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Critical factors affecting life cycle assessments of material choice for vehicle mass reduction

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1. INTRODUCTION

1.1. Vehicle Mass Reduction

New greenhouse gas (GHG) standards for cars and light trucks are taking effect for model year 2017, progressing towards an anticipated sales-weighted average level of 173 g/mile CO₂ for model year 2025, and fuel economy standards increasing each year to the Corporate Average Fuel Economy (CAFE) target of 51.4 mpg fleet-wide by 2025 (for a projected vehicle sales mix). As a result, vehicle manufacturers are looking for solutions that can meet these goals without sacrificing marketable vehicle attributes (Nehuis et al., 2014; U.S. EPA, 2012a, 2014). Reducing mass enables vehicles to operate more efficiently during the use phase because energy demands (e.g., acceleration, rolling friction) on the powertrain are reduced. This reduction in mass can have major benefits on the total life-cycle impacts of vehicles because the current use phase accounts for 84–88% of the total life-cycle energy consumption and GFIG emissions for conventional light-duty vehicles. Comparatively, the manufacturing contributes approximately 4–7% of the energy consumption over the life of a light-duty vehicle (Keoleian and Sullivan, 2012; Mcauley, 2003; Sullivan and Cobas-Flores, 2001; Sullivan et al., 1998). Because of this dominant contribution of impacts from the use phase, mass reduction efforts and other use-phase efficiency measures provide an effective means to reduce the total life-cycle impacts. However, the share of life-cycle impacts between the production and use phase for vehicles is likely to shift away from the use phase with increasing efficiency and with reduced light-duty vehicle GFIG emissions standards, as shown in the example comparison in Fig. 1

Vehicle mass reduction is an effective method to improve vehicle operating efficiency and reduce both fuel consumption and CO₂ emissions by reducing vehicle road loads. In some instances, vehicle mass reduction can be used to offset mass gained through the use of other technologies, such as the weight of additional battery capacity needed for vehicle electrification. Computer-aided design has enabled engineers to make significant gains in recent years through the use of structural optimization models and dynamic vehicle crash models that have contributed to increased safety and vehicle mass reduction by reducing the overall materials required within the structure of the vehicle (Das, 2014a; Nehuis et al., 2014). With optimization of the vehicle body structure and geometry now being an inherent step in the design of a new vehicle, further vehicle mass reduction must be accomplished by expanding the suite of materials utilized in vehicle construction through targeted efforts based on specific material properties, forming technologies, and integration into complete vehicle designs (Mayyas et al., 2012a; Modaresi et al., 2014). The primary contributors to a vehicle's weight and the focus of most mass reduction efforts are the components that make up the body-in-white (BIW) (the stage in automotive design and manufacturing where a vehicle's body components have been welded together to form a structure) and closures, which are primarily metal in most current light-duty vehicles (Caffrey et al., 2015).

Some materials perform better than others for certain vehicle applications based on the current range of characteristics of a particular metal, metal alloy, or composite. This range of characteristics is highlighted by the progression of conventional steels to steels with higher tensile strength such as high-strength steels (HSSs) and advanced high-strength steels (AHSSs). AHSS refers to multiphase steels which may contain combinations of martensite, bainite, and retained austenite phases and having tensile strength in excess of 500 MPa. AHSS is further broken down into even higher tensile strength categories that include ultra-high-strength steels (UHSSs) and gigapascal (GPa) steels (i.e., steels with greater than 1 GPa tensile strength) (World Auto Steel, 2016). The higher strength steels can offset conventional steel within a structure by using less material in the same applications. Additionally, each material may have many different methods of forming that can make it more or less appropriate for a specific application (Nehuis et al., 2014). Component specific material selection allows for significant vehicle mass reduction through material replacement and design optimization as well as multi-material design which often integrates a number of previously separate components into one piece. This sort of component integration can be used to reduce both component mass and the finished cost of the component, thus improving the cost-effectiveness for CO₂ reduction.

Fig. 2 illustrates the typical vehicle-based material composition based on curb weight (the full weight of the vehicle including consumables such as fuel, but excluding passengers and cargo) for 2010 and 2013, with an overall decrease of 20 kg (1.1%) over three years. Fig. 2 also shows a specific example of the 2013 Ford Fusion, which has 1560 kg of mass and is made up of nearly half the steel and iron, five times as much AHSS/UHSS, and approximately the same amount of aluminum as the 2013 average. In 2015, a collaborative effort between the U.S. Department of Energy (DOE), Ford Motor Company and Magna International completed a holistic vehicle mass reduction study in which they reduced the weight of a 2013 Ford Fusion. The resulting lightweight concept vehicle design (designated Mach 1) reached a target 23.5% mass reduction. The Mach 1, as shown in Fig. 2, has a

nearly threefold increase of aluminum and a decrease of 4.5 times the amount of AHSS/UHSS with a small amount of magnesium being included and a reduction of the steel and iron content by 34%. Aluminum alloy parts were used in the body, interior, chassis, and powertrain. This was achieved through the use of castings, extrusions, stampings, and forgings (Bush et al., 2015).

Vehicle designs implementing mass reduction must also fulfill multiple design constraints, which include attaining manufacturer-designated safety targets, cost targets, functional targets, component durability targets, and noise-vibration-harshness (NVH) targets, which can take a higher priority than fuel efficiency and emissions reductions. According to Nehuis et al., the trend of increasing vehicle weight “was stopped by developing new vehicle concepts and by using lightweight materials (light metals, high-strength steel and fiber-reinforced plastics) intensively” (Nehuis et al., 2014).

Fig. 3 shows the changes in light-duty vehicle weight, fuel economy, and horsepower over the last 40 years. Vehicle weight stopped increasing starting in approximately 2004 and remained static while horsepower and, to an even greater extent, fuel economy continued to increase. Because reducing the mass of a vehicle increases the power-to-weight ratio, efforts seeking to address the new efficiency standards may be able to reduce powertrain output requirements to achieve equivalent or even improved performance characteristics compared to existing vehicles while reducing the mass even further (Kelly et al., 2015).

1.2. Life-Cycle Assessment

Federal vehicle standards and vehicle mass reduction efforts are focused on the use-phase efficiency of vehicles, but the use of alternative materials in vehicle manufacturing means there are also changing impacts resulting from differing methods of production for each primary material (Hula et al., 2016). Life-cycle assessment (LCA) has been used by vehicle manufacturers, the metal and plastics industries, and the academic community to identify the impacts of the many alternative methods of reducing vehicle mass associated with not only the use phase but also the manufacturing and end-of-life (EOL) of vehicles and their components. LCA is an established method used to quantify emissions and other impacts that can be attributed to the life cycle of products. The International Organization for Standardization (ISO) defines LCA and its applications in ISO-14040 (ISO, 2006a) and ISO-14044 (ISO, 2006b). The LCA framework as defined by ISO includes an iterative process of: (1) Goal and Scope Definition, (2) Inventory Analysis, (3) Impact Analysis, and (4) Interpretation (ISO, 2006a). Assessments may be conducted for conceptual models, individual components in a larger system, complete vehicles, or even entire fleets. LCAs require a significant amount of data to provide accurate accounting for all of the processes associated with the production, use, and EOL for a product. Often, the inventories created for vehicle LCAs use processes from existing databases or models such as ecoinvent (Frischknecht et al., 2005; Weidema et al., 2013), U.S. Life Cycle Inventory (USLCI) (NREL, 2012), and the Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) Model (Argonne National Lab, 2015).

The use of LCA for vehicle mass reduction focuses largely on the comparison of two or more material options, the impacts from production, use-phase, and EOL. The most

common impact categories reported in LCAs included in these reviews were GHG emissions and energy use. However, there were some studies that examined additional impact categories. The flexibility of LCA methodology enables its use across industries and in varying levels of specificity, but also creates variability in scope, boundaries, and assumptions that can make comparing LCAs difficult. Fig. 4 presents a generic life-cycle system diagram that would be appropriate to describe a full vehicle or a component-specific LCA which includes the production, use, and EOL phases. The figure includes several processes associated with recycling and the need for assumptions when allocating the benefits of recycling to avoid double counting. Recycling credits can be allocated based on recycled content, which is part of the production process via 'Secondary Material', or allocation can be based on the creation of recyclable materials through 'Process Scrap Recycling' and/or 'Post-Consumer Scrap Recycling.' LCA does not inherently account for economic factors or technical feasibility, which play a critical role in vehicle design and production (Mayyas et al., 2012a).

This review focuses on LCAs for vehicle mass reduction published from 2010 to 2016, providing context with regard to their collective findings. The discussion below explores the assumptions and data used in the reviewed LCAs and the uncertainty associated with key factors including fuel reduction values, material substitution rates, lifetime vehicle travel distance, recycling allocation, and secondary mass reduction. The materials assessed in these LCAs include aluminum, steel, HSS, AHSS, UHSS, magnesium, and several composites including glass fiber-reinforced polymers (GFRPs) and carbon fiber-reinforced polymers (CFRPs). The aggregated review of the LCA findings and the critical factors that are used in the assessments helps highlight commonalities and differences so decision makers, LCA practitioners, and vehicle manufacturers can better understand the literature as a whole and address the life-cycle sustainability for future vehicle designs.

