profiles were not reported and hence it is difficult to compare or validate their results. Hao et al. (2016) estimated 16.5 t CO₂eq/t Al ingot in China, which is 15% lower than our estimation. This can be explained by the fact that Hao et al. (2016) used a simplified approach that accounts for direct emissions from Al industry and indirect emissions from electricity, instead of a complete life cycle analysis that accounts for upstream fuel-cycle emissions. McMillan and Keoleian (2009) reported 21.9 t CO₂eq/t Al primary ingot in Asia, which is 12% higher than our estimation for China. While our estimation is based on a higher share of coal-fired electricity (82%, compared with 60% in McMillan and Keoleian (2009) study), the PFC emission intensity is lower (2.5 versus 3.25 t CO₂eq/t Al primary ingot), reflecting improvements in emission control progress over the past decade. Moreover, our estimation also implies lower smelter electricity intensity compared with previous years. Gao et al. (2009) had a higher estimation of 22 t CO₂eq/t primary Al ingot but their study is based on 2003 industry data with outdated technology. GHG emissions for secondary Al ingot in China are higher than that in the US (GREET default value) because of the carbon-intensive electricity and emissions from long distance transport of imported scrap Al.

AHSS is estimated to have GHG emissions of 3.9 t CO₂eq/t, which is approximately 20% higher than the U.S. value (3.2 CO₂eq/t, GREET default). Sun et al. (2019) reported a value of 3.1 CO₂eq/t in their study for China, but did not provide system boundary and a process-by-process inventory. The World Steel Association reported much lower GHG emissions for global average steel production (2.4 t CO₂eq/t hot-dip galvanized steel (World Steel Association, 2016)), but did not provide a process-by-process inventory, making it difficult to compare results directly. We do not account for use of secondary steel (produced via EAF process) in the present work because the EAF process has only 6% share in total steelmaking in China, and also because secondary steel is less likely to be used to produce AHSS due to the need for specific alloys compositions that are less tolerant of impurities.

3.2. Fuel reduction values

Fuel reduction values without and with powertrain adjustment (FRVs and FRVs*) of eleven vehicles over five different driving cycles are presented in Fig. 5. The FRVs and FRVs* are 0.15–0.27 and 0.24–0.61 L/(100 km 100 kg), respectively (see Table A5 for calculation sheet), consistent with, but with upper-bounds 20% higher than, those reported in previous studies (Luk et al., 2017; Kim et al., 2015; Kelly et al., 2015). With powertrain resizing, FRVs* have a positive correlation with fuel consumption (see fit in Fig. 5a), indicating that vehicles with higher fuel consumption (depending on vehicle and driving cycle) generally have greater benefits from weight reduction. Without powertrain resizing, FRVs show no discernable correlation with fuel consumption for a given cycle (see fit in Fig. 5b), consistent with the observations by Kim and Wallington (2013). More aggressive driving cycles, e.g. BPHC and BOHC (see Table A2 for cycle parameters) tend to have higher FRVs and higher fuel consumption than milder driving cycles, because acceleration is a key contributor to FRVs and fuel consumption.

We now consider the influence of driving cycles on FRV and FRV*. As seen from inspection of Table 4, BPHC has the highest FRVs* [[(0.257 ± 0.007)/L/(100 km 100 kg)] and FRVs* [[(0.507 ± 0.058)/L/(100 km 100 kg)] of all the cycles. In terms of FRV*, an average lightweighted vehicle in this study has 19%, 39%, 61% greater fuel reduction benefits over the BPHC than over the CLTC, NEDC and WLTC. This difference reflects the different levels of congested driving in these cycles. BPHC and BOHC have lower average speed than the NEDC and WLTC, with more stop-and-go driving. Driving cycles that require more power demand overall have larger FRVs and FRVs*. The WLTC has an extra high-speed phase, requiring the highest power demand to overcome aerodynamic resistance, but this aerodynamic load is independent of vehicle weight and hence is not affected by vehicle lightweighting.