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2. RECENT LCA STUDIES

For the review, twenty-six LCAs on light-duty vehicle mass reduction were identified. The studies, listed in Table 1, were published in 2010 or later and focus on the application of LCA to investigate the impacts of reducing mass from vehicles or their components (Baroth and McKay, 2012; Bushi et al., 2015; Das, 2011, 2014b; Dhingra and Das, 2014; Ding et al., 2016; Du et al., 2010a, 2010b; Dubreuil et al., 2012; Ehrenberger, 2013; Kelly et al., 2015;

Kim et al., 2010; Koffler, 2013; Lewis et al., 2014; Li et al., 2015; Marretta et al., 2012; Mayyas et al., 2012c; PE International, 2012a,b; Raugei et al., 2015; Simões et al., 2016; Stasinopoulos et al., 2012; Steel Market Development Institute, 2016; Tempelman, 2011; Tharumarajah and Koltun, 2010; Witik et al., 2011). The cutoff year of 2010 was selected to focus the review on LCAs that may capture more recent innovations in mass reduction technology, changing material compositions in light-duty vehicles, updated inventory data, and policy changes that have driven mass reduction efforts to the forefront. The literature review was conducted to identify all of the studies on vehicle mass reduction from peer-reviewed sources and grey literature via Google Scholar searches, references from other relevant works, and communication with policymakers as well as industry stakeholders who provided up-to-date studies being conducted within the industry.

All of the identified literature was then sorted into two categories: LCA and non-LCA. The non-LCA literature was used to provide background information for the introduction and also contributes to the discussion section after the review of the LCA studies. The LCAs were broken into vehicle-specific studies and other LCAs (primarily material-specific footprinting). Studies that focused primarily on a standalone assessment of a single technology or the development of life-cycle methodology using vehicle mass reduction as an example were excluded because they often use limited inventories and simplify the assessment in order to focus on the authors' methodological approach. Of the 26 LCAs, 16 were explicitly ISO compliant. While the other 10 did not state they were ISO compliant, the methodologies and data sources used were found to be comparable and similarly robust. Although some studies were categorized as non-LCAs, they provided important insights regarding industry trends and broader sustainability considerations.

Several of the LCAs reviewed, which used various underlying modeling assumptions, explicitly highlight what has previously been shown: the use phase dominates the energy use and GHG impacts of a conventional light-duty vehicle (Das, 2014b; Ding et al., 2016; Kim and Wallington, 2013a; Simões et al., 2016; Tempelman, 2011). The continuous use of fuel over many miles of vehicle operation results in cumulative impacts that are significantly greater than the mining, smelting, forming, assembly, and disposal of a vehicle. Another common finding across the studies is the value of leveraging mass reduction efforts to achieve additional reductions in impacts. Several studies found that primary mass reduction can enable additional design changes through powertrain downsizing and secondary mass reduction that, when combined, can provide greater use-phase reductions than the primary efforts alone (Dhingra and Das, 2014; Du et al., 2010a; Koffler, 2013). The additional benefits of secondary mass reduction and powertrain downsizing can be achieved only if the mass reduction across the total vehicle is significant enough to enable further reductions in weight-dependent systems, which are identified by Alonso et al. (2012). Studies that went beyond component-specific analyses (14 of 26 papers reviewed) were better able to capture holistic design impacts.

Approximately half of the studies focused on the U.S., while others focused on the EU or China. Most studies were in peer-reviewed journals (21 of 26 papers) but a number were included as reports or conference proceedings. Author affiliation was spread across universities (13), industry (5), national laboratories (8), and consultants and contractors (3).

Steel was included in all but one of the studies, and was typically included as the baseline material, comparing mild steel with higher strength steels and other materials. Aluminum was most commonly examined (18), followed by composites (13) and magnesium (11). Four of the studies examined all four materials (Bush et al., 2015; Kelly et al., 2015; Li et al., 2015; Mayyas et al., 2012c). While some of the studies focused on substitution of one material by another, others assessed multi-material design changes with different materials being used in conjunction to achieve greater mass reductions through a holistic design. Each study has a different set of system boundaries and assumptions with mild steel serving as the typical baseline for comparison of materials. The following sections evaluate the features of these LCAs, their results, and major factors influencing LCA studies on vehicle mass reduction.

2.1. Aluminum

Many of the LCAs comparing aluminum and steel produced results that showed aluminum had lower life cycle energy use and/or GHG emissions relative to steel (Das, 2014b; Ding et al., 2016; Kelly et al., 2015; Mayyas et al., 2012b; Rauei et al., 2015) and HSS/AHSS (Das, 2014b; Kim et al., 2010; Mayyas et al., 2012b), entirely due to the use-phase efficiency improvements attainable with aluminum. In most cases examined in the studies the production of aluminum had higher energy and GHG emissions relative to steel, but the dominance of the use-phase impacts in the life cycle of a vehicle means that any efficiencies gained during operation can provide significant life-cycle benefits (Das, 2014b; Du et al., 2010b; Lewis et al., 2014; Marretta et al., 2012; Mayyas et al., 2012c; Simões et al., 2016). That is, the higher GHG and energy use impacts that occur during the production phase of aluminum are more than offset by GHG and energy savings during the vehicle's use phase. This means that aluminum materials have incurred greater impacts as the vehicles sit in the dealers' lots, but at some point in the use phase, these deficits are overcome and continued vehicle use leads to cumulative reductions compared to a conventional vehicle. A few studies that evaluated breakeven driving distances found that the GHG deficit of aluminum production is overcome after 19,000 (Das, 2014a) to 160,000 (Mayyas et al., 2012c) km of driving. Kelly et al. found an overall breakeven distance range from 0 to 200,000 km, which was based on theoretical calculations using a variety of factors (Kelly et al., 2015).

Aluminum has lower life cycle impacts when compared to steel in general, but these life-cycle findings are sensitive to the assumptions and scenarios used for the energy mix and recycling. Because of the energy intensity of aluminum production, the manufacturing efficiency and GHG emissions rates for the electric energy mix assumed are all key factors that influence the associated emissions. The aluminum industry in North America has historically sited facilities to access regional electric grid mixes with lower cost and often lower GHG-emissions sources like hydropower, so LCA studies often assume a high proportion of hydroelectric power such as 53% hydro, 15% nuclear, and 20% coal (Rauei et al., 2014, 2015) or up to 75% hydroelectric power (Bush et al., 2015; Das, 2014b), which is based on a 2013 report by The Aluminum Association (2013). This distribution might be representative of the current energy delivered to aluminum manufacturing facilities but additional global demand for aluminum would require additional electricity production. The

newly supplied electricity would not necessarily match current electric grid mixes in terms of their GHG emissions rates (Weidema et al., 1999).

The technologies used to provide the increased electricity generation have marginal emission factors that may influence the results of the LCA. Assuming more hydroelectric or less carbon-intensive electricity for aluminum lowers the production impacts and thus overall life cycle impacts, whereas use of conventional fossil fuel electricity generation for aluminum processing increases the impacts of the production phase. Current national or regional electric grid averages are not well suited to model the emissions factors associated with the additional electricity load. These marginal emissions factors can be modeled regionally or nationally using electric power sector models such as the Integrated Planning Model (IPM) (U.S. EPA). For example, IPM has been used in the context of vehicle electrification to estimate upstream marginal emissions factors for regional electric grid mixes (U.S. EPA, 2012b). Moreover, the grid mix is changing over time with some technologies being phased out and others quickly gaining market share. The 2016 Annual Energy Outlook highlights this in its projections of zero to negative annual growth rates for coal-based generation, continued growth in natural gas-based generation, and high annual growth rates of 3–4% for renewables, depending on the reference case (U.S. EIA, 2016).

Recycling is another critical factor that could function to offset the impacts from energy intensive production of virgin aluminum. For example, cast aluminum has an embedded energy value of around 168 MJ/kg compared with recycled cast aluminum at 19 MJ/kg (an 89% reduction) and virgin steel at 26 MJ/kg (Lewis et al., 2014). The offsets due to recycling are perhaps the most significant factor for assessments evaluating aluminum as a mass reduction technology. The assumed percent of aluminum recycled and the method of allocation in an LCA can tip the findings one way or the other when aluminum is compared to the high strength steels. HSS, however, is much more dependent upon its ability to reduce the mass in the vehicle to achieve savings (Das, 2014b; Lewis et al., 2014; Marretta et al., 2012). Kim et al. explored the benefits of optimized recycling systems and high scrapping rates which resulted in up to 78% reductions in GHG payback periods for aluminum intensive vehicles (Kim et al., 2010). However, as also noted by Kim et al., the gains from recycling would only be realized once there is adequate material in the existing stock of vehicles and the retirement of the first generation aluminum vehicles, so a robust system of closed-loop recycling could capture the material for reuse. Steel has a lower potential to make improvements in this way because steel is already recovered at high rates (Kim et al., 2010). Stasinopoulos et al. summarize the dynamics of temporal factors in life-cycle inventories (LCIs) like this:

The fleet-based LCI example of substituting aluminium BIWs for steel BIWs suggests that product-based LCIs (a) overestimate the short-term energy benefits of aluminium by not accounting for the time required for the stock of preexisting steel components to decay out of the fleet and (b) underestimate the long-term energy benefits of aluminium components by not accounting for changes in the availability of recycled aluminium. (Stasinopoulos et al., 2012)

The distinction between LCAs for vehicles or components (product-based) and fleets is important because manufacturers are working to improve and refine specific features of their

vehicles that subsequently influence a large material stock. With 16.7 million vehicles produced in the U.S. in 2015, the potential shift in material flows is significant and cannot be ignored (Hula et al., 2016). These material flows are particularly important for aluminum because the transportation sector already accounts for 26% of the aluminum shipments within North America (The Aluminum Association, 2011). Some of LCAs reviewed include very specific component assessments while others evaluate the larger fleet dynamics, all of which must be taken into context when comparing their assumptions, results, and conclusions.