3.3. Cradle-to-grave GHG benefits

To illustrate the cradle-to-grave GHG mitigation benefits of AHSS and Al, we consider a case study of vehicle body lightweighting. As shown in Table 5, replacing 50% of steel in the body (original weight 475 kg (GREET, 2017) with AHSS is estimated to give 47.6 kg mass reduction (median value adopted from Kelly et al. (2015)). Given that AHSS has a GHG emissions intensity that is assumed to be the same as conventional steel (3.9 kg CO₂eq/kg), this results in a
47.6 \times 3.9 = 185.6 \text{ kg/CO}_2\text{eq reduction in cradle-to-gate GHG emissions per vehicle. Note that the reduction in AHSS GHG emissions in the material cycle is the result of a reduction of material use (47.6 kg). Aluminum has greater potential for mass reduction (107 kg), but the GHG emissions intensity is much higher (17.5 kg CO}_2\text{eq/kg). This reflects the high carbon intensity electricity currently used in the Chinese aluminum industry. Therefore, replacing steel with wrought Al results in a net increase of material cycle GHG emissions (1358 kg/CO}_2\text{eq per vehicle). In the future as the Chinese grid CO}_2\text{ emissions intensity decreases, the Al production burdens will decrease and lightweighting using Al will become more attractive.}

A complete picture of life cycle benefits must take into account changes in vehicle material manufacture and vehicle operation over its lifetime (200,000 km total mileage for light-duty vehicles in China).
(Zhou, 2016); sensitivity analysis for 100,000 km and 300,000 km is given in Appendix). We present the life cycle lightweighting GHG savings in Fig. 6 and note that a generic gasoline vehicle has a cradle-to-grave GHG burden of 50.0 t CO₂eq per vehicle (Qiao et al., 2019) as a baseline for comparison. Four points are evident from Fig. 6.

First, compared with conventional steel, AHSS has GHG savings in all scenarios considered, 0.7–1.0 and 1.1–1.7 t CO₂eq per vehicle (1.4%–2.0% and 2.2%–3.4% of cradle-to-grave emissions for a generic gasoline vehicle) without, and with powertrain resizing. This is a combined result of decreased cradle-to-gate GHG emissions and mitigation of on-road GHG emissions. Wrought Al, on the other hand, does not have cradle-to-gate GHG savings. Therefore, whether there will be net life cycle GHG savings relies on the amount of GHG savings during the use phase. Without powertrain resizing, use of wrought Al results in lower GHG emissions for the three China cycles (BPHC, BOHC, and CLTC) and the WLTC, but higher GHG emissions for the NEDC.

Second, AHSS is more consistent in achieving GHG savings in all cases, while Al has greater GHG savings in cases with high GHG reduction during the use-phase but negative GHG savings in some other cases. Given the much higher GHG burden to produce Al than AHSS, wrought Al only outperforms AHSS when on-road fuel savings are high. For example, cases for congested driving cycles (BPHC and BOHC) with powertrain resizing have GHG savings of 1.9–2.2 t CO₂eq per vehicle using aluminum, and 1.6–1.7 t CO₂eq per vehicle using AHSS. Cases for long lifetime driving distance (300,000 km, see Fig. A3) with powertrain resizing have GHG savings of 1.9–3.9 t CO₂eq per vehicle using aluminum, and 1.6–2.5 t CO₂eq per vehicle using AHSS.

Third, the two Beijing cycles (BPHC and BOHC) have the highest cradle-to-grave GHG savings, the CLTC has intermediate GHG savings, and NEDC and WLTC have the lowest cradle-to-grave GHG savings. The results indicate that congested traffic conditions make lightweighting a particularly effective emission reduction strategy. BPHC and BOHC have lower average speed and more stop-and-go driving, requiring greater mass-related drag force, i.e., acceleration resistance. Vehicle lightweighting has greater fuel consumption and emission reduction benefits for the Beijing cycles than other cycles. This has not been reported in previous studies, which typically use type-approval fuel consumption values and generic FRVs (e.g., Sun et al. (2019)).