2.2. HSS and AHSS

Most of the LCAs include a baseline scenario based on steel, which is the traditional material for vehicle structures. However, this should not be confused with the application of HSS and AHSS for reducing vehicle mass. Newer steel alloys have improved material characteristics that allow for similar or improved performance while also reducing mass when compared to conventional steel. Most of the LCAs reviewed demonstrated that aluminum-intensive designs were able to achieve the largest reductions in life-cycle energy use and GHG impacts, specifically in the use phase, but HSS and AHSS achieve reductions through vehicle mass reduction options that offer different material properties enabling optimized multi-material designs.

Kelly et al. and Mayyas et al. found that AHSS reduced production and use-phase impacts relative to baseline components and vehicles (Kelly et al., 2015; Mayyas et al., 2012c). Unlike aluminum, AHSS contributes fewer GHG emissions than conventional steel from the beginning, not requiring a payback period during the use phase. Many of the multi-material designs assessed in the selected studies use HSS/AHSS (Bush et al., 2015; Das, 2014a; Dhir and Das, 2014; Tempelman, 2011). The application of steel alloys for reducing mass plays an important role in comprehensive designs, including two different aluminum-intensive designs evaluated in the LCAs (Bush et al., 2015; Das, 2014b). Steel plays an important role in the chassis, body, interior, and powertrain of these designs and lends itself well to the multi-material concept because it can be separated by magnets during EOL, enhancing the recovery rate for recycling (Ehrenberger, 2013; Steel Market Development Institute, 2016). Secondary steel production requires half as much energy with 13 MJ/kg compared to virgin steel production at 26 MJ/kg (Lewis et al., 2014). Generally, the steel recycling scenarios were based on current practices and ranged from 85% to 98% (Koffler, 2013; Mayyas et al., 2012c; Raue et al., 2015; Stasinopoulos et al., 2012; Steel Market Development Institute, 2016), while the aluminum recycling scenarios were based on theoretical scenarios exploring the expansion of aluminum recycling to achieve similar rates.

Unlike the other LCAs, the study by the Steel Market Development Institute found that AHSS had the lower life cycle energy and GHG impacts compared to aluminum as a material for vehicle mass reduction (Steel Market Development Institute, 2016). This finding was also true for two 'case studies' by World Auto Steel, although these studies were not included in the selected LCAs in Table 1 because they were based heavily on variable selection within a specific tool and lacked sufficient rationales for the selected inventories or explanation of the methodologies used (World Auto Steel, 2014a,b). While these findings

are not consistent with other reviewed LCAs that compare AHSS/HSS to aluminum, the Steel Market Development Institute used a publicly available model and data from both the steel and aluminum industries when possible. The differentiation between the findings of this LCA and others is the result of divergent assumptions.

The Steel Market Development Institute's (SMDI) study using the University of California Santa Barbara (UCSB) Automotive Materials GHG Comparison Model V4 includes material replacement coefficients (MRCs, also known as material substitution rates) based on values from A2Mac1 (an automotive benchmarking company that conducts vehicle teardown studies) and reports published by the Society of Automotive Engineers (SAE). These values, while in the range of plausibility, may lower the impact of steel compared to the range of values available in the literature (Kelly et al., 2015; Kim and Wallington, 2013a). The report cites a 2013 report by the Aluminum Association for CO₂ per kilogram of aluminum ingot production and uses the default value for steel production provided by the UCSB model (citing a 2010 World Steel Association report) to represent AHSS which is also credited for reducing weight compared to conventional steel. Additionally, for aluminum (MRC range of 0.6–0.7) the average MRC was used (0.65), but for steel (MRC range of 0.7–0.85) an MRC value that was 0.03 (3.9%) lower than the average was chosen (0.75) (Steel Market Development Institute, 2016). The rationale given in the SMDI study is that aluminum has a decreasing ability to reduce mass as vehicle designs advance and AHSS is expected to make additional advances in technology; however, there are studies that highlight advancements being made in aluminum alloys for weight reduction and strength (Autocar Pro News Desk, 2016).

The fuel reduction values (FRVs), which estimate the fuel savings based on weight reduction, used in the SMDI study were averages of estimates for vehicles with and without powertrain reductions. The starting values used in those estimates were low compared to other values in the literature, 0.094–0.112 for no powertrain reductions and 0.252–0.325 with powertrain reductions. Low FRVs translate to lower fuel efficiency gains from mass reduction, subsequently shifting more emphasis on production impacts. For recycling rates in the SMDI study, the value for the lower limit was chosen for modeling rather than the mean or median. The range for aluminum was 79–95% yielding an average of 87% while the range for steel was 90–95%, yielding an average of 92.5%. Thus, using the lower values (79% and 90%, a difference of 11 points) increases the difference between aluminum and steel, when compared to the averages (87 and 92.5, a difference of 5.5 points). The assumptions used for recycling more likely represent current recycling rates rather than hypothetical future scenarios where aluminum recycling would be increased. The stochastic comparison of thirteen scenarios using Monte Carlo Analysis (MCA) shows AHSS outperforming aluminum 0.6–6.8% of the time with 5000 runs per scenario (Steel Market Development Institute, 2016). The values applied as assumptions in the SMDI study may fall into plausible ranges but consistently represent factors that generate favorable results for steel over aluminum. Assumptions from reports that have divergent results from the majority of the literature need to be carefully analyzed so that they can inform rather than detract from the larger scientific consensus.

All of the LCAs of steel and aluminum reviewed show that both materials can be effective technologies for reducing vehicle mass and that material replacement and recycling rates are critical for the total life-cycle impacts of vehicles and their components.

2.3. Magnesium

Magnesium alloys also provide the opportunity for mass reduction in vehicle production. In multi-material designs, it can be used in targeted applications as multiple components can be combined into one lighter weight casting. The magnesium production processes are different from steel, which is often extruded or stamped and welded compared with magnesium that is primarily cast. Magnesium is lighter than aluminum, but in the majority of LCAs it has higher life cycle GHG impacts due to high production impacts and the lack of recycling. In most cases magnesium production was modeled to represent the Pidgeon process of magnesium production in China. The modeling of the Pidgeon process often included the use of sulfur hexafluoride (SF₆) as a cover gas which turns out to be very significant in the life-cycle GHG impacts associated with magnesium (Dhingra and Das, 2014; Rauegi et al., 2015; Tharumarajah and Koltun, 2010). SF₆ is the most potent GHG with a 100-year global warming potential 23,900 times greater than CO₂ and an estimated atmospheric life of 3200 years (Intergovernmental Panel on Climate Change (IPCC), 2007; Rauegi et al., 2015). Although the weight reductions result in greater reductions during the use phase, the difference between aluminum and magnesium is not significant enough to warrant the impacts associated with magnesium production, particularly the use of SF₆ (Dubreuil et al., 2012).

Many of the studies discuss the phase-out of SF₆ in favor of less climate-damaging cover gases such as SO₂ (Kelly et al., 2015; Li et al., 2015; Rauegi et al., 2015) or R134a (Ehrenberger, 2013; Rauegi et al., 2015; Tharumarajah and Koltun, 2010). Li et al. and Dubreuil et al. found that even with the use of SO₂ as a cover gas, the magnesium industry would need to rely on other technology improvements to provide life-cycle GHG emissions reductions relative to aluminum for vehicle mass reduction (Dubreuil et al., 2012; Li et al., 2015). Dubreuil et al. also highlighted the importance of recycling to reduce the impacts associated with the use of magnesium components and the need for more significant market penetration of magnesium components in order to achieve economically viable magnesium recovery at EOL. This is separate from its use in aluminum at EOL, where low amounts of magnesium are recycled with aluminum to provide material for aluminum alloys (Dubreuil et al., 2012). Magnesium has major production impacts but, in multi-material designs, it can be used strategically to attain significant powertrain or secondary mass reductions (Bush et al., 2015; Dubreuil et al., 2012; Mayyas et al., 2012c). Considering the current production processes for magnesium, the use phase emissions reductions do not offset the production phase emissions. Improvements in the cover gases and other magnesium production processes need to be implemented in order for magnesium to compete with other mass reduction technologies on a life cycle basis. LCAs need to be responsive to the changing magnesium industry, modeling the most current production technologies to provide useful feedback to the industry.

2.4. Composites and plastics

The composites evaluated in the selected LCAs are largely CFRP and GFRP (Bush et al., 2015; Das, 2011; Kelly et al., 2015; Koffler, 2013; Mayyas et al., 2012c; PE International, 2012a,b; Raugei et al., 2015; Simões et al., 2016; Tempelman, 2011; Tharumarajah and Koltun, 2010; Witik et al., 2011), but there was also an assessment comparing a polyphenylene ether resin-based polymer fender with a steel fender (Baroth and McKay, 2012). All of the lightweight polymers have lower impacts than steel in the use phase. GRFP was generally found to reduce both production and use-phase GHGs and energy impacts compared to steel. The polyphenylene polymer fender had similar production-phase energy use, 732 MJ vs 735 MJ for the steel alternative, and 18% lower production-phase GHG impacts, but resulted in a significantly greater 47% reduction in overall life-cycle GHGs and a 45% reduction in energy demand compared to the steel fender (Baroth and McKay, 2012). Carbon fibers have significant barriers due to high production impacts. Das notes that while “CFRP offers weight reduction potential in the range of 50–60% compared to conventional steel,” the energy intensity and cost limit its potential to scale up (Das, 2011).

Due to the significant weight reduction and strength of CRFP, it has seen some use in high-end and high-performance vehicles where vehicle dynamics and acceleration typically take precedence over emissions reductions. Two examples (not found in the LCAs) are the BMW i3 and i8 plug-in hybrid and electric vehicles, which utilize CFRP passenger cells with an aluminum chassis (BMW, 2016), and the Lamborghini Huracan, which utilizes a hybrid CFRP and aluminum frame (Automobili Lamborghini, 2016). Steel, which is largely recycled, tends to have lower impacts than composites at end-of-life due to the difficulty of recycling composite materials.

2.5. Multi-Material and Holistic Designs

Six of the reviewed LCAs evaluated multi-material systems through assessments for the BIW or a complete vehicle design. The multi-material systems are more holistic designs based on automotive engineering principles and making extensive use of computer aided design to meet the functional demands of light-duty vehicles. Four of the six studies concluded that aluminum intensive designs had lower impacts relative to HSS intensive designs with regard to both life-cycle energy use and GHGs (Bush et al., 2015; Das, 2014b; Mayyas et al., 2012c; Raugei et al., 2015), while one of the studies evaluated energy use only and found that an aluminum BIW had a 3.1–3.4% life cycle energy reduction compared to a steel/HSS baseline (Tempelman, 2011). The remaining study, Kim et al., found that the primary mass reduction provided by an aluminum intensive design could be matched only by HSS when secondary mass reduction was allocated to the HSS design alone, but that inclusion of secondary, recycled aluminum in vehicle production is necessary to offset production emissions and, until robust aluminum recycling is in place, HSS intensive designs may play an important role in reducing emissions (Kim et al., 2010).

Tempelman et al. and Raugei et al. found that composite inclusive designs significantly reduced life cycle impacts compared to HSS designs (Raugei et al., 2015; Tempelman, 2011), while Mayyas et al. found that a composite based BIW performed worst when compared to AHSS, aluminum, magnesium and the steel baseline (2012c). Similarly,

magnesium showed mixed results with Mayyas et al. finding it to have lower life cycle impacts compared to AHSS (2012c), while Raugei et al. found magnesium to have higher life cycle impacts than the steel benchmark (2015). The collective findings of these studies make it clear that a complete vehicle design incorporates many different materials but that aluminum-focused designs may provide the greatest life-cycle improvements.

3. CRITICAL FACTORS IN A VEHICLE MASS REDUCTION LCA

The following sections describe different factors that are assumptions commonly found in LCAs of vehicle mass reduction, their influence on the results, and insights from the literature regarding these factors. Based on the studies reviewed, key factors driving the life cycle impacts of different material choices were the following: secondary mass reduction, recycling allocation, powertrain reductions and FRVs, material substitution rates, lifetime vehicle travel distances, and production location and grid mix (Table 2). The first three discussed are factors used in calculations to modify scenarios for modeling purposes. The uncertainty associated with secondary mass reduction and FRVs could be avoided by using data from an actual test vehicle, but, when applied as variables in LCA, vary widely and have significant impacts on results. The last three factors are based largely on the underlying trends and characteristics of the system, but may also be associated with consequential shifts within a changing industry.

3.1. Secondary Mass Reduction

Secondary mass reduction provides mass savings in addition to the initial, or primary, mass reductions. The vehicle efficiency is further improved by downsizing the powertrain and other systems in vehicles which are sized, in part, based on the total vehicle mass and thus yield secondary mass savings. These systems include the suspension, structural components (subframes, attachment points), transmission, engine, fuel and exhaust systems, tires and wheels, steering and brakes, and exterior. Secondary mass savings is a critical factor in designing a vehicle in order to maximize vehicle mass reduction efforts. In a 2013 report on secondary mass reduction by World Auto Steel, the authors found through a review of published literature that secondary mass reduction estimates vary widely, from 0.16 to 1.0 kg of secondary mass savings per kilogram of primary mass reductions (Malen et al., 2013). In comprehensive BIW lightweight designs, it can be difficult to distinguish between 'primary' and 'secondary' mass savings because design optimization and material replacement can happen at the same time. As a result, secondary mass reduction is a useful concept for designers or in assessments of holistic mass reduction efforts, but estimates of secondary savings are highly uncertain and make it difficult to allocate secondary mass reductions, creating wide ranging definitional differences.

Table 3 shows which of the selected LCAs included secondary mass reduction and the factor that was used to calculate the additional weight reduction. The use of secondary mass reduction calculations in LCA is an assumption that the mass reduction can be leveraged in vehicle design. In recognition of the consequential nature of secondary mass reduction, Lewis et al. provide an assessment with and without secondary mass reduction, which they state must occur early in the design process (Lewis et al., 2014). Other authors avoid the

allocation of secondary mass reduction, stating that there is “lack of agreement in the literature” (Raugei et al., 2015) and that it is “highly speculative” (Kelly et al., 2015). Table 3 shows values for secondary mass reduction used in the reviewed LCAs that range from 5% to 70%. The combined significance, both in regards to goals for mass reduction efforts and ability to decrease life cycle impacts, and uncertainty of secondary mass reduction as a factor in LCA suggests that scenario analysis, stochastic modeling, and bounding studies will help address any uncertainty.

The use of secondary mass reductions may not be appropriate for standalone components if the entire vehicle is not undergoing comprehensive mass reduction as part of an initial model design or as part of a competitive model redesign cycle. In LCAs that do use secondary mass reduction calculations, the results will provide additional benefits for lighter components during the use-phase. As described by Koffler:

Downsizing components like engines, transmissions, brakes, etc. is not a continuous exercise where any percentage weight reduction can be directly transformed into secondary mass changes, but rather a step function whose characteristics depend on the specific components available to the OEMs from their product portfolio or suppliers. (Koffler, 2013)

There are LCAs that do evaluate system wide design changes to achieve mass reduction goals, such as the Mach-I vehicle design assessed by Bushi et al. in which test vehicles were actually built (2015), or the virtual designs created and optimized by EDAG Engineering, assessed by Das (2014b). As previously mentioned, in studies such as these, in which an entire vehicle is assessed, the secondary mass reduction may be inherent to the LCI for the vehicle. This avoids the use of additional calculations and assumptions that may introduce high levels of uncertainty. In other theoretical redesigns such as in Lewis et al. and Ding et al., a secondary mass reduction figure may aid in evaluating systemic changes because these works are based on major reductions in bulky vehicle components, but lack data for the systems that are likely to have secondary mass reductions (Ding et al., 2016; Lewis et al., 2014).

3.2. Recycling Allocation

The GHG and energy benefits of recycling are significant, and the allocation of these benefits can influence the results of LCAs. The energy and GHG reductions associated with recycling can be allocated (or attributed) to the materials after they are used in products and then eventually supply recyclable material to the market, the EOL approach. Alternatively, the reductions can be allocated based on the recycled content of products that make use of secondary materials, the recycled content approach. The allocation of recycling for metals in LCA has been debated for many years and is thoroughly discussed in the context of vehicle mass reduction by Liu and Müller (2012) as well as Kim and Wallington (2013a).

The specific EOL approach endorsed by the International Council on Mining & Metals (ICMM) in the 2007 Declaration by the Metals Industry on Recycling Principles by Atherton (2007), is simply the recycling potential for a material. Geyer refers to this as consequential system expansion and models it through a material balance in the scrap

market (2008). This approach may overestimate real-world recycling rates and is challenged in the response to the ICMC paper by McMillan et al.:

... the blanket statement that recycling displaces primary production ignores the effects of the economics of removing contamination accumulated during recycling and the different tolerances for contamination across alloys.

and

The combination of modest recycling rates (similar to municipal paper and paperboard products), the sustained demand for metal-containing products, and the likelihood that metals are recycled only a small number of times before ultimate disposal indicate the inevitability of primary metal production and associated emissions. (McMillan et al., 2012)

A recycled content approach may result in scenarios that are more representative of the current dynamics in the recycling market while an EOL approach can provide a recycling potential that may be more useful in projecting scenarios for future recycling rates.

For many of the lightweight materials that do not currently have a large market share such as CFRP, the LCAs assume low or no recycling. This may be representative of the current status for newer products, but may not be the case in the future as new methods of recycling are developed (Gardiner, 2014). In the time it takes for lightweight vehicles to be designed, produced, go through the use phase, and enter EOL, a robust system of recycling for lightweight materials may already be well established (Keoleian and Sullivan, 2012; Kozdras et al., 2016; Raugei et al., 2015; Sverdrup et al., 2015a, b).

When higher recycling rates in LCA are used for all materials, lightweight materials with greater production-phase impacts such as aluminum are favored compared to conventional steel because the offset of virgin production accounts for a larger percentage of the total life-cycle impacts associated with the material. When lower recycling rates are used for all materials, it does not help reduce the total impacts for any material. However, compared to other material options such as aluminum, lower recycling rates favor materials such as GFRP and AHSS/UHSS that have comparatively low production impacts while still being lightweight. Table 3 shows the recycling approach used by each of the selected LCAs.