Fourth, lightweighting with powertrain resizing is substantially more effective than that without powertrain resizing. AHSS and wrought Al have 1.1–1.7 and 0.8–2.2 t CO₂eq/vehicle GHG reductions with powertrain resizing, much higher than the cases without powertrain resizing (0.7–1.0 and (−0.1)–0.4 t CO₂eq/vehicle). This highlights the importance of the physics-based estimation of FRV and FRV+ applied in this study which captures the impact of powertrain resizing. This emission reduction benefit is further enhanced in cases for two Beijing cycles that have higher FRVs, and higher substitution ratio (see Fig. A4).

4. Conclusion

Life cycle GHG mitigation benefits of vehicle lightweighting using AHSS and aluminum in China were evaluated. Production of AHSS and

| Table 4 | Fuel reduction values without (FRV) and with (FRV+) powertrain resizing for eleven vehicles over the five driving cycles (mean ± standard deviation). |
|---------|------------------|---------------------|------------------|-----------------------|---------------------|
|         | BPHC             | BOHC               | CLTC             | NEDC                  | WLTC                |
| FRV     | 0.257 ± 0.007    | 0.241 ± 0.008      | 0.227 ± 0.009    | 0.176 ± 0.011         | 0.210 ± 0.014       |
| FRV+    | 0.507 ± 0.058    | 0.464 ± 0.054      | 0.425 ± 0.054    | 0.365 ± 0.047         | 0.314 ± 0.049       |

<table>
<thead>
<tr>
<th>Table 5</th>
<th>Weight and material-cycle GHG emissions impact of replacing steel in the vehicle body.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Metric</td>
<td>Lightweight material</td>
</tr>
<tr>
<td></td>
<td>AHSS</td>
</tr>
<tr>
<td>Cradle-to-gate GHG, kg CO₂eq/ kg</td>
<td>3.9</td>
</tr>
<tr>
<td>Baseline mass, kg</td>
<td>475 (Conventional steel)</td>
</tr>
<tr>
<td>Lightweighed mass, kg</td>
<td>427.4 = 189.9 (AHSS) + 237.5 (Conventional steel)</td>
</tr>
<tr>
<td>Change in material-cycle GHG emissions, kg/ CO₂eq per vehicle</td>
<td>−185.6</td>
</tr>
</tbody>
</table>

Note: Original weight of the steel in vehicle body is 475 kg (GREET, 2017). Replacement ratio of original material is set at 50%, since it is unlikely that all original material could be replaced. Replacement ratios of 20% and 80% are discussed in Appendix. Weight change values are taken from Kelly et al. (2015).
wrought aluminum have average cradle-to-gate GHG emissions of 3.9 and 17.5 kg CO₂eq/kg. A physics-based model was used to evaluate lightweighting benefits for eleven passenger car models over five driving cycles (including real-world and regulatory cycles). Replacing steel with AHSS results in GHG savings in all scenarios, 0.7–1.0 and 1.1–1.7 t CO₂eq per vehicle without, and with powertrain resizing. Despite the much higher GHG emissions associated with wrought Al production, its use results in life cycle GHG savings in most cases analyzed except when driving distance is short (100,000 km per lifetime) or driving conditions are mild. Maximum GHG savings occur with Al versus AHSS in cases where the powertrain is resized, travel is congested, or lifetime travel distance is long. Beijing driving cycles (BPHC and BOHC) have the highest GHG savings, 1.9–2.2 t CO₂eq per vehicle (aluminum, with powertrain resizing), compared to 0.8–1.6 t CO₂eq per vehicle for other cycles, which implies that congested traffic conditions in Beijing make lightweighting a particularly effective emission reduction strategy. The results also highlight the difference in electricity carbon intensity between China and the U.S. and its impact on the breakeven of lightweighting material production and vehicle operation, which affects aluminum more significantly than steel. In the future, as the Chinese grid CO₂ emissions intensity decreases, the aluminum material production burdens will decrease in China and lightweighting using aluminum will become more attractive.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:10.1016/j.resconrec.2019.104497.

References