3.3. Powertrain Reductions & Fuel Reduction Values

Powertrain downsizing decreases the fuel consumption of an engine due to reductions in cylinder displacement and/or cylinder count. While it can contribute to secondary mass reduction, the calculation of powertrain downsizing is a separate factor for efficiency gains during the use phase in LCA. The powertrain and mass reduction form a ratio that result in the use phase performance profile of a vehicle. Given a lower weight and a fixed powertrain a vehicle will have increased use phase performance. Powertrain downsizing is often assumed in LCAs so that the power to weight ratio remains the same before and after reducing the mass of a vehicle. This downsizing and the mass reduction itself result in improved fuel economy for lower mass vehicles within a vehicle segment, but there are significant uncertainties when accounting for these reductions in LCA (Carvalho et al., 2013). To calculate the fuel savings associated with the changes to a reduced mass vehicle, a

fuel reduction value (FRV) is often used. Eleven of the LCAs reviewed cite the method published in 2010 by Koffler and Rohde-Brandenburger (Bush et al., 2015; Das, 2014b; Dubreuil et al., 2012; Ehrenberger, 2013; Kelly et al., 2015; Koffler, 2013; Marretta et al., 2012; PE International, 2012a, b; Rauei et al., 2015; Simões et al., 2016). High and low FRVs are often used in scenarios to represent the effects of mass reduction with and without powertrain downsizing. Table 3 shows the FRVs for the selected LCAs. FRVs are most commonly reported in L/(100 km*100 kg), which is a measure of how much fuel is saved per 100 km given a 100 kg reduction in vehicle mass. Some of the selected LCAs used alternative methods for estimating fuel savings, while others stated that the savings were accounted for but did not provide a value for their assumptions (see Table 3).

Kim et al. provide a detailed discussion of the methods used to calculate FRVs and provide an accounting of FRV in the literature (Kim and Wallington, 2013a, b; Kim et al., 2015). They also recommend the use of an improved method of mass-induced fuel consumption to account for mechanical efficiencies due to mass reduction in addition to traditional FRV calculations to achieve more accurate calculations for energy savings in vehicle component LCAs (Kim and Wallington, 2013b; Kim et al., 2015). Kelly et al. demonstrated the significance of varying the FRV from 0.15 to 0.50 L/(100 km*100 kg) for different mass reduction options (2015). Their work showed that the materials with higher production impacts are most dependent upon achieving higher rates of efficiency during the use-phase to achieve a realistic breakeven driving distance, which is the point at which a vehicle achieves enough use-phase reductions to offset the additional impacts associated with production when compared to a baseline.

3.4. Material Substitution Rates

Material substitution rates are used to calculate the mass of a lightweight substitute based on a percentage of the mass of the material being replaced. These values are often generalized assumptions utilized in lieu of vehicle component masses that are known. The lower the material substitution value, the greater the mass savings potential of the alternative material. While the average mass of a unit of aluminum divided by the average mass of a unit of steel may provide a factor for estimating mass savings when aluminum replaces steel, this does not account for differences in alloy densities for either metal or component design modifications that may occur throughout a vehicle design. Additionally, due to differing material properties, the lightweight material may be used in different volumes than the materials that are being replaced, which some studies account for while others do not.

LCAs can avoid the uncertainty associated with generalized material substitution rates by modeling known components, identifying component-specific substitution rates, or parameterizing the substitution rates using existing data, all of which were methods used in the selected LCAs. Table 3 shows the material substitution rates used in the selected LCAs. Some of these values were calculated based upon the weights provided in the studies. In addition to the material substitution rates from the selected LCAs in Table 3, Kelly et al. and Kim and Wallington provide a range of substitution factors used within the literature that are helpful to create a range or distribution that could be used for stochastic modeling if component-specific data is lacking (Kelly et al., 2015; Kim and Wallington, 2013a).

3.5. Lifetime Vehicle Travel Distance

The lifetime vehicle travel distance is a factor used in the calculation of use-phase impacts. Because vehicle mass reduction is inherently focused on reducing use-phase impacts, this value is a critical and unavoidable assumption in modeling the life-cycle impacts of a vehicle. Lower lifetime travel distances favor materials with low production-phase impacts and higher mass such as AHSS/UHSS compared to high production-phase impacts and lower mass such as aluminum or CFRP. Some studies also use breakeven driving distances in addition to lifetime distances. A breakeven distance is when additional production impacts are made up for in savings that are accumulated during the use phase. Some materials such as AHSS/UHSS may not have a breakeven driving distance when the production impacts are lower than that of the material it is replacing (Das, 2014a; Kelly et al., 2015; Steel Market Development Institute, 2016). In this case, the benefits of mass reduction coincide with material reductions, and impacts are reduced in both the production phase and use phase.

The selected LCAs had wide ranging lifetime vehicle travel distances with the maximum assumption more than doubling the minimum, from 150,000 km (Raugei et al., 2015) to 322,000 km (Mayyas et al., 2012c). The values for the lifetime vehicle distances of the selected LCAs are shown in Fig. 4. The figure shows two distinct groupings, one at 190,000 km to 210,000 km and another 231,000 km to 250,000 km, the latter of which corresponds to the useful life of 241,000 km (150,000 miles) used for regulating light-duty vehicles in the US (U.S. EPA, 2014). As noted by Kim and Wallington (2013a), there are also major regional differences associated with lifetime vehicle travel distances with Japanese vehicles being modeled with less than half the driving distance, 92,000 km (57,000 miles) to 100,000 km (62,000 miles), compared to US vehicles, 190,000 km (120,000 miles) to 290,000 km (180,000 miles). The LCAs reviewed often did not provide a justification for the lifetime vehicle travel distance assumption despite the importance of comparing total life-cycle impacts for vehicle mass reduction and the wide variations found within the literature. The previously discussed breakeven range for GHG emissions of 19,000 km (Das, 2014a) to 160,000 km (Mayyas et al., 2012c) borders on the range of lifetime vehicle driving distances (Table 3 and Fig. 4). The hypothetical range of 0 to 200,000 km published by Kelly et al. (2015) captures the driving distance variable used in fourteen of the twenty-six (54%) LCAs. Liu and Müller in their 2012 review of the aluminum industry note that lifetime driving distance is a key assumption which often differs by industry, with the aluminum industry often using longer distances and the steel industry using shorter distances, which results in contradictory conclusions in LCA. Lifetime vehicle driving distances are also likely to change based on shifts in travel behavior and vehicle ownership trends which adds another layer of uncertainty when assessments model new designs which will be driven in the future.

3.6. Production Location and Grid Mix

Location is a critical aspect for production-phase impacts associated with energy intensive metallurgical processes because there are significant differences in power generation technologies and production methods that result in drastically different production impacts. Aluminum and steel both have a production base in North America, but with increasing pressure from Chinese trade, there is a great deal of uncertainty around the source of these commodities in a global market (Buckley and Perlez, 2016; Reed and Bradsher, 2016).

As previously mentioned, many of the LCAs assumed an electric grid mix with a high percentage of hydroelectric power generation for the production of aluminum (Bush et al., 2015; Das, 2014b; Raue et al., 2014; Raue et al., 2015; The Aluminum Association, 2013). That assumption is based on existing North American production, but is challenged by the consequential increase in demand if more aluminum is used and the potential for China to supply cheaper aluminum. The increase in demand would require additional aluminum manufacturing with new facilities using an available electric grid mix and not necessarily a hydroelectric or other renewable electric-dominant source, unless additional renewable electric facilities were built in conjunction (Weidema et al., 1999). Likewise, if Chinese production fills a significant fraction of the gap between supply and demand, the aluminum used for vehicle manufacturing will be sourced from a primarily coal-based electricity production system which also requires additional raw material shipping from China (Henriksson et al., 2014). Experts in the aluminum industry said that the new state-of-the-art Chinese coal-based electric smelting facilities now make up 65 % of global aluminum production capacity and generate 17.5 kg CO₂ eq. per kg aluminum compared to the 20.1 kg CO₂ eq. of older, less efficient, North American coal-based electric smelting production and the 4.8 kg CO₂ eq. of North American hydroelectric power based facilities (personal communication with Kaiser Aluminum). Steel, being needed in lower volumes but higher grades relative to non-lightweight designs, may have distinctly different market dynamics. Magnesium is different in that the majority is produced in China using the Pidgeon process, but the electrolytic production method used in Australia and the United States may have very different production-phase impacts. There are, however, limited data on the electrolytic production of magnesium (M.C. Johnson, 2014).

The electric grid mix and the associated emission rates for vehicle mass reduction LCAs should explicitly represent the actual or anticipated production locations for lightweight materials based on well-defined functional units. If the goal is to baseline an existing design, the current location for production and its source of electric power generation may be the most appropriate. If, on the other hand, an aluminum-intensive design which is being proposed for fleet-wide integration is being modeled, it makes sense to evaluate scenarios with alternative production pathways to better illustrate the possible market futures as a way of evaluating uncertainty. Grid mix and production location are assumptions that should be carefully selected and made explicit. Projecting GHG and other emissions from electric generation into the future can also be problematic and may require complex dispatch modeling to properly capture the impacts of current and future regulations, grid interconnections, and introduction of new technology on future electricity production and distribution in a country or region.

4. DISCUSSION

4.1. Vehicles and Fleet Dynamics

Policies aimed at reducing tailpipe emissions from light-duty vehicles such as the U.S. Light-duty Vehicle GHG and CAFE Standards are focused on curtailing use-phase emissions. The emissions associated with the use-phase are dominant drivers of the total life-cycle energy and GHG impacts for most vehicles today, but when vehicle mass

reduction options provide comparable use-phase benefits, the impacts from production and EOL can help distinguish which materials achieve lower environmental impacts over the complete life-cycle. Additionally, factors such as the location of production and use can be used in LCA to place a vehicle in an operational context to more accurately model life-cycle impacts. Modaresi et al. address the expansion of LCA beyond single vehicles to capture larger dynamics associated with material choices:

...one should move beyond single-product LCAs and consider the entire vehicle fleet, its development over time, and its connection to the material industries. Only by assessing emissions reduction strategies on the full scale and over time, the future impact of emergent phenomena, such as material recycling, can be correctly estimated. This allows for coupling policies on use phase emissions reductions to those addressing emissions in material producing industries. (Modaresi et al., 2014)

The ‘dynamic fleet’ approach they recommend may help define impacts over time for fleet level impacts associated not only with recycling, which was their focus, but also with fleet turnover rates and demand for virgin materials. A fleet-based perspective may help LCAs better define some of the critical factors highlighted in this review.

The effort to reduce vehicle mass for a specific make and model occurs in the broader context of product development and fleet turnover dynamics, which buffer the effects of change. Although manufacturers can rapidly bring a product to market and sell it in high volume, it takes several decades for vehicle mass reduction to take full effect by replacing the existing vehicle stock over time. According to Modaresi et al., this time lag is due to the delay mechanisms associated with the application of mass reduction to affect all new vehicles and the replacement of existing stock with the lighter weight vehicles (2014). As with any technology-based efficiency improvement, the fleet-wide benefits of mass reduction will be incremental, but through the investments in new materials and designs, vehicle mass reduction is a promising step towards achieving the new federal standards (Nehuis et al., 2014; U.S. EPA, 2012a, 2014). According to the National Highway Traffic Safety Administration (NHTSA) and the Environmental Protection Agency (EPA) these new standards “will result in MY 2025 light-duty vehicles with nearly double the fuel economy, and approximately one-half of the GHG emissions compared to MY 2010 vehicles—representing the most significant federal actions ever taken to reduce GHG emissions and improve fuel economy in the U.S” (U.S. EPA and NHTSA, 2012).

A shift towards more fuel efficient vehicles with lower use-phase impacts means that production becomes a larger relative share of the overall life-cycle impacts (Fig. 1). A future increase in the sales of battery and plug-in hybrid electric vehicles could make this more important as the use-phase emissions for those vehicles are no longer predominantly tailpipe emissions, but instead result in part from electricity generation. Additionally, the generation mix and GHG emissions of the electricity consumed can change over a specific vehicle model’s lifetime, and is also sensitive to the assumptions regarding where and when that vehicle is charged. Cleaner sources of energy for the use-phase of vehicles may make the material management associated with the production and EOL of a vehicle a larger determinant of life-cycle impacts, emphasizing the importance of material selection and robust recycling systems (Keoleian and Sullivan, 2012). These circumstances may favor the

materials with lower production impacts that can still provide vehicle mass reduction benefits such as AHSS/UHSS rather than those that may be incurring more production-phase impacts to gain use-phase reductions like CFRP and Mg under their current production paradigms.

4.2. Costs

The costs associated with vehicle mass reduction include not only the difference between the original components and the mass-reduced components but also the offset in lifetime fuel costs to consumers as well as broader social costs and benefits. While the LCAs reviewed here were focused on the life-cycle GHG and energy impacts, cost is an important consideration as well. In particular, the cost of production and total cost of ownership to the vehicle owner, as these factors may affect the consumer acceptance and fleet turnover. It also influences the cost of mass reduction in the context of other vehicle GHG reduction strategies. In an assessment of mass reduction for the EU, Meszler et al. found that mass reduction is a cost effective strategy to reduce carbon emissions (2013). European standards are on a per-vehicle-mass basis that deemphasizes vehicle mass reduction by shifting the vehicle into categories with lower CO₂ standards as mass is reduced, compared with the U.S. approach that is on a vehicle footprint basis (National Highway Traffic Safety Administration, 2012). Meszler et al. also found that the cost of meeting emissions reduction goals would be twice as high if mass reduction was not one of the available options for automobile manufacturers (2013). Hofer et al. found that “light weighting has the potential to lower vehicle costs, however, the results are very sensitive to parameters affecting lifetime fuel costs for conventional and battery costs for electric vehicles.” They found an optimum cost reduction for conventional and electric vehicles using a primary mass reduction of 22 % to 39 % (Hofer et al., 2014). The higher the fuel costs, the more cost effective vehicle mass reduction can be, since it increases efficiency during the use phase. Similar to electric vehicles, conventionally fueled low-mass vehicles may cost more initially but have the potential to provide savings throughout the operational life of the vehicle.

4.3. Production, Material Flows, and Advances in Technology

The future of vehicle manufacturing is heavily dependent upon the available materials, and regulatory efforts are steering designers to focus on vehicle mass reduction because it is one of the most cost effective ways to increase efficiency while maintaining or improving the performance and utility expected by consumers. Therefore, the suite of lightweight alloys and polymers available to designers will continue to grow. There is a large trend towards aluminum because of its mass reducing potential, cost, and performance, which is represented both in the academic literature and through R&D investments by the industry (Automobili Lamborghini, 2016; BMW, 2016; Bushi et al., 2015; Caffrey et al., 2015; Ducker Worldwide, 2014; Gina-Marie Cheeseman, 2016; Hirsch, 2011; Modaresi et al., 2014). Ford, Mercedes, BMW, Jaguar-Land Rover, and Tesla are all integrating more aluminum into their vehicles with Ducker Worldwide estimating that 17.5 million vehicles produced in 2015 contain nearly 3 billion kg (7 billion lbs) of aluminum, a 28 % increase over the aluminum content of vehicles produced in 2012 (Ducker Worldwide, 2014).

In addition to more aluminum being used per vehicle, the light-duty fleet in general is growing world-wide, driving a net increase in demand for aluminum and steel (Modaresi et al., 2014). The transportation sector is already the largest market for aluminum in North America, and the market share for aluminum is expected to expand (The Aluminum Association, 2011). With billions of kilograms of aluminum going into new vehicle stock each year and a projected global rate of 16 billion kg (35 billion lbs) per year with 248 kg (547 lbs) of aluminum in the average North American light-duty vehicle by 2025 (Ducker Worldwide, 2014), there is some concern within the literature that aluminum production will not be able to keep pace with these rates of demand. Sverdrup et al. state that aluminum will become more scarce by the end of the century and that the decline of available virgin metal highlights the importance of robust recycling systems for aluminum (Sverdrup et al., 2015a).

The availability of secondary aluminum will lag behind the expanded use of aluminum as the existing vehicle stock turns over and lightweight vehicles enter the EOL-phase (Liu and Müller, 2012; Modaresi et al., 2014). An increase in prices would be expected as demand for new aluminum grows during the transition towards lightweight vehicles and, not surprisingly, higher prices are associated with an increase in collection rates for recycling (Chen, 2013). The cost of aluminum may promote investments in more robust collection and recycling systems, but it may also push vehicle manufacturers to other mass reduction options as cost optimizations shift in favor of other materials. The efficacy of aluminum recycling systems will be critical to keeping long-term prices down and achieving additional emissions reductions (Modaresi et al., 2014). Ford Motor Company has recently demonstrated the potential of closed loop systems that recapture 30-40 % of the aluminum used in their F-150 production process through scrap recycling (Ford Motor Company, 2016). The new procedures and technologies employed in recycling systems have the potential to increase overall collection rates as well as maintain the purity of the scrap streams by differentiating alloys and processing them accordingly (Paraskevas et al., 2015).

The North American aluminum industry has been optimizing their production process to improve environmental performance. As described by the Aluminum Association in their 2013 update for life-cycle inventories of North American production, as of 2010, approximately 95 % of primary aluminum is produced using prebake facilities that have replaced the older, less efficient Soderberg facilities. In addition to the move towards more efficient production, the industry has also shifted away from coal-fired electric generation, while maintaining their cleaner hydroelectric generation facilities resulting in a hydroelectric-dominant energy mix (The Aluminum Association, 2013). For example, in 2015, Rio Tinto's refurbished Kitimat facility started producing aluminum using their existing hydroelectric generation facility and a new proprietary smelting technology to increase aluminum production by 48 % and "effectively halve the smelter's overall emissions" (Rio Tinto, 2015). The production impacts from the North American aluminum industry have decreased due to reductions in energy inputs and the closure of higher polluting coal-fired thermoelectric facilities. In addition to production efficiency, aluminum manufacturers such as ALCOA are now developing stronger alloys that may broaden the potential applications for aluminum and enable additional mass reductions (Autocar Pro News Desk, 2016). Despite these gains, if vehicle manufacturers source the aluminum for

their production needs from China's coal-based aluminum industry, the emissions associated with production are likely to increase.

Because the current aluminum LCIs represent a largely optimized industry, there may be a greater potential for materials with suboptimal production such as magnesium and CFRP to achieve additional reductions in the production phase that will improve their relative life-cycle performance. Similar to steel and aluminum, there are methods for magnesium production that can decrease production-phase impacts and alloys being developed that can improve the manufacturing and performance in light-duty vehicle applications (Dubreuil et al., 2012; Easton et al., 2008; Ehrenberger, 2013). Magnesium components, which are not separated from aluminum at EOL, are currently recycled through other metal manufacturing, such as for secondary aluminum alloys (personal communication with Kaiser Aluminum) (Dubreuil et al., 2012). To date, the volume of magnesium component recycling has not been sufficient to warrant a separate industry to recover magnesium during EOL, compared to a closed loop recycling process at companies such as AMACOR, in which scrap is recovered from the production process and reused. New processes for magnesium recovery by vehicle dismantlers will need to account for material purity and value in order to create an economically viable recycling system for magnesium at EOL (personal communication with AMARCOR).

According to Mayyas et al., composites may be less preferable from a sustainability perspective due to high initial costs, low recyclability, and low durability (2013). These factors, however, are being improved through R&D and the application of CFRP for vehicle mass reduction as well as concerted efforts across multiple industries to develop EOL solutions to reuse or recycle glass and carbon fibers (European Alliance for SMC/BMC, 2011; Gardiner, 2014; Tempelman, 2011). Given the current focus on reducing use-phase emissions, lightweight materials such as CFRP and magnesium, which have higher life cycle GHG emissions than high strength steels or aluminum, will be used in vehicles to achieve emissions reductions despite overall emissions being higher due to elevated production impacts. Steel remains as the primary competitor for aluminum in providing lightweight material options, and improvements in steel alloys continue to promise cost-effective mass reduction which may not face as many barriers with regards to production capacity, cost, or EOL processing when compared to aluminum or other vehicle mass reduction options (Ducker Worldwide, 2015; Lewis et al., 2014; Modaresi et al., 2014; Stasinopoulos and Compston, 2014; Tempelman, 2011).

4.4. LCA for Vehicle Mass Reduction

LCAs depend on inventory data for their accuracy, and the inventory data are often generated through retrospective evaluations of existing technologies. There are periodically updates for specific products or industries (Hakamada et al., 2007; Johnson and S, 2014; The Aluminum Association, 2013), but often practitioners are using data from large sets such as ecoinvent, which includes inventories that are more than a decade old (Frischknecht et al., 2005; Weidema et al., 2013). Even with the most up-to-date inventories, the use of retrospective data is problematic for the dynamic modeling of nascent materials and technologies (Wender et al., 2014a, 2014b). Materials for vehicle mass reduction are being

developed and deployed rapidly to help manufacturers meet new efficiency and emissions standards. LCAs can help guide this development by using appropriate assumptions and methods that explore sensitivity and uncertainty regarding processes and factors used to calculate impacts, which is especially true for consequential impacts associated with material flows in the light-duty vehicle fleet (Du et al., 2010b; Modaresi et al., 2014; Stasinopoulos and Compston, 2014; Stasinopoulos et al., 2012). The wide ranging hypothetical assessment carried out by Kelly et al. demonstrates the breadth of the possible outcomes, which show overlapping ranges of breakeven driving distances for most mass reduction technologies given just some of the uncertainty for variables found within the literature, namely material substitution rates, production emissions, and FRV (Kelly et al., 2015). Complete designs through evaluation of concept cars such as the study by Bushi et al. (2015) or engineering models like the one used by Das (2014a) enable more prospective assessments that can help inform future vehicle designs.

Production location, electric grid mixes, recycling, and even lifetime vehicle travel distances are examples of critical factors in vehicle mass reduction LCAs. These factors are influenced by consequential impacts that can be associated with marginal production, temporal shifts, or both, which are important because vehicles are long-lived and represent a major manufacturing industry. Even a minor change in the material selection for light-duty vehicles may cause consequential shifts in global material flows, and the systems available for EOL processing of these products will change during their useful life. It may, for example, make sense to use current production inventories for steel, a modified production inventory for aluminum which accounts for new production capacity using marginal energy sources, and projected future recycling rates for both materials (note that with an EOL recycling approach recovery rates would not change over time). LCAs must be explicit about assumptions underlying the assessment and the rationale behind the choices, so comparisons across LCAs are consistent. Additionally, if temporal factors are used for one material type, similar efforts must be taken to apply that methodology to the other materials if comparisons are to be made. There is some confusion in the industry-based LCAs, particularly with inconsistencies in energy mixes from the aluminum industry reports and recycling rates in the steel industry reports that favor their relative life cycle energy GHG emissions and energy use reductions, but make it harder for manufacturers, researchers, consumers, and regulators to understand the current state of mass reduction technologies.

The LCAs reviewed here focused primarily on GHG and energy impacts but the broader literature on vehicle mass reduction also discusses factors beyond the scope of most LCAs. Some researchers express a desire to gain greater perspective on the industry using social and economic methods to evaluate the larger sustainability of mass reduction technologies. These assessments may identify different optimal materials for reducing vehicle mass than LCAs focused on energy consumption and GHGs (Attri and Grover, 2015; Liu and Müller, 2012; Mayyas et al., 2012a; Mayyas et al., 2013; Pallaro et al., 2015; The Aluminum Association, 2011). While the research community continues to refine methods for assessing material options, the industry is focused on meeting environmental standards, market price points, and most importantly, expectations of safety. Current U.S. policies for light-duty vehicles are focused on tailpipe emissions, which are the primary contributor to the life-cycle impacts of currently available light-duty vehicles. However, there is increasing

awareness of life-cycle emissions within regulating agencies (Hula et al., 2016). As the existing policies guide the reduction of use-phase impacts, vehicle production and EOL will become even more important in reducing overall impacts from the automotive sector, and LCA plays an important role in guiding the development for the fleet of the future.

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Fig. 1. Energy consumption for material production, manufacturing, and use for a light-duty vehicle now and trends for the future.

Total life-cycle energy use will decrease as the proportion of impacts for the material production and manufacturing phases are expected to increase and the use-phase impacts are expected to decrease. This is the expected trend with substitutions of more intensive but lighter materials such as aluminum alloys or carbon fiber replacing less intensive but heavier materials such as mild steel. Stronger, higher strength steels are also expected to displace mild steels by reducing total volumes required. Values for the current chart are a simplified breakdown based on Keoleian and Sullivan (2012), Sullivan and Cobas-Flores (2001), and Sullivan et al. (1998).

Fig. 2. Vehicle curb weights and material contributions for average vehicles.

Material composition based on curb weight for average vehicles in model years 2010 and 2013 (Ducker Worldwide, 2015) shows little change in vehicle composition. The 2013 Ford Fusion is an example of an AHSS-intensive vehicle available in the consumer market. The Mach1 lightweight concept vehicle achieved aggressive mass reduction, primarily with aluminum (Bushi et al., 2015). AHSS= advanced high-strength steel, UHSS = ultra-high-strength steel.

Fig. 3. Percent change in fuel economy, weight, and horsepower of light-duty vehicles from 1975 to 2015.

Adapted from 'Light-Duty Automotive Technology, Carbon Dioxide Emissions, and Fuel Economy Trends: 1975 Through 2016' (Hula et al., 2016).

Fig. 4. Generic life-cycle system diagram for vehicles or vehicle components.

The three phases of the life cycle, are shown with different contributing life-cycle processes. Energy is applied to each process along the life cycle, and emissions result from the collective activity.

Fig. 5. Lifetime vehicle distances for use-phase modeling in the selected LCA studies.
The dashed line represents the regulatory useful life as defined by USEPA, which is 241,000 km (150,000 miles).

Table 1.

Selected LCAs for vehicle mass reduction since 2010.

Publications by year including factors that help to define the scope and boundaries for each study. The majority of the studies included production, use, and EOL phases to capture the full cradle-to-grave impacts associated with reducing vehicle mass. The studies by Li and Tharumarajah did not include the EOL phase in their assessments (Li et al., 2015; Tharumarajah and Koltun, 2010).

First Author	Year	Publication	Country / Affiliation / Funding	Explicitly ISO compliant	Component Specific	Component(s)	Body-in white explicit	Steel	Al	Mg	Composites
Simões	2016	Journal of Cleaner Production	EU / University & Bosch / Nat. & Bosch	y	n	-	n	y	y	n	y
Steel Market Dev. Inst.	2016	Report	USA/ Industry Group / -	n	n	-	n	y	y	n	n
Bushi	2015	SAE International	USA/ Ford & Magna / -	y	n	-	n	y	y	y	y
Ding	2015	Journal of Industrial Ecology	China / Nat. / Nat.	y	y	various	n	y	y	n	n
Kelly	2015	Environmental Science and Tech	USA/Nat. Lab/DOE	y	n	-	n	y	y	y	y
Li	2015	Resources, Conservation, and Recycling	China / Nat. / Nat.	n	n	-	n	y	y	y	n
Raugei	2015	Journal of Cleaner Production	U K / U niversity / Research grant	y	n	-	n	y	y	y	y
Das	2014	SAE International	USA/Nat. Lab/-	y	n	-	y	y	y	n	n
Dhingra	2014	Journal of Cleaner Production	USA / University & Nat. Lab/-	n	y	engine	n	y	y	y	n
Lewis	2014	Applied Energy	USA/University/CERC, DOE	n	n	-	y	y	y	n	n
Ehrenberger	2013	Report	EU / Institute / -	y	y	steering wheel	n	n	y	y	n
Koffler	2013	Int Journal of Life Cycle Assessment	USA / Consultant / -	y	y	step assist, front end bolster	n	y	n	n	y
Baroth	2012	SAE International	India / GE & SABIC / -	y	y	fender	n	y	n	n	y
Dubreuil	2012	SAE International	Global / Universities & Labs / -	y	y	front end	n	y	y	y	n
Marretta	2012	Conference on LC Engineering	EU & USA/ Universities & GM/-	n	y	fender, hood	y	y	y	n	n
Mayyas	2012	Energy	USA/ University/-	y	n	-	y	y	y	y	y
PE International	2012	Report	USA/Consultant/ ACC	y	y	step assist	n	y	n	n	y
PE International	2012	Report	USA/Consultant/ ACC	y	y	bolster	n	y	n	n	y

First Author	Year	Publication	Country / Affiliation / Funding	Explicitly ISO compliant	Component Specific	Component(s)	Body-in white explicit	Steel	Al	Mg	Composites
Stasinopoulos	2012	Int Journal of Life Cycle Assessment	AU / University / Nat.	n	n	-	y	y	y	n	n
Das	2011	Int Journal of Life Cycle Assessment	USA/Nat. Lab/-	n	y	floor pan	n	y	n	n	y
Tempelman	2011	Transportation Research Part D	EU / University /-	n	n	-	y	y	n	n	y
Witik	2011	Composites: Part A	EU / University / Research Network	y	y	rear bulkhead	y	y	n	y	y
Du	2010	Journal of Cleaner Production	China / University / Ford & Shanghai	y	n	-	n	y	n	y	n
Du	2010	Energy	China / University / Ford & Shanghai	n	n	-	n	y	y	n	n
Kim	2010	Journal of Industrial Ecology	USA / University / NSF & Alcoa	n	n*	-	y	y	y	n	n
Tharumarajah	2010	Resources, Conservation, and Recycling	AU/Nat/-	y	y	instrument panel	n	y	n	y	y
81% in peer reviewed journals				62%	46%		31%	96%	69%	42%	50%

Table 2.

Critical factors identified in the 26 mass reduction LCAs reviewed.

Critical factor	Phase(s) Influenced	Influence on results
Secondary mass reduction	Use	Not all studies include secondary mass reduction. Materials with higher production impacts may need to reduce the vehicle mass enough to trigger systemic changes in weight dependent systems, providing additional reductions
Recycling allocation	Production & EOL	Lower impacts associated with the EOL allocation approach, more so for materials with higher production impacts, because it gives credit for all of the recyclable material generated by a product
Powertrain reduction and FRVs	Use	Materials with higher production impacts depend on meeting thresholds which allow for downsizing of the powertrain (e.g., reducing cylinder displacement and/or cylinder count while maintaining performance) to achieve greater efficiency during use phase in order to have an lower overall life cycle impact
Material substitution rates	Production	Is used to calculate how much of one material must be used to replace another in lieu of component specific data where actual material masses are known
Lifetime vehicle travel distances	Use	Determines the assumed lifetime of the vehicle based on driving distance, directly related to tailpipe emissions: Longer ranges favor lighter weight materials (e.g., aluminum), shorter ranges favor lower production impact materials
Production location and grid mix	Production	Location of material production is critical for energy intensive metallurgical (e.g., aluminum) processes due to upstream energy generation (e.g., hydroelectric vs coal) and the associated GHG emissions

Table 3.
Common factors included in LCAs for vehicle mass reduction and the chosen assumptions for the selected studies.

The values in the table are adapted from each study and some values required unit conversion. The 'Estimated Secondary Mass Reduction Factor' presents the secondary mass reductions divided by initial mass reductions. 'FRV/Powertrain Reduction' presents the fuel reduction value (FRV) in the format L/(100 km * 100 kg). However, some studies provided other methods of accounting for increased efficiency due to vehicle mass reductions, which were included. An 'x' indicates that the information could not be identified through the review process. For 'Estimated Secondary Mass Reduction Factor' and 'FRV/Powertrain Reduction,' the studies that indicated that these were accounted for but did not provide a specific value are marked with a 'y' while an 'n' indicates that the factor was not used in the study. Similarly, for 'Material Substitution Rates,' studies were marked with 'unknown' when these values could not be determined. 'Recycling Allocation' refers to an end-of-life (EOL) or recycled content (RC) approach and is based on the terms used by Kim and Wallington (2013a). 'Material Selection' describes how the types and mass of materials were determined for each assessment. For several factors the source of the values was evaluated; 'Actual' means the study was based on vehicles or parts that were built and tested, 'Theoretical' means the study used calculated estimates to determine feasible mass reduction factors for vehicles or parts, and 'Hypothetical' means the study compared light-weight materials independently from their application within a vehicle.

First Author	Year	Material Source Region	Manufacturing Region	Market Location	Vehicle Useful Life (km)	Estimated Secondary Mass Reduction Factor	FRV / Powertrain Reduction	Recycling Allocation	Material Substitution Rates	Material Selection
Simões	2016	x	Portugal and Singapore	x	200,000	n	0.15, 0.35 L/(100 km*100 kg)	EOL	various	Theoretical
Steel Market Dev. Inst.	2016	USA	USA	USA	245,000 cars 290,000 ldt	0.05 to 0.25	various	EOL	0.70 to 0.75 (AHSS/steel) 0.60 to 0.70 (Al/steel)	Theoretical
Bushi	2015	USA and Global	USA and Global	USA	250,000	n	0.40 L/(100 km*100 kg)	EOL	various	Actual
Ding	2015	China	China	China	200,000	0.50, 0.23	y	EOL	various	Theoretical
Kelly	2015	x	x	x	260,000	n	0.15 to 0.5 L/(100 km*100 kg)	none	various	Hypothetical
Li	2015	Shanxi/Shaanxi China (Mg)	x	x	200,000	n	n	RC	0.50 (Mg/steel)	Theoretical
Raugei	2015	China (Mg), Global (Al)	Europe	Europe	150,000	n	y	EOL	0.70, 0.60, 0.50 (composite, Mg, Al/steel)	Theoretical

First Author	Year	Material Source Region	Manufacturing Region	Market Location	Vehicle Useful Life (km)	Estimated Secondary Mass Reduction Factor	FRV / Powertrain Reduction	Recycling Allocation	Material Substitution Rates	Material Selection
Das	2014	USA	USA	USA	250,000	y	0.38 L/(100 km*100 kg)	EOL	various	Theoretical
Dhingra	2014	x	x	x	193,121	0.50	+1 mpg	EOL	0.66, 0.67, 0.70 (Mg/Al)	Theoretical
Lewis	2014	x	x	x	257,495	0.54 to 0.70	y	RC	unknown	Hypothetical
Ehrenberger	2013	China (Mg), Europe (Al)	x	x	200,000	n	0.35 L/(100 km*100 kg)	EOL	none	Theoretical
Koffler	2013	US (composites), Global (steel)	USA	x	241,401	n	0.16, 0.38 L/(100 km*100 kg)	EOL	0.51, 0.54 (GFRP/steel)	Actual
Baroth	2012	Global (steel) Europe (composites)	Europe	Europe	193,000	n	n	EOL	0.46 (Al/steel)	Actual
Dubreuil	2012	China (Mg), US (steel), Global/US (Al)	N. America	N. America	200,000	n	0.39 L/(100 km*100 kg)	EOL	0.55 (Mg/steel) 0.74 (Al/steel)	Theoretical
Marretta	2012	Global	USA	x	200,000	n	n	RC	0.60, 0.65 (Al/steel)	Theoretical
Mayyas	2012	x	x	USA	321,869	n	n	EOL	various	Theoretical
PE International	2012	USA	USA	USA	241,401	n	0.16, 0.38 L/(100 km*100 kg)	EOL	0.49 (composite/steel)	Actual
PE International	2012	USA	USA	USA	241,401	n	0.16, 0.38 L/(100 km*100 kg)	EOL	0.54, 0.63 (composite/steel)	Actual
Stasinopoulos	2012	Australia	Australia	x	300,000	n	n	RC	0.70 (Al/steel)	Hypothetical
Das	2011	x	x	x	200,000	0.50	y	EOL	0.83 (composite/steel)	Theoretical
Tempelman	2011	x	x	Europe	225,000	0.50	y	RC	various	Hypothetical
Witik	2011	China (Mg), Europe (steel)	Europe	Europe	200,000	n	0.34 to 0.48 L/(100 km*100 kg)	EOL	0.31, 0.40, 0.41, 0.43 (composite/steel) 0.38 (Mg/steel)	Theoretical
Du	2010	China (Mg)	x	x	200,000	0.28	0.5 to 2.0 L/100 km	RC	0.50 (Mg/steel)	Theoretical

First Author	Year	Material Source Region	Manufacturing Region	Market Location	Vehicle Useful Life (km)	Estimated Secondary Mass Reduction Factor	FRV / Powertrain Reduction	Recycling Allocation	Material Substitution Rates	Material Selection
Du	2010	China (Al)	China	China	x	n	0.3 to 1.1 L/100 km	none	unknown	Hypothetical
Kim	2010	x	x	x	193,121	n	3.0% to 8.2% fuel-savings	EOL	unknown	Theoretical
Tharumarajah	2010	China (Mg), USA (composites), World (Fe/Steel)	USA	USA	200,000	n	n	RC	0.75, 0.83 (composite/steel) 0.69 (Mg/steel)	Theoretical

Actual

Theoretical

Hypothetical